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Planning for action: Bridging the gap between systematic conservation planning and conservation action

Thesis submitted by
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Bachelor of Marine Studies (Hons) University of Queensland

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for the degree of Doctor of Philosophy
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Abstract

To halt the decline of marine biodiversity, networks of interacting marine protected areas (MPAs) – intertidal and subtidal areas “reserved by law or other effective means to protect part or all of the enclosed environment” (Kelleher 1999) – need to be expanded. Systematic conservation planning (hereafter ‘systematic planning’) offers a way forward with its explicit methods for locating and designing resource management (hereafter ‘management’) in time and space to promote the conservation of biodiversity (Margules and Pressey 2000). The implementation of systematically planned MPA networks has been demonstrated in some regions. However, numerous challenges (e.g. understanding the willingness of people to engage in conservation) impede translations of many spatial prioritisations into management. Spatial prioritisations are the technical activities within systematic planning that identify the configuration of priority areas for conservation action. Conservation actions are interventions that contribute to conservation goals (e.g. establishing education programs or management) (Salafsky et al. 2008). The failure of many spatial prioritisations to motivate conservation action is referred to as the knowing-doing gap.

The conservation biology literature contains a heated discussion about the best investment of conservation resources, leading to a polarization between systematic planning and opportunistic conservation – conservation that takes advantage of opportunities without considering spatial context or regional conservation goals. Although it can be useful to polarize these perspectives to better understand their respective strengths and limitations, academics and resource managers are now exploring how they can be made complementary. Opportunistic conservation actions can be ‘scaled up’ to better achieve fisheries and conservation objectives that require perspectives broader than individual local governance units. At the same time, spatial prioritisations must be ‘scaled down’ or adapted to better inform implementation of conservation actions by incorporating local objectives, unforeseen constraints on conservation actions, and errors in data. The goal of this thesis is to better

understand options for integrating systematic planning with local management. With this goal in mind, my thesis has two main objectives:

1. To investigate methods for scaling down systematic planning to inform conservation actions, focusing on opportunities for implementing multiple forms of management with different contributions to conservation goals; and
2. To explore considerations for scaling up conservation actions to achieve regional conservation goals.

To achieve these objectives, first I examine the mismatch of spatial scales between systematic planning and conservation actions. I review key decisions about spatial scale in systematic planning, and the considerations and implications of these key decisions for informing conservation actions (Chapter 2). In Chapter 2, I develop a framework in which decisions about spatial scale can be made explicit, investigated further, and potentially addressed during systematic planning. In this framework, I identify five decisions about spatial scale: extent and delineation of the planning region, resolution of data, size and delineation of planning units, MPA network design, and applying conservation actions. Each of these decisions involves several considerations, including the extent of available data, extent of bioregions, and social, economic and ecological characteristics of study areas. My framework helps to link theory and application in systematic planning, facilitates learning, and promotes the application of conservation actions that are both regionally and locally significant.

To scale up conservation actions or scale down systematic plans, the differential contributions of several forms of management (e.g. permanent and temporary closures to resource extraction) to conservation goals must be understood, so I develop a method to do so (Chapter 3). Using Fiji as a case study, I gather expert knowledge through dialectic inquiry to obtain perceived effectiveness scores for four forms of management, and use these in a national gap analysis. Permanent closures were the benchmark with an ecological effectiveness score of 1.0. Temporary closures with controlled harvesting had relatively high scores and temporary closures with uncontrolled harvesting and ‘other management’ had relatively low scores. Understanding the relative contribution of different forms of management to conservation goals facilitates scaling up and down in three ways: (1) forms of management that complement each other in terms of the level of protection they offer and their social acceptability can be identified; (2) conservation achievements in countries where multiple forms of management will be needed to achieve national conservation goals can be assessed; and (3) spatial prioritisations can be tailored to management that is relevant within the ecological, social, economic and political context of the selected planning regions.

To contribute to existing knowledge on opportunities for implementing multiple forms of management, I develop a method to model conservation opportunity at fine resolution for different forms of management (Chapter 4). I also develop an approach to investigate the social characteristics of villages with different forms of management, thereby providing insights into conservation opportunity (Chapter 5).

In Chapter 4, I use key informant interviews and remotely collected data, and model conservation opportunity for different forms of management at regional scales using Maxent. This model provides information on the relative suitability of one area for a particular form of management. I find that two of the most important predictors of suitability for the different forms of management are distance from the nearest road and proportion of inshore fishing ground already closed. This approach is promising, because it produces good fits to the existing data (cross-validated AUC at least 0.98). It also provides insight into the factors influencing the presence and characteristics of different forms of management, and matches accounts in the literature on factors important to establishing closures.

In Chapter 5, I use the social-ecological systems diagnostic framework, and compare the performance of data at different resolutions for informing conservation opportunity. Even though conservation opportunity is context-specific, using a well-recognized diagnostic framework allows identification of characteristics that lead to effective governance within different contexts. I use canonical correspondence analysis to examine the association between the presence and form of management on one hand and, on the other, human and social characteristics of villages. I find that, in the Solomon Islands, human and social characteristics that influence the presence and absence of management can be more easily differentiated than those related to different forms of management. Furthermore, I find that household-scale data are more effective than village-scale data at identifying the human and social characteristics associated with management. Understanding these characteristics and mapping conservation opportunity facilitate the scaling down of systematic plans by informing planners of priorities for feasible forms of management within a social-ecological system.

To further explore considerations for scaling up conservation actions to achieve regional conservation goals (objective 2), the conservation opportunity model developed in Chapter 4 is used as the basis for comparing systematic planning and opportunistic approaches to conservation. I carry out 10-year simulations of additional conservation actions with systematic planning and opportunistic conservation approaches, identifying the difference between the upper and lower bounds of plausible future conservation achievements (Chapter 4). To predict future conservation action, I use data on conservation opportunity (Maxent suitability model), established MPAs, key informant interviews, and Marxan with Zones (systematic planning software). The opportunistic approach achieves quantitative conservation objectives for half the ecosystems, while all objectives are achieved or nearly achieved with the systematic planning approach. Chapter 4 informs policy makers about what incentives and regulations are needed to steer Fiji toward achieving national conservation goals into the future.

My thesis contributes to the theoretical advancement of the field of conservation biology by investigating the results of different approaches to conservation (i.e., systematic planning and opportunistic approaches) and developing methods that help integrate them. Chapter 2 informs both scaling up and scaling down by identifying the considerations needed and the trade-offs between considerations when making decisions about spatial scale. Chapter 3 provides a method to understand the differential effectiveness of management and integrates this understanding into a national gap analysis, which facilitates scaling down of outputs from systematic plans by tailoring them to specific regions. Understanding differential effectiveness also helps to scale up management by informing decision makers about complementary forms of management. Together, chapters 4 and 5 provide methods to understand opportunity for conservation. This is critical if spatial prioritisations are to identify priority areas that are most likely to be implemented, thereby facilitating the scaling down of systematic plans. Lastly, Chapter 4 also demonstrates a method to understand the benefits of coordinating opportunistic management to facilitate the development of policies and incentives.

Glossary

Term	Definition
Bioregions	Areas with relatively homogeneous biological and physical composition, distinct from adjacent regions and large enough to ecological and evolutionary processes (Spalding et at 2007).
Conservation actions	Interventions that contribute to conservation goals (e.g. establishing education programs or management) (Salafsky et al. 2008).
Conservation objectives	Quantitative interpretations of the broader goals of planning
Ecological effectiveness	The relative contribution of a form of management to realizing conservation objectives.
Implementation strategy	The development of a plan of how spatial prioritisations are going to inform management (Knight et al. 2006b).
Knowing-doing gap	The difference between the number of spatial prioritisations produced and the number that actively informs management. The knowing doing gap is composed of the research-implementation and the planning-implementation gap.
Locally Managed Marine Area (LMMA)	An area of inshore waters governed by local residents and involving a collective understanding of, and commitment to, management interventions in response to threats to marine resources. In terms of management it is equivalent to an MPA.
Management effectiveness	The feasibility of a management approach for a particular biological, social and economic context, and its ability to promote the persistence of all levels of biodiversity (based on Hockings et al 2006).
Martine protected area (MPA)	"Any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher 1999).
Marine protected area networks (MPA networks)	"A collection of individual marine protected areas operating cooperatively and synergistically …to fulfil ecological aims more effectively and comprehensively than individual sites alone could" (WCPA/IUCN 2007).
Opportunistic conservation	Conservation that takes advantage of opportunities without considering spatial context or regional conservation goals.
Permanent closure	Areas where the extraction of resources is prohibited indefinitely (a.k.a. no-take areas, marine reserves).

Planning units	Spatial units of assessment and comparison used in most planning exercises that employ decision-support software and are the building blocks of regional MPA designs.
Planning region	The geographic domain within which areas are evaluated and compared as candidates for conservation action.
Protected areas	Areas “designated or regulated and managed to achieve specific conservation objectives” (CBD 2008).
Resource management (management)	Any action directed at protecting, enhancing or restoring species and ecosystems.
Regional priorities	Areas identified for generic or specific conservation actions during regional scale design.
Regional scale	An area which shares common “patterns and processes of biodiversity and human uses” (Pressey and Bottrill 2009).
Spatial scale (scale)	The extent and resolution of study regions, data, and areas of assessment.
Spatial prioritisation	The “technical activities that identify the location and configuration of priority areas for conservation action” (Knight et al. 2006b).
Systematic conservation planning (systematic planning)	An explicit operational model for locating, designing and implementing conservation actions in time and space to promote the conservation of biodiversity and sustainable use of natural resources (Margules and Pressey 2000).
Temporary closure	Areas where the extraction of resources is prohibited temporarily. These can be ‘controlled’, where harvests are allowed once per year as dictated by a management plan or collective decision at the community level; or, uncontrolled, where areas are harvested without any predefined frequency and duration.

Chapter 1

General Introduction

1.1. Context and rationale for thesis

1.1.1. Marine biodiversity crisis

A growing human population is putting increasing demands on natural resources, often leading to unsustainable resource use, resulting in ecosystem degradation (for reviews see Pandolfi et al. 2003), and declines in marine biodiversity, fisheries and other ecosystem services (Myers and Worm 2003; Sala and Knowlton 2006). Unless we mitigate this decline, we risk losing marine species and entire marine ecosystems within a single generation (Rogers and Laffoley 2011), and ultimately endangering the livelihoods of people. Until recently, management of marine systems was focused primarily on single species important for fisheries. However, single-species fisheries management is insufficient to mitigate declining marine biodiversity, fisheries and other ecosystem services (Pauly et al. 2002). Consequently scientists have been calling for the complementary establishment of ecosystem-based management (Pauly et al 2002). Ecosystem-based management is informed by knowledge of ecological processes that underpin ecosystem composition, structure and function so that these can be sustained (Christensen et al. 1996). The most popular component of marine ecosystem-based management, although not a panacea for the marine biodiversity crisis, is the implementation of marine protected areas (MPAs) (Pauly et al. 2002; Sala and Knowlton 2006).

An MPA is “any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment” (Kelleher 1999). The implementation of MPAs depends on the growing number of people who compete to use marine resources, and who could also be negatively affected by the establishment of MPAs. Consequently, an intimate and detailed understanding of peoples’ attitudes, values and behaviours will be crucial in designing and implementing networks of MPAs. Ultimately, human behaviour must be motivated to change if biodiversity is to be protected (Mascia et al. 2003).

This thesis focuses on the implementation of MPA networks informed by systematic conservation planning (hereafter ‘systematic planning’). Systematic planning is an explicit operational model for locating, designing and implementing conservation actions in time and space to promote the conservation of biodiversity and sustainable use of natural resources (Margules and Pressey 2000). Conservation actions are defined as interventions that contribute to conservation goals (e.g. establishing education programs or management; Salafsky et al. 2008). One of the major gaps in the existing body of knowledge in systematic planning is how to translate outputs from systematic plans to effective on-ground or in-water resource management (Pierce et al. 2005; Knight et al. 2008). I address this major challenge by examining approaches for better informing resource management with systematic planning. The broad aim is to guide and implement management that is both locally acceptable and ecologically functional. Below I introduce MPAs and MPA networks, summarize marine conservation accomplishments to date, and describe the advantages and disadvantages of existing approaches to achieving conservation goals.

1.1.2. Marine protected areas

MPAs can encompass numerous forms of resource management (hereafter ‘management’) simultaneously. Management includes any action directed at protecting, enhancing or restoring species and ecosystems. Examples are: permanent closures (a.k.a. no-take areas, marine reserves), where the extraction of resources is prohibited indefinitely; temporary closures, where harvest is allowed some of the time; and restrictions on the types of species that can be fished, gears that can be used for fishing, or the amount of catch allowed (Kelleher 1999).

Of the various forms of marine management, the benefits of permanent closures are most studied, and include: protecting vulnerable species and sensitive ecosystems (e.g. Roberts et al. 2005), increasing the abundance, density and diversity of species (e.g. Halpern 2003; Russ and Alcala 2003; Russ et al. 2008), restoring community structure (e.g. Micheli et al. 2004) and increasing the ecologic resilience of fish communities (e.g. Babcock et al. 2010). Areas surrounding permanent closures can also benefit because of a net export of adult and larval fish to adjacent areas (Abesamis et al. 2006; Almany et al. 2009). The extent of individual permanent closures limits the potential benefits of management. Consequently, it is suggested that connected networks of MPAs should be established.

Guidelines for MPA networks are fairly well developed. An MPA network is “a collection of individual MPAs operating cooperatively and synergistically …to fulfil ecological aims more effectively and comprehensively than individual sites alone could” (WCPA/IUCN 2007). This allows MPAs to protect species and processes that move or operate across a range of spatial extents (Roberts et al. 2003). MPA networks should be comprehensive, adequate and representative (Ballantine 1997; Stewart et al. 2003). Comprehensive means MPAs encompass the full range of biodiversity, including species composition and community structure and function, e.g. habitats required by a particular species at different life stages (Ballantine 1997; Stewart et al. 2003). Adequate implies that the degree of protection is sufficient to ensure long-term viability of species, suggested to be 20-30% of the area of bioregions, ecosystems and habitats (Ballantine 1997; Stewart et al. 2003). The percentage area required for adequacy depend, however, on the size and quality (i.e., intactness) of the managed areas and the ecology of the target species (Crowder et al. 2000; Agardy et al. 2003). Representative implies that all the biotic diversity within the geographic domain within which areas are evaluated for conservation action is sampled (Ballantine 1997). Consequently, managers should seek to identify areas for permanent closures that are complementary (i.e., areas that contribute biota that are underrepresented within the existing protected areas), so a combination of the species contained within the protected areas will together sample an ecologically viable population of the identified species (Stewart et al. 2003). To ensure species persistence (e.g. survival against local catastrophic events), multiple representations of species across the MPA network are recommended (Ballantine 1997), as is connectivity among permanent closures (Allison et al. 1998; Almany et al. 2009). Connected areas are those that allow for the "exchange of individuals between populations", for example though the movement of planktonic larvae (Almany et al. 2009).

Management other than permanent closures can also contribute to the protection of species, but there is a paucity of knowledge on their relative contribution to conservation goals. McClanahan et al. (2006) found that, in areas where enforcement of restrictions of harvest is poor, management driven by community goals (including temporary closures and gear restrictions) can be more effective at protecting target fish species than permanent closures. Bartlett et al. (2009) also suggested that temporary closures can be as effective at protecting target reef species as permanent closures in small communities in Vanuatu. Increased understanding of the relative contribution of different forms of management to biodiversity conservation goals is critical to assess how they can be employed to complement one another in different contexts (e.g. Mora et al. 2006).

Regardless of the form of management, people will determine whether, and how, management is implemented and respected (Mascia et al. 2003). While numerous studies have focused on the design of MPA networks that allow biodiversity to persist through time, we know much less about the social factors that motivate people to support the establishment of MPAs. Several social characteristics have been identified that engender support for effective management in different contexts. For example, in the Pacific, if the management proposed has affinity with cultural traditions (Johannes et al. 2000) or clearly defined resource ownership (Aswani 2005), it is more likely to be supported. However, the study of these characteristics are often context-specific and understanding which social characteristics can promote different forms of management, and contribute to conservation goals, is still at its infancy (Ostrom 2007).

1.1.3. Marine conservation accomplishments to date

Scientists and country leaders have reached a consensus on the importance of establishing MPAs to mitigate biodiversity decline, but limited progress has been made in establishing them. For example, in 2003 at the 5th World Parks Congress, participants called on the international community to establish by 2012 a network of MPAs enforcing permanent closures across 20-30% of coastal and marine ecosystems. Furthermore, 168 nations have signed the Convention on Biological Diversity, which commits them to managing a portion of their marine areas to halt biodiversity declines. However, a global marine gap analysis of all ecosystems found that MPAs cover 0.65% of oceans, of which only 0.08% comprise permanent closures (Wood et al. 2008). This lack of progress towards broad conservation goals is thought to be partly a result of the failure to identify how to implement terrestrial or marine protected areas where they are most needed (Whitten et al. 2001; Redford and Sanjayan 2003). There is continuing debate about the most effective approach to establishing protected areas (Knight and Cowling 2007; Pressey and Bottrill 2008; Smith et al. 2009; Noss 2010). Below, I introduce two conservation approaches, an opportunistic and a systematic approach. For the purpose of this thesis, I define them as mutually exclusive, although some blend of the two, depending on socio-political circumstances, will usually be needed to establish effective MPA networks.

1.1.4. An opportunistic approach to achieving conservation goals

Most terrestrial and marine conservation actions have been implemented in an ad hoc way, taking advantage of opportunities but lacking spatial context (Pressey et al. 2000; Rouget et al. 2003; Scott et al. 2001; Weeks et al. 2009). Many conservation actions were not established specifically for achieving conservation goals. Instead they were established to accumulate resources for festivities or for recreational or cultural values (e.g. Govan 2009; Johannes 2002). This is also referred to as opportunistic conservation (Pressey et al.

1996). In many regions, opportunistic conservation actions focus on local goals (e.g. protection of sacred sites), integrate local knowledge and customs, and involve local resource users, empowering them to implement management. Such characteristics are seen as the key to effective management (Berkes 2004; Brown 2002; Johannes et al. 2000; Rodriguez et al. 2007; Smith et al. 2009). Opportunistic conservation actions are not necessarily locally motivated; they also result from expediency on the part of governments and non-governmental organizations (NGOs). For example, many national parks are on rugged or barren lands of low commercial value, which minimizes resistance to implementation (Pressey, 1994; Pressey et al. 2000; Pressey and Tully 1994) but also minimizes their value in mitigating loss of biodiversity. Regional conservation goals, e.g. comprehensive and representative sampling of a nation's biodiversity within protected areas, are generally not considered with opportunistic conservation action.

Theoretically, opportunistic actions can coalesce into MPA networks that achieve regional conservation goals as well as local conservation goals (Chomitz and Buys 2007). However, they can also waste limited conservation resources by focusing on areas that duplicate, or contribute marginally, to regional conservation goals (Pressey and Tully 1994). For example, a recent gap analysis of locally-driven opportunistic conservation in the Philippines examined the coverage of 985 MPAs and found that while MPAs can have satisfied local fisheries goals (e.g. Alcala and Russ 2006), they were biased in protecting mapped bioregions, with marine biodiversity corridors rarely protected (Weeks et al. 2010). Consequently, they did not achieve national goals to protect biodiversity (Weeks et al. 2010). Systematic planning was developed as an approach to guide conservation and counter the biased representation of ecosystem types, often found where there is an opportunistic approach to conservation (Margules and Pressey 2000).

1.1.5. A systematic approach to achieving conservation goals

Systematic planning is an explicit operational model for locating, designing and implementing management in time and space to promote the conservation of biodiversity and sustainable use of natural resources (Margules and Pressey 2000). Systematic planning can be undertaken at a range of scales, for prioritising global conservation action to prioritising the implementation of resource management by single villages. Systematic planning is characterized by the use of explicit objectives, considerations of spatial context through complementarity and connectivity, collaboration with stakeholders (including resource managers), and the development of a strategy in guiding implementation of conservation actions. I define conservation objectives as quantitative interpretations of the broader goals of planning. If a goal is representation of marine biodiversity, one subsequent objective could be to sample a certain percentage of each mapped ecosystem within permanent closures. Systematic planning ensures conservation actions protect the full range of biodiversity and ecosystem processes that enable their persistence through time (Margules and Pressey 2000).

The process of systematic planning is well established and has been progressively fine-tuned and adapted to different contexts. Systematic planning involves: a spatial prioritisation of areas for conservation (hereafter 'spatial prioritisation', i.e. "technical activities that identify the location and configuration of priority areas for conservation action" (Knight et al. 2006b)); the development of implementation strategies, i.e. "how conservation initiatives are undertaken" (Knight et al. 2006b) and the implementation, management and monitoring of management effectiveness (Margules and Pressey 2000; Knight et al. 2006a; Pressey and Bottrill 2009). Historically focused on biological and ecological imperatives, the process of systematic planning has expanded to emphasise the importance of stakeholder collaboration and social and economic considerations (Knight et al. 2006a; Pressey and Bottrill 2009).

Intuitively, systematic planning is a more strategic approach than the implementation of opportunistic actions and should allocate actions more efficiently to achieve objectives. To date, many millions of dollars have been invested in its application. The planning process in Kimbe Bay for example, one of hundreds of planning processes by The Nature Conservancy (TNC), cost US$400 000 (Green et al. 2007). However, few systematic plans have led to extensive on-ground implementation (e.g. Cabeza and Moilanen 2001; Knight et al. 2006a; Knight et al. 2008; Prendergast et al. 1999) and the reported exceptions (e.g. Airame et al. 2003; Fernandez et al. 2005; McCook et al. 2010) were facilitated by atypical circumstances, such as lack of private tenure and uncomplicated governance.

The difference between the number of systematic plans (or parts of systematic plans) undertaken and the number that actively informs management is referred to as the knowing-doing gap (Knight et al. 2006b). According to Knight et al (2006b), in the context of systematic planning, the knowing-doing gap is composed of the research-implementation and the planning-implementation gap. The research-implementation gap is the divide between the spatial prioritisation and the process of developing an implementation strategy. And the planning-implementation gap is the divide between the spatial prioritization or the process of developing an implementation strategy and the implementation of management (Figure 1.1).

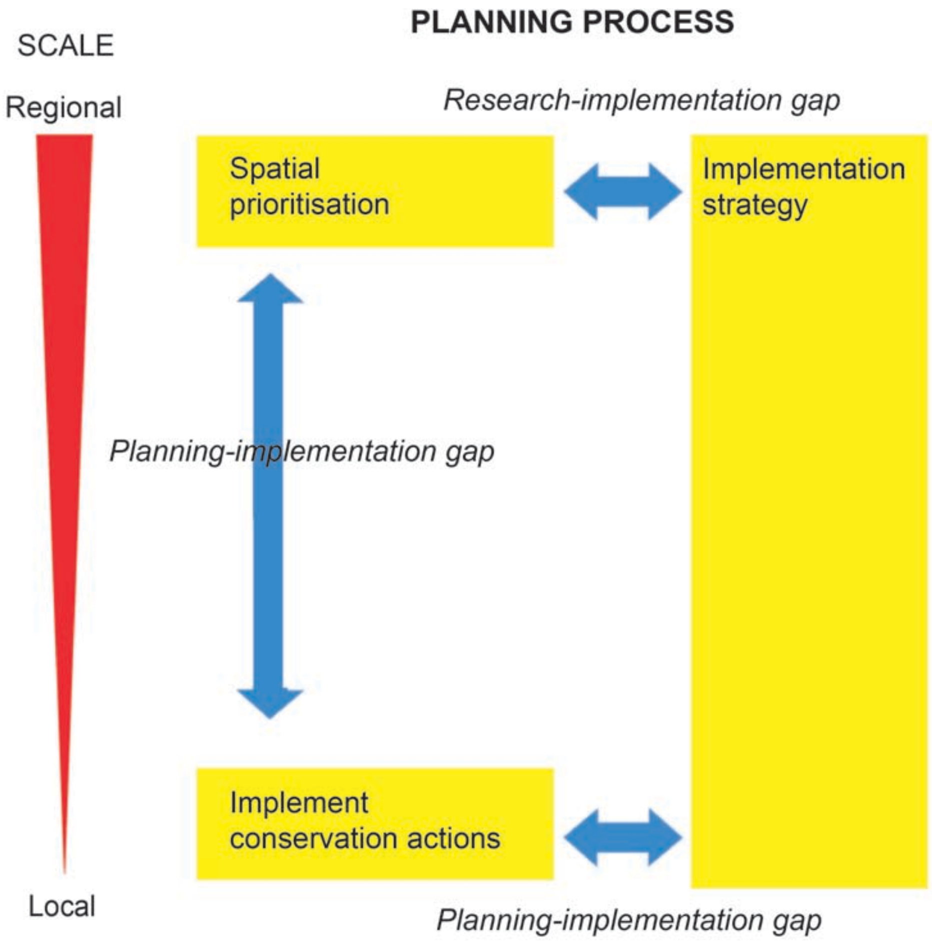

Figure 1.1. A schematic of the 3 main parts of the planning process and knowing-doing gaps. The 3 main parts of the planning process are represented by the yellow boxes, and include; (1) undertaking the spatial prioritisation, (2) developing the implementation strategy, and (3) implementing conservation actions. The blue arrows between the stages represent the gaps that together make up the broader knowing–doing gap (Pfeffer and Sutton 1999). The extent of focus narrows as planners move from undertaking spatial prioritisations to implementing management. Modified from Knight et al. (2006a).

The knowing-doing gap can be attributed to a variety of interlinked challenges, including: (1) the scale mismatch between systematic planning and conservation actions (Briggs 2001); (2) limited collaboration between planners and resource managers (Cowling and Pressey 2003; Knight et al. 2008); (3) lack of comprehensive and in-depth social assessments (Knight and Cowling 2007); (4) lack of institutions promoting and supporting dynamic planning processes (Knight et al. 2006a), and; (5) lack of understanding of the optimal suite of conservation actions, incentives and institutions (Ferrier and Wintle 2009).

(1) Scale mismatch between systematic planning and conservation actions

Increasing recognition that protected areas are subject to local and regional threats, and need to be complementary and connected to each other, means that a regional perspective is crucial in planning for effective management (Franklin 1993; Poiani et al. 2000). However, to implement management, a local perspective that incorporates fine-scale knowledge on costs and values ensures that good decisions are made regarding what should be protected (Aswani and Hamilton 2004a; Christie et al. 2009a). Engagement with locals also builds support and ensures compliance with established management (Smith et al. 2009). However, the complexity of extensive planning regions - the geographic domain within which areas are evaluated and compared as candidates for conservation action - often allow for only limited input of fine-scale information and limited collaboration with local stakeholders because of time and resource restrictions. Consequently, local costs and values are often not considered in spatial prioritisations, so conservation priorities identified broadly need to be re-evaluated as areas are visited on the ground and corrections to information are found to be needed. Moving between regional and local spatial scales is essential to inform effective management, but there is little guidance of how to do this (exceptions include Smith et al. 2006; Noss et al. 2002; Knight et al. 2008; Seddon et al. 2010; Game et al. 2011).

(2) Limited collaboration between planners and resource managers

Collaboration between planners and managers is essential for spatial prioritisations to inform management (Knight et al. 2008). Many spatial prioritisations are undertaken as academic exercises only, with little intent of the research being translated into conservation actions (Knight et al. 2008). Consequently the spatial prioritisation does not involve managers and the products of the spatial prioritisation do not address their needs. In systematic planning processes intended to lead to implementation, limited input into stakeholder engagement can result from strategic decisions by those undertaking the systematic plan. Reasons include avoidance of raised expectations about implementation of all designed conservation actions (e.g. Green et al. 2009), or struggle to keep implementation organisation staff trained and informed due to high staff turnover (Knight et al. 2011), or planning fatigue amongst stakeholders (Bottrill et al. Submitted). Furthermore, planners often limit stakeholder engagement because of time and resource constraints (Bottrill et al. Submitted). Even when planners have deliberately limited involvement of stakeholders for reasons they consider valid, their decisions could hinder implementation by reducing buy-in or relevance to day-to-day management decisions (Pierce et al. 2005). More collaboration between planners and managers is therefore necessary to ensure that systematic planning is relevant to managers and other stakeholders.

(3) Lack of comprehensive social assessments

The recognition that a biology-focused assessment is unlikely to result in management, because management depends on people, has led to calls for incorporating comprehensive social assessments into systematic plans (Cowling and Wilhelm-Rechmann 2007; Polasky 2008). Informed and constructive information exchange between conservationists (mostly from a biological background) and social scientists is thought to be the key for advancing conservation goals and improving conservation practice (Redford 2011). Comprehensive inclusion of stakeholders throughout the planning process provides some of the insight into social complexities. However, given the scale of most planning regions and the numbers of stakeholders within them, including all stakeholders in systematic planning is often prohibitively difficult. Additionally, it may only superficially inform conservation scientists about the history and value-ridden conflicts driving conservation challenges. Consequently, social assessments informing on threats to biodiversity, costs of management and opportunities for and constraints on implementation should be used in parallel with stakeholder engagement (Cowling et al. 2003; Cowling et al. 2004; Pressey and Bottrill 2009).

Social assessments are increasingly used in systematic planning but still need to be improved. Most recent work on social assessments for systematic planning has focused on the incorporation of the economic costs of conservation (for review see Ban and Klein 2009; Naidoo et al. 2006). Minimizing costs to local stakeholders is likely to foster compliance of established management and community support (Klein et al. 2008). Some advancement has also been made in understanding threats (Wilson et al. 2005), which allow planners to target areas that are most likely to be degraded. However, research exploring opportunities for conservation action (i.e., the likelihood of effective implementation of conservation actions) is just starting to emerge in the conservation literature (e.g., Game et al. 2011; Guerrero et al. 2010; Knight et al. 2010; Raymond and Brown 2011). The further development of this work is crucial for comprehensive social assessments to guide systematic planning effectively. The rapid advancement of this area requires that more conservation resources be spent in social rather than biological research. The budget, time and effort dedicated to understanding the social drivers and consequences of conservation initiatives on communities is increasing (e.g. MPAs in the Birds Head Seascape).

(4) Lack of institutions supporting dynamic planning processes

Systematic plans need to be updated if they are to inform management through time. Consequently, planning processes must be linked to institutions supporting dynamic planning processes (Knight et al. 2006b; Pierce et al. 2005). Systematic planning is envisioned as a dynamic process with feedbacks and reassessments of priority areas as opportunities and threats change (Pressey and Bottrill 2009). This dynamism allows for the progressive incorporation of fine-scale data and new knowledge, resulting in adjustments to priorities and management, e.g. in relation to new threats, to maximize the benefit from conservation actions (Cowling et al. 2003; Margules and Pressey 2000; Pressey and Bottrill 2009). If plans are not made dynamic, both systematic planning products and the recommendations from systematic planning are likely to become irrelevant or be forgotten in response to, for example, staff turnover in institutions responsible for systematic planning (Knight et al. 2011; Bottrill et al. Submitted). Currently, few institutions support dynamic planning and therefore little is known about how to build institutions and support for such processes (exceptions include Theobald et al. 2000; Knight and Cowling 2006; CMP 2010; Game et al. 2011; Stokes et al. 2010).

(5) Lack of understanding of the optimal suite of conservation actions and incentives to achieve conservation goals.

To achieve conservation goals, multiple forms of management will be needed to suit the social and ecological complexities of planning regions (e.g. Young et al. 1996; Rouget et al. 2006; Wilson et al. 2009). Understanding how to incorporate multiple forms of management into systematic planning, informed by social and ecological variables, is therefore critical to influencing management, and to closing the knowing-doing gap. Historically, spatial prioritisations focused on identifying areas for terrestrial or marine reserves (strict protection, such as permanent closures), providing limited direction to managers working in production landscapes where conservation goals must be balanced with goals driving resource use. More recently, the focus has changed to identifying areas for a mix of conservation actions (e.g. for catchment management or restoration) that play complementary roles in protecting biodiversity (Wilson et al. 2007). Furthermore, new systematic planning software (Marxan with Zones) has been developed, and allows for the consideration of costs and contribution to goals of different forms of management (Watts et al. 2009; Wilson et al. 2009). However, there is still limited understanding of the relative contribution of different forms of management to conservation goals and of which forms of management are most effective, according to social characteristics, within different planning regions (Cinner 2007; McClanahan et al. 2006). Consequently, more studies on how to plan for multiple complementary forms of management using systematic planning are needed.

For the purpose of this thesis, I chose to define systematic and opportunistic approaches as mutually exclusive. However, there are some attempts (in the scientific literature and in practice) to integrate them. Opportunistic actions are being ‘scaled up’ to better achieve fisheries and conservation objectives that require perspectives broader than individual local governance units (e.g. in Danajon Bank, in the Philippines; Armada et al. 2009). At the same time, spatial prioritisations are being ‘scaled down’ or adapted to deal with local objectives, unforeseen constraints on conservation actions, and errors in data (Henson et al. 2009; Knight et al. 2010). Additionally, spatial prioritisations have incorporated elements of opportunistic action such as community preferences, local knowledge and local tenure (e.g. Ban et al. 2009a; Pressey and Bottrill 2009). Scaling down potentially loses some of the theoretical advantages of spatial prioritisations, while also probably increasing the likelihood of implementing conservation actions in selected areas. I chose to compare opportunistic and systematic approaches to conservation to add to this small body of knowledge and better understand how to combine the two.

1.2. Study region

The Coral Triangle (CT) has the most marine biodiversity in the world and a global priority for conservation (Roberts et al. 2002; Green and Mous 2008; CTI Secretariat 2009a). Six countries form the CT: the Philippines, Malaysia, Indonesia, Papua New Guinea, the Solomon Islands, and East Timor. Of about 350 million people living in the region, more than 120 million depend directly on local marine resources (CTI Secretariat 2009a). In 2007, the CT Initiative - the world’s largest conservation initiative - was launched to mitigate threats to marine resources, with over U.S. $500 million committed. Donors include Australia, the Global Environment Facility, and United States Agency for International Development (CTI Secretariat 2010b). Sustainable resource use within the CT depends on integrated management including, importantly, the effective implementation of networks of MPAs (Roberts et al. 2001; White 2008). Fiji and Vanuatu are found just outside the boundaries of the CT and similar to some of the CT countries in marine biodiversity, social characteristics, and governance. Chapters in this thesis will focus either on the Coral Triangle (CT), or individually on the Solomon Islands and Fiji. .

1.3. Goal and objectives of the thesis

Given the urgency of understanding how to implement MPA networks, the goal of this thesis was to investigate options for integrating systematic planning with local management. To achieve my thesis goal, I established two objectives:

1) Investigate approaches for scaling down systematic planning to inform management, focusing on opportunities for implementing multiple forms of management and their contribution to conservation goals.
2) Explore considerations for scaling up conservation actions to achieve regional conservation goals.

Objective 1. Investigate approaches for scaling down systematic planning to inform management, focusing on 1) opportunities for implementing multiple forms of management, and 2) their contribution to conservation goals.

This thesis investigates how to scale down systematic planning to address three of the challenges that contribute to the knowing-doing gap: (1) the scale mismatch between systematic planning and conservation actions; (2) lack of comprehensive social assessments; and (3) lack of understanding of the optimal suite of conservation actions and incentives to achieve conservation goals. To address the scale mismatch between systematic planning and conservation actions, I investigate the key decisions about scale in systematic planning, starting from a regional perspective. I identify the matters needed to be considered (hereafter ‘considerations’) when making key decisions about spatial scale, the trade-offs between these considerations, and the implications of the decisions on-ground (Chapter 2). To address the general lack of comprehensive social assessments and the lack of the optimal suite of conservation actions and incentives to achieve conservation goals, I describe two projects. First, I develop a method to integrate the different contributions of locally relevant forms of management to regional conservation goals (Chapter 3). Second, I develop two methods for understanding and mapping conservation opportunities for multiple forms of management (Chapter 4 & 5). The first involved spatial surrogates, for factors that influence the presence, location and extent of different forms of management, using remotely available data. This research gives maps of opportunities for different forms of management (Chapter 4). The second method for understanding conservation opportunities involved the application of pre-existing diagnostic frameworks for understanding human and social characteristics associated to conservation opportunity and collecting social data at the resolution of households and villages. This research is aimed at better understanding the social characteristics associated with different forms of management (Chapter 5).

Objective 2. Explore considerations for scaling up conservation actions to achieve regional conservation goals.

This thesis also investigates how to scale up management, in an attempt to address three of the challenges that contribute to the knowing-doing gap, including: (1) the scale mismatch between systematic plans and conservation actions; (2) lack of understanding of the optimal suite of conservation actions and incentives to achieve conservation goals; and, (3) lack of institutions supporting dynamic planning processes. The framework I developed in Chapter 2, to address the scale mismatch between systematic plans and conservation actions, can also be used to understand the barriers to scaling up conservation actions, and how these can be overcome. For example, if a barrier to scaling up is the extent of the existing governance units, the development of social networks across governance units can encourage adjacent units to coordinate. I also develop an approach to predicting the benefits of scaling up management through time, which informs the design of incentives that steer communities towards complementary forms of management (Chapter 4). To address the challenges related to the lack of understanding of the optimal suite of conservation actions to achieve conservation goals, and lack of institutions supporting dynamic planning processes, I develop a method for incorporating multiple forms of management into gap analysis in countries where data are limited; which sets the scene for learning and adaptive management (Chapter 3).

1.4. Thesis structure and outline

This thesis consists of six chapters, arranged in four parts. First (Chapter 1, this introductory chapter), I provide the background to the body of my thesis. Second (Chapter 2), I examine the drivers and implications of the mismatch of scales between systematic planning and conservation action. Third (Chapters 3-5), I develop methods to inform spatial prioritisations with information on the contributions and conservation opportunity of different forms of management and simulate the conservation outcomes of plausible future scenarios for MPAs. In this part, I also develop two methods of understanding conservation opportunity. Fourth (Chapter 6), I provide a general discussion about how to design systematic planning processes that will lead to the implementation of effective conservation action and suggest directions for future studies in this research area (for details see Table 1.1).

All of my data chapters (Chapters 2-5) have been submitted to international scientific journals. Two of these have been accepted and two are in review.

Table 1.1. Thesis chapters, objectives and techniques used.

Chapter	Objective	Techniques
1	• To provide the context for this study and introduce the terminology used throughout the thesis • To introduce the reasons for the knowing-doing gap in systematic planning	• Literature review
2	• To review the systematic planning literature and identify the key decisions about spatial scale undertaken during systematic planning • To identify the considerations involved in each major decision about spatial scale, the trade-offs between considerations and the implications of undertaking systematic planning by deciding about scale in different ways • To create a framework in which the key decisions about spatial scale can be made explicit and investigated further	• Literature review • Review of case studies
3	• To develop an expert-based method to estimate the relative ecological effectiveness of different forms of management • To discuss the relative contribution of the different forms of management to conservation objectives and their relative importance in assessing progress towards conservation goals	• GIS • Ecosystem mapping • Focus groups • Gap analysis
4	• To model the spatial suitability of Fiji's inshore waters for different forms of management • To simulate the expansion of conservation actions to 2020 based on two different approaches to conservation • To understand the difference in the achievements of conservation goals between two different approaches to conservation	• Maximum entropy modelling • Scenario building and simulations • Use of systematic planning software • Interviews with key informants
5	• To develop a method to identify the human and social characteristics associated with the presence and form of management • To compare the performance of data at different resolutions to inform conservation opportunity	• Literature review • Household interviews • Canonical correlation analysis
6	• To summarize the main findings of this thesis • To discuss the methods developed and the approach of this thesis • To describe the remaining gaps in knowledge and identify key future research questions	• Literature review • Self-reflection

Chapter 2

A mismatch of scales: challenges in planning for implementation of marine protected areas in the Coral Triangle[1]

[1] Mills M., Weeks R., Pressey R.L., Foale S. and Ban N.C. 2010. A mismatch of scales: challenges in planning for implementation of marine protected areas in the Coral Triangle. Conservation Letters **3**, 291-303.

2.1. Abstract

Systematic planning is an effective approach to MPA network design, ensuring complementarity and functional connectivity of areas. However, systematic planning and conservation actions do not properly inform one another. One outcome is the failure of systematic plans to guide conservation actions. Another is that site-based MPAs constitute collections rather than functional systems for marine conservation. Understanding decisions related to spatial scale in systematic planning is essential for the development of ecologically functional networks of MPAs. Decisions about spatial scale require that planners address trade-offs between the respective advantages and limitations of different considerations in several parts of the systematic planning process. I provide the first comprehensive review of decisions about spatial scale that influence systematic planning outcomes. I illustrate these decisions and the trade-offs involved with planning exercises undertaken in the Coral Triangle. I provide a framework in which decisions about spatial scale can be made explicit and investigated further. The framework helps to link theory and application in systematic planning, facilitates learning, and promotes the application of conservation actions that are both regionally and locally significant.

2.2. Introduction

Declining fisheries and increasing deleterious human impacts on the marine environment have prompted international commitments for increasing protection of the ocean, this increase is recommended to be through systematic planning (Sala et al. 2002; Wood et al. 2008). Systematic planning involves many decisions about spatial scale. Here, I define spatial scale (hereafter 'scale') as extent and resolution of study regions, data, and areas of assessment. Five of eleven stages of the process of systematic planning (Pressey and Bottrill 2009) require decisions about scale. These decisions address ecological and social considerations and influence the final configuration of conservation actions. In marine environments, decisions about scale include: (1) the extent and delineation of the planning region; (2) the resolution of data to represent biophysical and human attributes of the region (Margules et al. 2002; Ban et al. 2009b); (3) the size and delineation of planning units for assessment and comparison (Pressey and Logan 1998); (4) MPA network design (Halpern and Warner 2003); and (5) the extent and delineation of management, including implementation of MPAs.

Regional-scale systematic planning initiatives identify areas where some form of local-scale conservation action should occur, assuming that ecologically functional networks of complementary actions will result (Franklin 1993; Poiani et al. 2000). Following Pressey and Bottrill (2009), we define 'regional' scale qualitatively as demarcating common "patterns and processes of biodiversity and human uses". This allows planners to consider the spatial context for conservation decisions, complementarity and connectivity between areas, threats to natural features, and relationships between different human activities. Most regional-scale systematic plans are prioritisation exercises only: they do not directly inform local-scale actions. However, the importance of linking systematic plans to local actions is increasingly recognized (Knight *et al.* 2006a,b). Understanding scale-related decisions in conservation planning is therefore essential if regional planning is to have local outcomes.

Globally, marine conservation action commonly results from local or site-based initiatives, focused only on the issues and values of one or a few communities. Within the South Pacific, a social network of over 500 communities across 15 countries and territories (the Locally Managed Marine Area network) has established MPAs that include small (usually <0.5 km^2) permanent or temporary closures (Govan et al. 2009). In the Philippines, there are more than 1000 community and local government MPAs (Weeks et al. 2010). Local initiatives can be rapidly implemented because they address local issues in a culturally sensitive manner (Govan et al. 2009). However, while, locally motivated MPAs have benefits for biodiversity and sustainable harvest, they have failed to coalesce into systems of complementary, functionally connected areas (Sala et al. 2002) and are unlikely to sustain regional-scale processes or associated biodiversity. Numerous site-based initiatives might, however, be the first significant step towards the creation of ecologically functional MPA networks (Lowry et al. 2009).

I define the mismatch of scales in systematic planning as the failure of systematic planning and management to inform one another. The advantages of systematic planning include its broad perspective on complementarity and functional connectivity. The advantages of locally driven management are their ownership and support by affected stakeholders. Additionally, local management better match the scale of government jurisdictions to which management is being devolved in several Asian and Pacific countries (Foale and Macintyre 2000; White 2008). The importance and urgency of bringing together these two scales of activity are nowhere better illustrated than in the CT (for description see Chapter 1, Section 1.2).

Challenges in bridging the gap between systematic planning and local management are acute in the CT because: (1) central governments have limited influence over marine management; (2) data for systematic planning are minimal and mainly at coarse resolutions; and (3) options for conservation are constrained by social, economic and political complexities, such as high dependence on marine resources and unresolved boundaries of customary tenure. Here I develop a framework to discuss decisions about scale in systematic planning and explore how these decisions ameliorate or exacerbate the mismatch of scale between systematic planning and management. I discuss the five kinds of decisions about scale (Figure

2.1) in the sections that follow, describing key considerations and the necessary trade-offs between them. I illustrate the decisions in marine conservation initiatives directed at the establishment of MPA networks within the CT. Given the few published systematic planning initiatives within the CT, I also use examples from Pacific Island nations. Opportunities and constraints on conservation actions are similar in both contexts, including dependence on fishing for protein, customary tenure and boundary issues, and central governments with limited resources for marine conservation. I finish by outlining the implications of decisions about scale for future systematic planning processes within the CT and elsewhere to more effectively translate systematic plans into management.

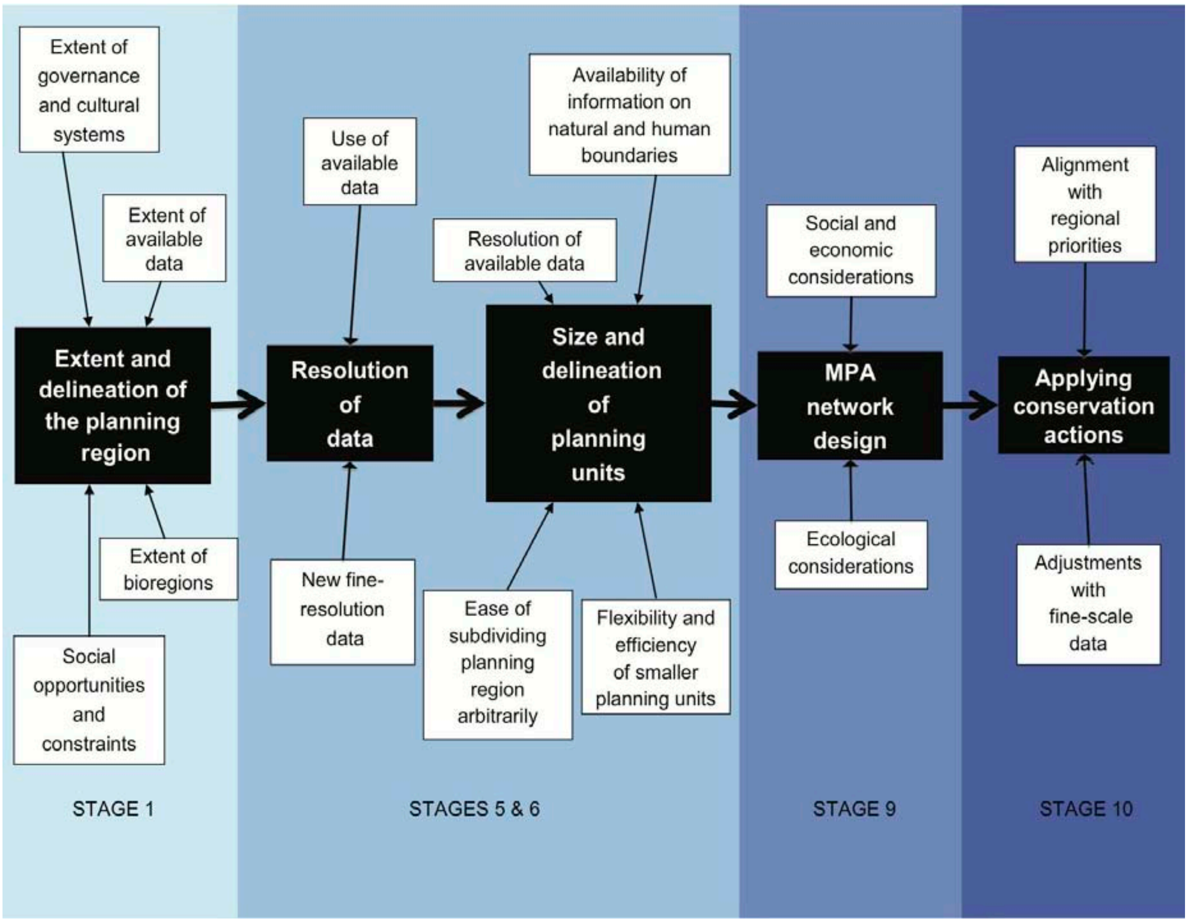

Figure 2.1. Decisions relating to spatial scale made during a systematic planning process (black boxes) and considerations that influence those decisions (white boxes). Planners will face trade-offs between the advantages and limitations of adjusting their decisions to competing considerations. Stages in the systematic planning process are adapted from those in the framework for systematic planning proposed by Pressey and Bottrill (2009). Of their 11 stages, decisions related to scale that I consider here are made within five stages: (1) scoping and costing; (5) compiling data on socio-economic variables; (6) compiling data on biodiversity and other natural features of interest; (9) selecting new conservation areas; and (10) applying conservation actions.

2.3. Extent and delineation of the planning region

Delineating the planning region (Figure 2.2b) is a prerequisite for planning (Pressey and Bottrill 2009). At least four considerations are important (Figure 2.1): (1) the extent of bioregions (or ecoregions) (Beck and Odaya 2001); (2) the extent of governance and cultural systems (Brunckhorst and Bridgeway 1995); (3) the extent of available data for planning (Pressey 2004); and (4) social opportunities and constraints for conservation action (Figure 2.1; Table 2.2). As I discuss below, when choosing among these considerations, planners have no single, correct prescription for identifying planning region boundaries. Instead, they will face trade-offs between the respective advantages of large and small extents and alignment of regional boundaries with other kinds of information.

2.3.1. Extent of bioregions

Bioregions have relatively homogeneous biological and physical composition, distinct from adjacent regions, and are large enough to encompass ecological and evolutionary processes (Spalding et al. 2007). Working within bioregions (Figure 2.3a) enables planners to compare areas with similar physical and biological compositions while also considering extensive processes that promote species' persistence and ecosystem functions (Margules and Pressey 2000; Olson and Dinerstein 2000). Marine bioregions, in the order of 10^4 - 10^5 km^2, have been delineated within the CT by international NGOs (Green and Mous 2008). Although extensive planning regions aligned with bioregions are advantageous ecologically, their large size can conflict with the considerations below that make smaller regions more manageable. In practice, planning regions delineated in the CT encompass only parts of bioregions (Appendix Table 8.1). These studies considered that smaller, biologically distinctive regions were more suitable for the design and management of MPA networks because they combined relatively uniform natural attributes, similar human activities, and aspects of governance that facilitated management (Green et al. 2004; Green and Mous 2008).

2.3.2. Extent of governance and cultural systems

Uniform governance and cultural systems are generally much less extensive than bioregions in the CT (Figure 2.3a,b). In much of the CT, except for East Timor and Malaysia, management has been decentralised to local governments (Alcala and Russ 2006; Siry 2006) or is held by kinship groups (Carrier 1987; Johannes 2002), delegating decision-making to those most reliant on natural resources for their livelihoods. For example, customary tenure regimes of the Solomon Islands and Papua New Guinea govern resources along hundreds of meters to a few kilometres of coastline (Foale and Macintyre 2000). This is likely a result of central governments within these countries acknowledging their limited reach in relation to management as compared to more developed countries (Figure 2.4). Planning regions aligned with bioregions will therefore encompass multiple governance and cultural systems in much of the CT, with application of conservation actions challenged by negotiating agreeable outcomes for multiple parties. Governance and cultural heterogeneity in the CT therefore encourages smaller planning regions with corresponding advantages for effective conservation action. Boundaries based on customary tenure or provincial or district governance have therefore been used in Pere in Papua New Guinea and Karimunjawa and Berau in Indonesia (Appendix Table 8.1, rows E,I,J). The boundaries for Kimbe Bay (Appendix Table 8.1, row B) were based primarily on biophysical data, then modified to account for provincial and village boundaries.

Planning regions, rather than being predefined, can also emerge from local initiatives, determined by social or cultural connections such as language groups and religion. Two regional conservation initiatives (Appendix Table 8.1, rows A,D) started as locally but then expanded. In Cebu (Philippines), the planning region was based on biophysical information and a social network among communities implementing MPAs for livelihood or conservation purposes (Appendix Table 8.1, row A). Social networks encouraging conservation action are becoming more prominent globally (e.g. Govan et al. 2009) and are considered critical to successfully scaling up conservation initiatives by promoting communication, learning, identification of common problems, and coordination (Pretty and Smith 2004; Lowry et al. 2009). The initiative within the Roviana and Vonovana Lagoons (Solomon Islands) incorporated areas associated with the Christian Fellowship Church (Aswani 1999; Aswani and Lauer 2006) and has proven effective in encouraging resource protection. The wider potential of religious groups, language groups and other cultural groups to define planning regions has yet to be investigated. In the CT, planning regions that emerged by consolidation of local initiatives are much smaller than bioregions (Appendix Table 8.1, rows A, D).

2.3.3. Extent of available data

Ideally, planning regions would have consistent biophysical, economic and social data. Otherwise, conservation attention will be biased towards areas with more data (Margules et al. 2002; Pressey 2004). In reality, very few regions have consistent data at a suitable resolution for systematic planning. Most data within the CT are limited in extent, highly fragmented, and spatially biased towards NGO offices and research stations (Johannes 1998; Christie et al. 2009b). Therefore planners face a trade-off: limit planning regions to the extent of available fine-resolution datasets (usually very small areas) for unbiased selection of areas; or combine all available data (coarse- and fine-resolution) for regions delimited using bioregional, governance or cultural boundaries, resulting in biased or less efficient planning processes (Margules et al. 2002). Within the CT, planners chose inconsistent data across regions aligned with biological and/or governance boundaries (Appendix Table 8.1).

2.3.4. Social opportunities and constraints

Institutional capacity and support for conservation actions can indicate strongly whether conservation actions will be feasible (Cowling and Wilhelm-Rechmann 2007). For example, regional conservation initiatives of The Nature Conservancy (TNC) in the CT are in areas where TNC or Mahonia Na Dari (a local NGO) previously established relationships with communities (Green and Mous 2008). Conversely, political unrest, poor governance or corruption might lead planners to avoid areas because of risk to staff or poor prospects for effective conservation action. Considerations of opportunities and constraints are likely to constrain the extent of planning regions and shape their boundaries, while also facilitating the application of conservation actions.

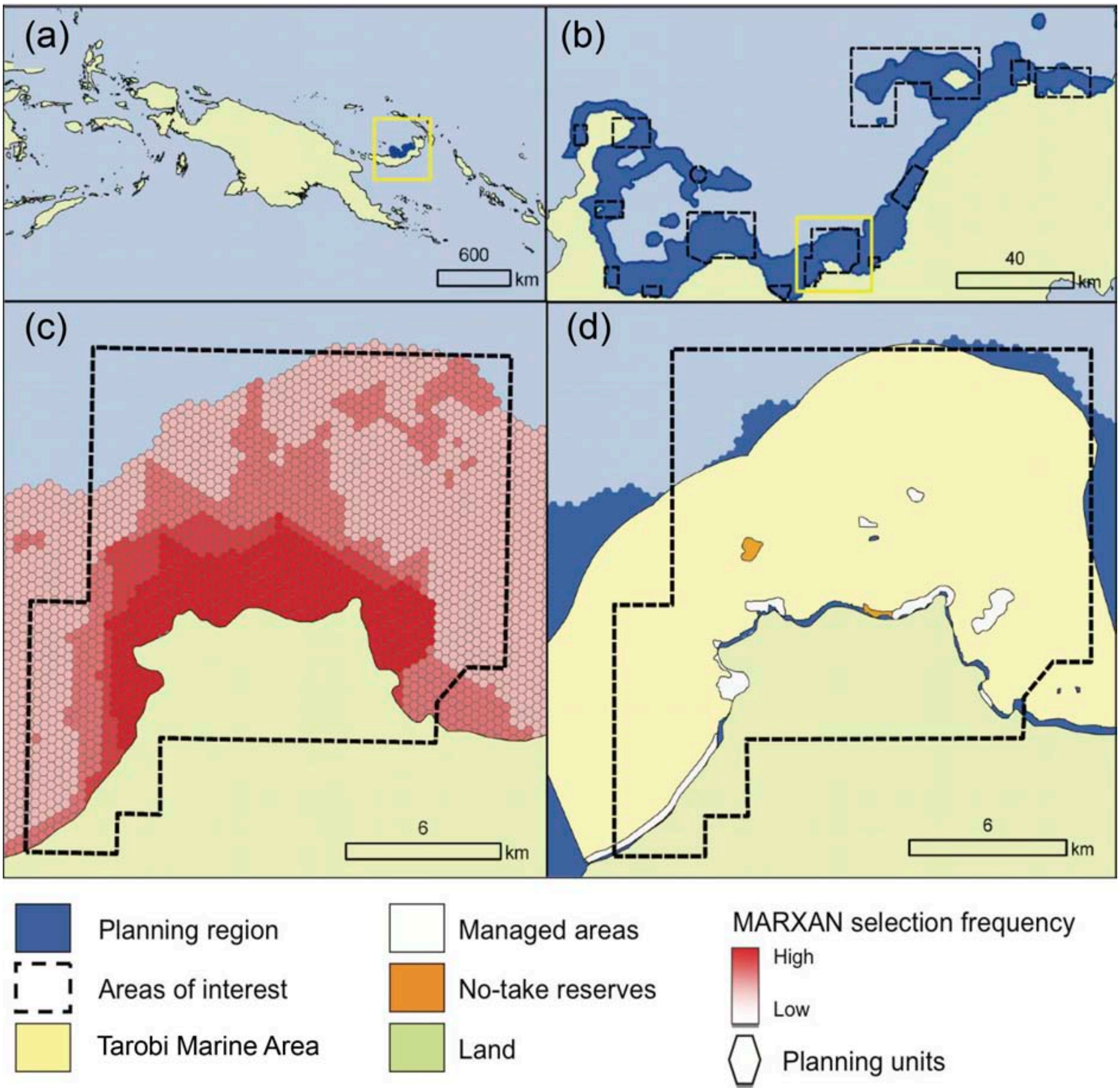

Figure 2.2 Differences in spatial scale considered during the systematic plan undertaken for Kimbe Bay [see Green et al. (2009) for further information]. (a) The island of New Guinea and surrounding islands of Indonesia, Papua New Guinea and the Solomon Islands, with yellow box highlighting the location of the Kimbe Bay planning region. (b) The Kimbe Bay planning region resulted from trade-offs involved when deciding the extent and delineation of the planning region (Figure 2.1). The dashed lines represent areas of interest (AOI) or the areas within the planning region identified as conservation priorities where further work was focused. Tarobi AOI is highlighted by the yellow box. (c) The Tarobi AOI showing selection frequency of planning units analysed by MARXAN. The pattern of selection frequency resulted partly from trade-offs regarding the size and delineation of planning units (Figure

2.1). Areas with darker shades of red had higher selection frequencies and were assessed to have higher value for the achievement of representation targets. (d) The Tarobi Marine Area, covering 202 km^2, is a general use zone recognized by local communities. Managed areas within the Tarobi Marine Area total 8.9 km^2 and include habitat protection zones, sacred sites and preservation and conservation areas. The permanent and temporary closures are a subset of the managed areas and cover 0.7 km^2. These managed areas are the results of trade-offs involved in applying conservation actions (Figure 2.1).

Table 2.2 Decisions about scale, considerations that shape decisions, and trade-offs.

Decisions about scale	Considerations	Tradeoffs
Extent and delineation of the planning region	1. Extent of bioregions 2. Extent of governance and cultural systems 3. Extent of available data 4. Social opportunities and constraints	Competing considerations: 1 & 2 & 3 & 4 - Delineating the planning region based on bioregion boundaries to maximise the ecological effectiveness of management. - Using governance and cultural boundaries to facilitate coordination and management. - Limiting the planning region to areas where consistent fine-resolution data are available to promote unbiased area selection. - Adjusting the planning region to benefit from social opportunities and avoid constraints to promote effective conservation action.
Resolution of data	1. Use of available data 2. New fine-resolution data	Competing considerations: 1 & 2 - Working with the resolution of available data to focus time and funds on immediate conservation action. - Collecting new fine-resolution data to better reflect the variability of the natural and human attributes of the planning region and so improve planning decisions.
Size and delineation of planning units	1. Flexibility and efficiency of smaller planning units 2. Resolution of available data 3. Availability of information on natural and human boundaries 4. Ease of subdividing the planning region arbitrarily	Competing considerations: 1 & 2 - Selecting small planning units to maximise flexibility in configuring potential MPAs and minimise the total cost of achieving objectives. - Adjusting planning unit size to the generally coarse resolution of available consistent data (unless rigorous procedures can be applied to downscale data and understand resultant errors). Competing considerations: 3 & 4 - Delineating planning units based on information on natural and human boundaries to facilitate the transition from spatial prioritisations to management. - Using arbitrary boundaries to facilitate the subdivision of the planning region for regional design and avoid boundary-related ownership issues (although the conversion of arbitrary units to units of conservation action might entail additional later costs).
MPA network design	1. Ecological considerations 2. Social and economic considerations	Competing considerations: 1 & 2 - Designing extensive MPAs to maximise their effectiveness in achieving ecological objectives. - Addressing social and economic constraints with small MPAs that facilitate the application of conservation actions.
Applying conservation actions	1. Alignment with regional priorities 2. Adjustment with fine-scale data	Competing considerations: 1 & 2 - Attempting to apply conservation actions that are aligned with regional priorities to implement a theoretically functional and resilient MPA network. - Departing judiciously from regional priorities and adjusting MPA design as local insight reveal new information, thereby facilitating the application of conservation actions.

2.4. Resolution of data

Data extent is related to both consistency and resolution. I focus here mainly on data resolution (see above for consistency). The examples provided predominantly concern biophysical data because these have been the focus of much of the systematic planning research to date. Issues regarding resolution of social and economic data, however, are likely to be similar. As surrogates for biodiversity (e.g. marine habitats) used in systematic planning are defined more finely, their representation within existing protected areas can change, the total area required to represent them increases, and the relative conservation value of planning units alters (Pressey and Logan 1995; Rouget 2003). Consequently, choices about data resolution inevitably influence which and how many areas are identified for conservation actions. Two considerations (Figure 2.1) will influence the resolution of data used for MPA design: (1) use of available data; and (2) collection of new fine-resolution data (Figure 2.1; Table 2.2). Planners must trade-off the respective advantages of working immediately with available data, despite their limitations, and spending time and money to collect new information (Table 2.2).

2.4.1. Use of available data

Using available data allows the systematic planning process to proceed without delay, minimising the progressive attrition of natural features. Additionally, resources that would be used for further data collection can be used for applying conservation actions (Grantham et al. 2009). However, available data are likely to have limitations. The extent and resolution of data are positively correlated (Donald and Fuller 1998) so large planning regions are likely to have consistent data available only at coarse resolution while coverage of fine-resolution data will generally be patchy. Methods are available to combine data at different resolutions while maximizing accuracy and currency (Keith and Simpson 2008), but the resulting composite data will inevitably result in biased selection of conservation areas. Based on statements about data in the CT, all spatially explicit data were used, independent of their resolution and generally in combination with new data (Appendix Table 8.1).

2.4.2. New fine-resolution data

Although time-consuming and expensive, collecting additional fine-resolution data has two related advantages. First, fine-resolution data better represent spatial variability in biodiversity (and probably in social and economic attributes), which is obscured by coarse-resolution data (Rouget 2003; Banks and Skilleter 2007). Second, collection of additional fine-resolution data throughout the planning region will allow for more consistency in the design of potential MPAs. In the Kimbe Bay planning process, TNC undertook new data collection for six of the eight datasets used. Aswani and Lauer (2006) collected indigenous knowledge on habitats, spawning and nursery sites and fishing grounds when planning MPAs in Roviana and Vonovana Lagoons (Appendix Table 8.1, row B,D).

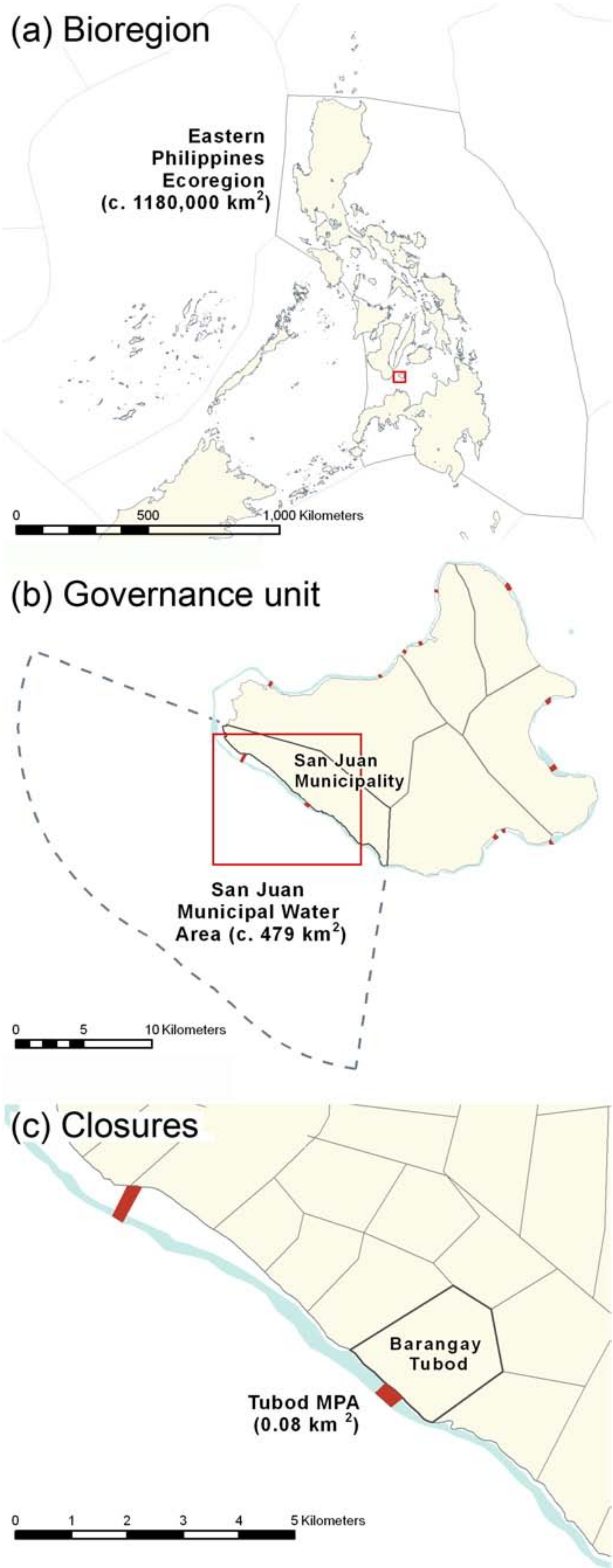

Figure 2.3. Contrast in extent of bioregions (or ecoregions), governance units, and marine protected areas (MPAs) in the Philippines. (A) The Eastern Philippines Marine Ecoregion (Green and Mous 2008). Bioregional classifications such as this are often recommended as appropriate planning regions. The small red rectangle is the approximate extent of part B. (B) A municipality in the Philippines. Municipalities are the most important governance units for coastal management in the country. MPAs are generally established cooperatively by municipal and barangay (local) governments. Small red dots indicate the location of no-take MPAs on the island of Siquijor. Red rectangle indicates the approximate extent of part C. (C) A barangay and its permanent and temporary closure (Tubod MPA) within the San Juan Municipality. Permanent and temporary closures of this size are commonly established in the Philippines (Weeks et al. 2009).

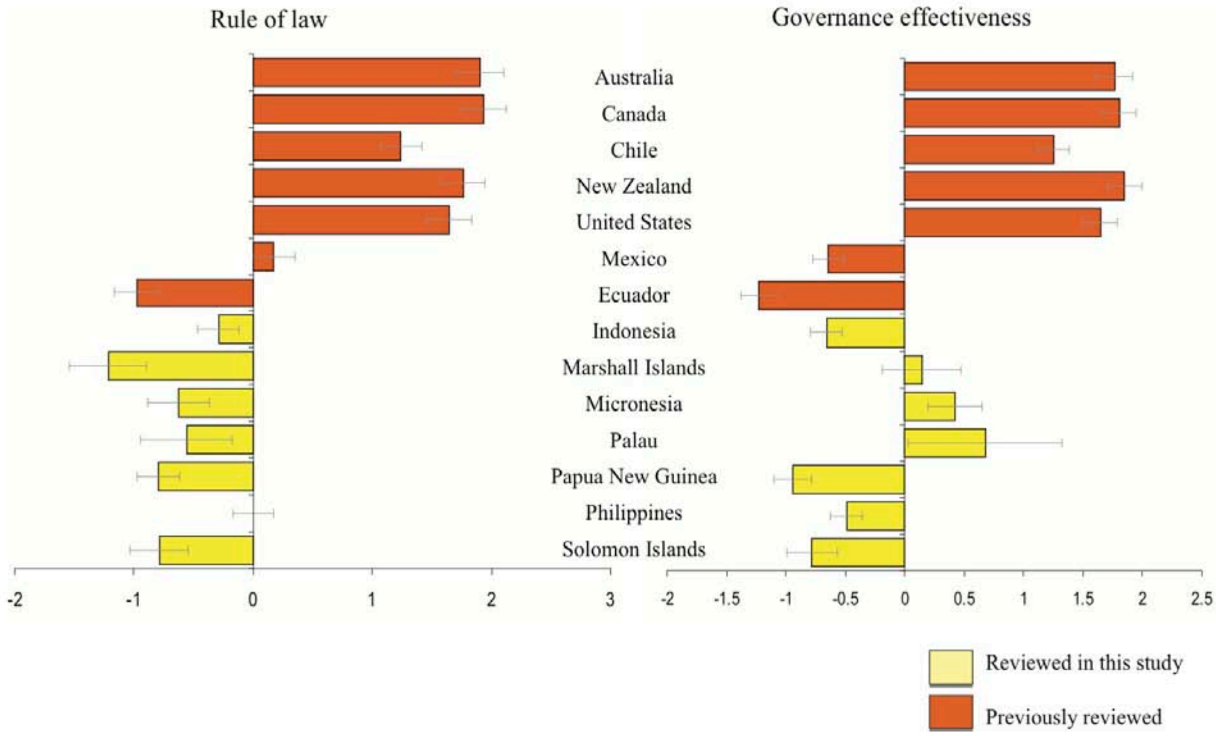

Figure 2.4. Governance effectiveness and rule of law indicators in countries with systematic planning exercises for MPA networks. Countries shown are those covered by Leslie (2005), grouped as ‘previously reviewed’, and by this study, grouped as ‘reviewed in this study’. Systematic plans have been undertaken more extensively and over a longer period in developed countries with strong governance and rule of law. Systematic plans are now being developed for Pacific and CT countries with weaker governance and rule of law, limiting the applicability of planning models from developed countries. Governance effectiveness and rule of law indicators are from 2008 governance indicators in Kaufmann et al. (2009). Governance effectiveness includes the “perceptions of the quality of policy formulation and implementation, and the credibility of the government's commitment to such policies” (Kaufmann et al. 2009), indicating the likelihood of the application of conservation actions by central governments. Rule of law includes “perceptions of the extent to which agents have confidence in and abide by the rules of society, and in particular the quality of contract enforcement, property rights, the police, and the courts” (Kaufmann et al. 2009), indicating the likelihood of enforcement of management regulations by central governments. Both the governance effectiveness and rule of law scores lie between -2.5 and 2.5 and higher scores indicate better governance effectiveness and rule of law.

2.5. Size and delineation of planning units

Planning units are the spatial units of assessment and comparison used in most systematic planning exercises that employ decision-support software, and are the building blocks of regional MPA designs (Figure 2.2c). Four considerations (Figure 2.1) influence choices about the size and delineation of planning units: (1) the greater spatial flexibility and efficiency of using smaller planning units; (2) matching the resolution of the available data; (3) availability of information on natural and human boundaries suitable for defining planning units; and (4) ease of subdividing the planning region arbitrarily (Figure 2.1; Table 2.2). When choosing among these considerations, planners will make two trade-offs. The first, concerning planning unit size, is between the relative advantages of working with smaller planning units or respecting the (usually coarse) resolution of available data that often requires larger planning units. The second, concerning delineation of planning units, is between the relative advantages of using human and natural boundaries or deriving them arbitrarily (Table 2.2).

2.5.1. Flexibility and efficiency of smaller planning units

The total area required to reach representation targets for features (e.g. habitat types) depends on planning unit size: small planning units are more efficient than large planning units, requiring less total area to achieve targets (Pressey and Logan 1998). Smaller planning units are also likely to achieve targets with smaller overall costs measured in other ways, for example as opportunity costs to communities. Smaller planning units can also be clustered more flexibly to create more appropriate protected area configurations (Rouget 2003). For the two studies that reported on planning units, sizes varied from 10 to 15 ha to allow for spatial precision in selecting areas of interest (Appendix Table 8.1, rows, B F).

2.5.2. Resolution of available data

Without rigorous procedures for downscaling, working with planning units smaller than the actual data resolution has no benefit for interpreting those data and overstates their quality. For example, resource maps at 1:100,000 have recommended minimum mapping units (smallest recognisable polygons) of 40 ha (Hupy et al. 2004), even though finer-resolution variability is expected. Using planning units (or pixels in geographic information systems) smaller than this size confers no advantage for interpreting data, unless the data can be downscaled and the resulting uncertainties considered explicitly (e.g. Gardner et al. 2008). In the CT, satellite imagery (15-60 m spatial resolution, or 1:20 000 to 1:24 000) has been widely used to construct habitat maps (Appendix Table 8.1, rows B,D,F,G,J). The minimum mapping unit from these data varies from 1.6 to 2.3 ha (Hupy et al. 2004). These and other data (for which information on resolution is unavailable) were recorded for planning units of 10 to 15 ha (Appendix Table 8.1, rows B,F), thereby respecting the spatial limitations of the data. In some cases, there might be advantages in using planning units smaller than the minimum data resolution, for example to identify specific fine-resolution areas of interest to communities and managers. A disadvantage of this approach will be an increased need to adjust systematic plans as conservation actions are applied because data are more likely to be found incorrect at local scales (below).

2.5.3. Availability of information on natural and human boundaries

The delineation of planning units can be arbitrary (e.g. hexagons, Figure 2.2c), based on ownership or political (e.g. district) boundaries, based on natural boundaries (e.g. coral reef edges), or derived from some combination of these (e.g. Lewis et al. 2003). Explicit consideration of human and natural boundaries can aid the transition from MPA network design to applying conservation actions. For example, it is easier to implement conservation actions within a planning unit that falls within a single governance unit than one that falls across a boundary. However, data on fine-scale natural or human subdivisions are generally unavailable across whole planning regions in the CT. Boundaries of customary tenure, for example, are often unresolved, and attempts to delineate them can lead to conflict, a detrimental outcome for management (Akimichi 1995; Foale and Macintyre 2000).

2.5.4. Ease of subdividing the planning region arbitrarily

Delineating planning units arbitrarily is a simple way of subdividing the planning region, avoiding problems surrounding social boundaries. Additionally, the use of arbitrary boundaries avoids funds and time being spent on the collection of boundary-related data, although there might be consequent later costs in reinterpreting arbitrary boundaries for conservation actions. In all cases in the CT, arbitrary planning units derived from square or hexagonal grids were the preferred option for the NGO planners involved (Figure 2.2c; Appendix Table 8.1).

2.6. MPA network design

During the design stage, planners make decisions about the size and configuration of either generic conservation areas or areas designated for different conservation actions, influenced by ecological, social and economic considerations. I focus here on considerations for permanents because these involve the most restrictive management. Similar considerations will have to be addressed for other forms of management such as gear or catch restrictions. For permanents, planners will be faced with trade-offs between the respective advantages of large and small areas (Figure 2.1; Table 2.2).

2.6.1. Ecological considerations

A large literature discusses the ecological and fisheries considerations involved in decisions about the size and configuration of permanents, e.g., the “single large or several small” (SLOSS) debate (Kingsland 2002; Halpern and Warner 2003). Ecological considerations, such as protecting extensive processes and species with large home ranges, have been reviewed comprehensively elsewhere (Roberts et al. 2001; Palumbi 2004) and arguments prevail for larger permanents (e.g. at least 10 or 20 km across). However, permanents of this extent are not necessarily feasible in some CT countries because of social and economic constraints.

2.6.2. Social and economic considerations

The social and economic context of the CT favours smaller permanents (Figure 2.2d; Figure 2.3c) (Aswani and Hamilton 2004a; Cinner 2007). Most coastal communities in the CT are highly dependent on marine resources for subsistence and commerce (e.g. Burke et al. 2002). Most artisanal and subsistence fishermen operate small vessels close to home, for example on inshore reefs (Aswani and Lauer 2006), and women contribute to household subsistence by gleaning in intertidal areas (Vunisea 2008). While near-shore permanents close to villages are preferred because of easier enforcement (Aswani and Hamilton 2004a; Aswani and Lauer 2006; McClanahan et al. 2006), these are also more costly to villagers who generally have limited ability to switch fishing grounds or livelihoods. Large permanents in coastal areas therefore disproportionately constrain the livelihoods of some communities (Foale and Manele 2004) while numerous small permanents distribute their costs and benefits more equitably between communities and avoid conflict (Aswani and Hamilton 2004a).

2.7. Applying conservation actions

The transition from MPA network design to applying conservation actions on the ground or in the water will be influenced by two main sets of considerations (Figure 2.1; Table 2.2). First, regional priorities (areas identified for generic or specific conservation actions during regional-scale design) can inform where conservation actions are applied, including the implementation of MPAs, education and awareness campaigns, alternative livelihoods, and incentives for conservation (Salafsky et al. 2008; Kapos et al. 2009). Second, new fine-resolution data that become available or useable for the first time when conservation actions are applied will cause planners to depart from regional-scale prescriptions to some extent (Table 2.2). There are no consistent guidelines for deciding on the extent to which conservation actions should align with regional priorities or be adjusted to accommodate new fine-resolution data. Planners face trade-offs between the respective advantages and limitations of each choice. A dynamic interaction between systematic plans and management is likely needed to reconcile these two sets of considerations, and to implement an ecologically functional MPA network (Pressey and Logan 1998).

2.7.1. Alignment with regional priorities

MPA networks designed at a regional-scale can incorporate information on complementarity and extensive ecological processes such as larval dispersal and river plumes. Strategic decisions about MPA size and placing are intended to promote the persistence of species within individual MPAs and the network as a whole, resulting in a theoretically functional and resilient system (Botsford et al. 2001; Sala et al. 2002). However, the data on which MPA design is based are always limited and to some extent incorrect. Locating conservation actions only in regional-scale priority areas can therefore ignore unforeseen constraints in these areas and opportunities elsewhere, lead to systematic plans being difficult to apply, and fail to address the limitations of regional-scale data on biodiversity. Difficulties in implementing regional scale plans have been found in Wakatobi and Karimunjawa, Indonesia (Appendix Table 8.1, rows H,I). The advantages of aligning conservation actions with the regional design must be weighed against those of altering this design as new data become available, usually at a local scale.

2.7.2. Adjustment with fine-scale data

Fine-scale variability in biodiversity, costs, threats and opportunities within the planning region will shape where, when and how conservation actions are eventually applied. Fine-scale social complexity, such as customary tenure boundaries, resource use patterns, and community support for conservation, influence the potential to implement MPAs (Figure 2.2d; Figure 2.3c) (Aswani and Hamilton 2004b; Christie et al. 2009a). However, at a regional-scale, much of this information is difficult or impossible to collect. For example, many customary tenure boundaries are not mapped (Johannes 2002). Adjusting MPAs to local information, and thereby departing to some extent from regional design, will increase local support and compliance with resource regulations. In Kimbe Bay, fine-resolution customary tenure information was used after regional design of areas of interest to negotiate managed areas with communities (Appendix Table 8.1, row B; Figure 2.2d).

2.8. Discussion

I provide the first comprehensive review of decisions about spatial scale that influence systematic planning outcomes, including the effectiveness of conservation actions (Kapos et al. 2009). This study provides a framework in which these decisions can be made explicit and investigated further, and illustrates trade-offs particularly for the CT. Explicitness about the considerations that shape decisions about scale in different contexts, and how these considerations are weighed against one another, will help to link theory and application in systematic planning and facilitate learning so that planners can make better decisions in the future (Knight et al. 2006).

Decisions about scale require planners to deal with trade-offs between the respective advantages and limitations of different decisions (Table 2.2). When choosing among considerations that influence decisions about scale, planners have no prescribed, correct decision. Instead, they need to anticipate what balance between considerations will be most effective in their particular ecological, social and economic context. The trade-offs identified in this review are relevant globally, and to terrestrial and freshwater as well as marine realms. They will, however, be resolved differently in CT countries than in countries where systematic planning has been developed and extensively applied (e.g. Australia or the United States; Leslie 2005). For example, the central governance of most countries in which MPA planning exercises were reviewed by Leslie (2005) was relatively strong, with better implementation and enforcement of policies than in the CT countries (Figure 2.4) and less dependence on resources for subsistence.

The social, political and economic context of the CT countries drives many decisions about scale to respond to local perspectives. This is often critical for the application of conservation actions to be feasible. However, while these perspectives shape trade-offs in the CT differently than in developed countries, an essential role remains for systematic planning in this region. Community-driven conservation actions can achieve local management objectives, but will not achieve regional conservation objectives without a broader planning outlook (Weeks et al. 2010). Systematic planning provides the essential context for management decisions, allowing planners to coordinate individual conservation actions for complementarity, connectivity, and the avoidance of threatening processes (Pressey and Bottrill 2009). With regional perspectives, individual community-based and government-initiated conservation actions can add up to more than the sum of individual parts: they can contribute to ecologically functional systems (Groves et al. 2002). This will be critical if management is to be resilient to emerging threats in the CT such as climate change. Nonetheless, for the potential of regional-scale perspectives to be realised in the CT, heightened awareness of extensive ecological processes and education about the foundations of effective marine conservation are needed.

To bridge the gap between systematic planning and conservation action, a dynamic interaction between them will be needed so that both perspectives progressively inform one another. For information to flow between systematic plans and conservation actions and for coordination to occur among different conservation actions, marine conservation needs: (1) Strengthening of institutional capacity, and (2) consistent engagement between community groups, NGOs and all levels of government. Additionally, those involved in marine conservation must make long-term commitments to their regions (Christie et al. 2009a). These issues are beginning to be addressed by NGOs in the CT where NGOs intend to work collaboratively for long periods and strengthening management legislation and capacity building have prepared the ground for the development of effective systematic plans (e.g. in the Solomon Islands by TNC and the Locally Managed Marine Area Network). Some of these projects are not only tackling threats to resources such as overfishing and pollution, but also the root causes of such threats, including population growth (White et al. 2005).

Chapter 3

Incorporating effectiveness of community-based management in a national marine gap analysis for Fiji[2]

[2] Mills M., Jupiter S., Pressey R.L., Ban N.C. and Comley J. Assessing effectiveness of community-based management strategies towards achieving national biodiversity targets: A new approach for marine conservation in Fiji. Conservation Biology. In Press.

3.1. Abstract

Every form of management in a systematic plan has a different level of effect and consequently contributes differentially to conservation. I examined how several community-based, marine, forms of management differed in their contribution to national-level conservation goals in Fiji. I held a workshop with experts on local fauna, flora and local marine management to translate conservation goals developed by the national government into ecosystem-specific quantitative objectives and to estimate the relative effectiveness of Fiji's community-based management in achieving these objectives. The national conservation objectives were to effectively manage 30% of the nation's fringing reefs, nonfringing reefs, mangroves, and intertidal ecosystems (30% objective) and 10% of other benthic ecosystems (10% objective). The experts evaluated the contribution of the various forms of management toward national objectives. Scores ranged from 0 (ineffective) to 1 (maximum effectiveness) and included the following forms of management: permanent closures (i.e., extractive use of resources prohibited indefinitely) (score of 1); temporary closures harvested once per year or less as dictated by a management plan (0.50-0.95); temporary closures harvested without predetermined frequency or duration (0.10-0.85); other forms of management, such as regulations on gear and species harvested, (0.15-0.50). Through 3 gap analyses, I assessed whether the conservation objectives in Fiji had been achieved. Each analysis was based on a different assumption: (1) all parts of locally managed marine areas (including closures and other management) conserve species and ecosystems effectively; (2) closures conserve species and ecosystems, whereas areas outside closures, open to varying levels of resource extraction, do not; and (3) management that allow different levels of resource extraction vary in their ability to conserve species and ecosystems. Under assumption 1, Fiji's national conservation objectives were exceeded in all marine ecosystems; under assumption 2, none of Fiji's conservation objectives were met; and under assumption 3, on the basis of the scores assigned by experts, Fiji achieved the 10% but not the 30% objectives for ecosystems. Understanding the relative contribution of various forms of management to achieving conservation objectives is critical in the assessment of conservation achievements at the national level, where multiple forms of management will be needed to achieve conservation objectives.

3.2. Introduction

Unsustainable levels of fishing have contributed to rapid declines of global marine biological diversity, including the ecosystem functions that benefit humans (e.g., decrease in productivity of fisheries) (Sala and Knowlton 2006). Consequently, many countries have committed to reducing declines in marine biological diversity, for example, by signing the Convention on Biological Diversity [CBD]. Signatories to the CBD commit to establishing networks of “representative and effectively managed” protected areas in marine environments (CBD 2008) aimed at the conservation of all levels of biological diversity. Protected areas are defined as areas “designated or regulated and managed to achieve specific conservation objectives” (CBD 2008). Approaches used to maintain and increase biological diversity generally rely on measures of the representation of selected species and ecosystems within protected areas as surrogates for data on overall genetic and species diversity (Margules and Pressey 2000). I assessed Fiji’s progress toward meeting its marine conservation goals, which reflect its commitments to the CBD. I conducted gap analyses (e.g., Scott et al. 1993) under different assumptions about the relative effectiveness of community-based management.

Many countries, including Fiji, set a broad national goal of effectively managing 30% of inshore marine ecosystems (Jupiter et al. 2010; Rondinini and Chiozza 2010). For gap analyses, such goals must be translated into quantitative conservation objectives, at least for measures of biological diversity for which spatial data on distributions are available. Objectives defined as fixed percentages of each ecosystem imply that society believes all ecosystems warrant equal levels of conservation. To reflect the unequal distributions of species across ecosystem types, explicit criteria can be developed to formulate objectives that vary among ecosystems (Desmet and Cowling 2004). Although limited biological data often make development or measurement of such criteria difficult (Rondinini and Chiozza 2010), criteria that are based on the rarity of and threats to an ecosystem can ensure that ecosystems subject to high levels of human use are managed extensively (Pressey and Taffs 2001).

Forms of management intended to protect biological diversity are not equally effective (Shahabuddin and Rao 2010). Gap analyses can be used not only to show representation of species and ecosystems within protected areas (Scott et al. 1993), but also, by inference, to assess the relative effectiveness of different forms of management in achieving representation objectives. I use the term *effectiveness* to describe the level of effect a forms of management has on biological, social, and economic conditions, including persistence of biological diversity (Hockings et al*.* 2006). Effective management relies partly on human behaviour and partly on ecology (e.g., species' life histories and behavioural responses to management). I focused on the ecological aspects of management effectiveness (hereafter ecological effectiveness), which I define as the relative contribution of various forms of management to realizing conservation objectives.

Ecological effectiveness of different forms of management is likely to vary widely across species and ecosystems. Marine management include permanent and temporary closures to fishing, size limits on fish harvested, seasonal bans on fishing during breeding seasons, bans on taking certain species, and restrictions on fishing gear. There are few empirical studies on the ecological effectiveness of such forms of management. Results of some studies show that temporary closures can be as effective as permanent closures in increasing the abundance and biomass of target species (e.g., Bartlett et al*.* 2009). Results of other studies show that no-entry areas protect some species more effectively than permanent closures, where entry is allowed but resource extraction is not, and that permanent closures are more effective than partial-take areas (McCook et al*.* 2010). Global analyses indicate variability in the effectiveness of permanent closures in increasing species richness and the biomass, density, and size of organisms within their boundaries, perhaps because of differences, both within and outside closures, in the degree of previous resource use (Russ and Alcala 1999; Lester et al*.* 2009).

Conservation assessments are based on different assumptions about ecological effectiveness and researchers generally assume a positive correlation between effectiveness and extent of protection. The simplest and most common approach to gap analysis is to assume effectiveness is binary: areas are either protected or not. Previous marine gap analyses focused on the extent of permanent closures and managed areas in aggregate (e.g., Mora et al. 2006; Wood et al. 2008; Weeks et al. 2010). To the best of my knowledge, marine gap analyses have not previously included the relative contribution of different forms of management to conservation objectives.

I based my examination of Fiji's progress toward its national conservation goal on the extent to which different forms of management resulted in inclusion of species groups and ecosystems in protected areas and the ecological effectiveness of those forms of management. I used Fiji as a case study because I had the opportunity to collaborate with the Fiji Protected Area Committee, which is charged with expanding the national network of MPAs. Fiji is a good case study because the national government has committed to protecting 30% of its inshore and offshore waters within MPAs by 2020 (Jupiter et al. 2010) and it is the country with the greatest spatial coverage of community-based management in the Pacific. These forms of management were established by communities, primarily to maintain livelihoods (Govan et al. 2009). I believe my results will help in the understanding of how community-driven conservation efforts can contribute to national conservation objectives.

3.3. Methods

3.3.1. Study region

Fiji's nearshore waters are divided into 410 traditional fishing grounds the boundaries of which are legally demarcated by the Native Lands and Fisheries Commission from the low water mark to outer barrier reefs (Figure 3.1). Traditional fishing grounds are areas where fishing rights for indigenous Fijians are legally recognized by the Fiji Fisheries Act, but the state owns the seabed and overlying waters. Over 10,000 km^2 of Fijian waters are included within locally managed marine areas (LMMAs). The number of LMMAs in Fiji grew from 1 in 1997 to over 100 by 2009. This growth came from community requests for assistance to the Fiji LMMA network to stem a perceived decline in fish (Govan et al. 2009). Fiji LMMA network is a group of resource managers who focus on lessons learned about the benefits and shortcomings of marine management in Fiji. An LMMA in Fiji is defined as an area of inshore waters governed by local residents and involving a collective understanding of, and commitment to, management interventions in response to threats to marine resources (Figure 3.2). Equivalent to MPAs, LMMAs can be subject to multiple, simultaneous forms of management. Within the boundaries of an LMMA, community members may choose to establish permanent closures or closures in which temporary harvest is allowed. The application of temporary harvest is based on long-standing Pacific traditions of management (e.g., Clarke and Jupiter 2010). Permanent closures prohibit all extractive use of resources indefinitely. I call temporary closures that allow harvests once per year or less as dictated by a management plan or collective decision at the community level temporary closures with controlled harvesting (or 'controlled closures'). Many of the temporary closures in Fiji are harvested without any predefined frequency and duration. I refer to these as temporary closures with uncontrolled harvesting (or 'uncontrolled closures'). Other management is the suite of forms of management, including bans on fishing gear, take of certain species, and seasonal prohibitions, that operate in LMMAs but outside closures. After one or more closures are implemented within a fishing ground, other management is applied across the remainder of that fishing ground (Jupiter et al. 2010). Management not associated with the LMMA network exist (e.g., licensing controls), but spatial data on their implementation are unavailable.

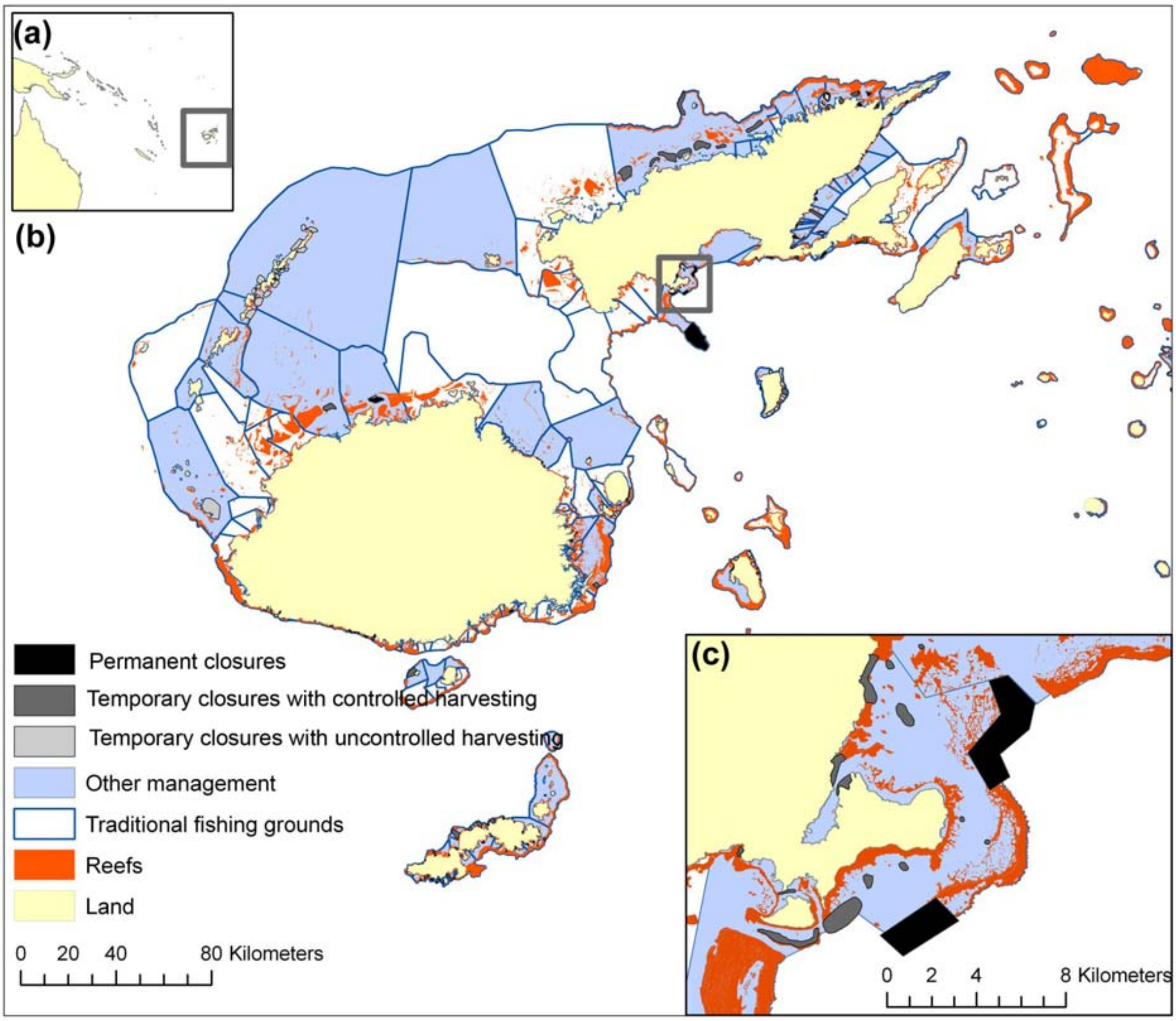

Figure 3.1. Marine inshore management in Fiji. Location of (a) Fiji in the western Pacific Ocean; (b) map of Fiji showing current locally managed marine areas, which include all permanent closures, temporary closures with controlled harvesting, temporary closures with uncontrolled harvesting, and other management (unshaded areas are traditional fishing grounds with no known management); and (c) part of the Kubulau traditional fishing ground and its permanent and temporary closures (square on map [b]).

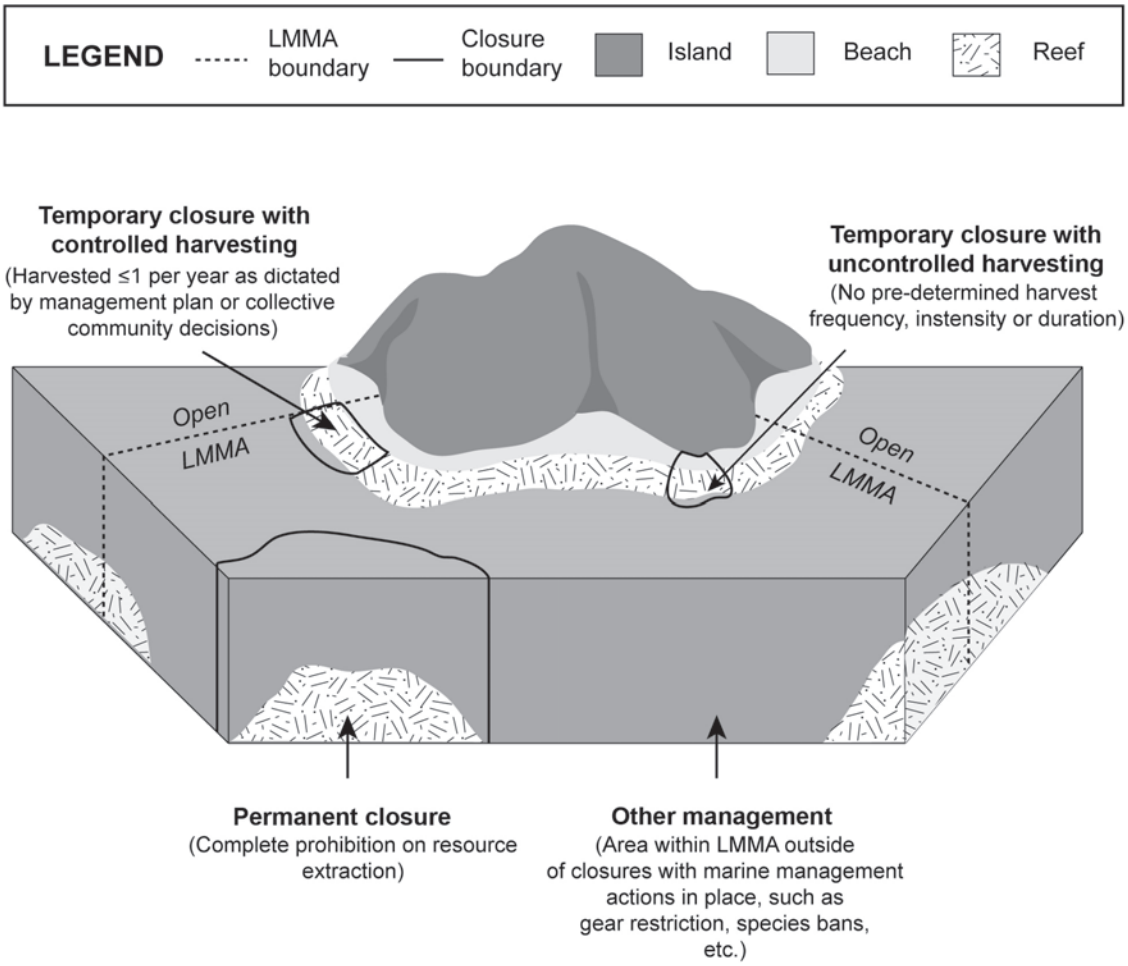

Figure 3.2. A schematic diagram of a locally managed marine area (LMMA). An LMMA is an area of inshore waters governed by those with traditional fishing rights and involving a collective understanding of, and commitment to, management intervention in response to threats to marine resources. As shown these areas can be subject to multiple, simultaneous forms of management.

3.3.2. Fiji’s inshore marine ecosystems

Fiji’s Protected Area Committee has identified 7 priority ecosystems (i.e., ecosystems of high priority for conservation because of their ecological role, cultural significance, uniqueness and rarity) in Fijian coastal and inshore marine waters (Jupiter et al*.* 2010; Table 3.1; Figure 3.3). National-level spatial information (from the Fijian Federal Government and the Intergovernmental Oceanographic Commission) is available only for mangroves, fringing reefs, nonfringing reefs, intertidal areas, and other benthic substrata (soft-bottom lagoons and seagrass combined in 4 depth classes [0-5 m, 5-10 m, 10-20 m, 20-30 m]; Table 3.1). I processed these data in ArcInfo 9.3 (ESRI, Redmond, California; Table 3.1) and assembled them into a map of Fijian marine ecosystems used for the gap analysis. Data processing took approximately 2 months.

Table 3.1. Spatial data on Fiji marine ecosystems, sources and data processing.

Priority ecosystem types	Mapped ecosystems	Source	Dataset details	Processing undertaken in ArcInfo 9.3 (ESRI)	Processing details
(1) Mangroves	(1) Mangrove[a]	Fiji Department of Forestry, Federal government	Digitized for main Fiji islands from 2001 Landsat ETM+ data	None	
(2) Reefs	(2) Fringing reefs[a] (3) Non-fringing reefs[a]	Fiji Department of Lands, Federal government	Digitized from aerial photographs captured in 1994 and 1996	Submerged and exposed reef data converted to polygons with X-tools pro	Polylines joined by hand to allow for conversion to polygons. Reclassified into fringing reef and non-fringing reef. Reefs that had sections less than 100m from the coastline were classified as fringing, all others were non-fringing Erased overlap with mangroves by overwriting reefs[b]
(3) Intertidal	(4) Intertidal[c]	Fiji Department of Lands, Federal government	Digitized from aerial photographs captured in 1986 and validated in 1995	None	Erased overlap with mangroves and reefs by overwriting intertidal mud flats[b]
(4) Soft-bottomed lagoons (5) Seagrass	(5) Other benthic, 0-5 m depth [d] (6) Other benthic, 5-10 m depth [d] (7) Other benthic, 10-20 m depth [d] (8) Other benthic, 20-30 m depth [d]	General Bathymetric Chart of the Oceans (IOC et al. 2003)	Interpolated from available information on contours, coastlines and land elevation	Contours derived	Raster converted to shapefile Erased overlap with mangroves, reefs and intertidal mud flats by overwriting benthic ecosystems
(6) Sandy cays/beaches	NA	NA	NA	NA	NA
(7) Coastal littoral forests	NA	NA	NA	NA	NA

[a] There are notable gaps in both the reef and mangrove maps when these are compared to Google Earth images. I do not expect, however, that these gaps will alter my results substantially
[b] Ecosystems that overwrote other ecosystems had maps assessed as more accurate based on their resolution and the amount of data processing they required
[c] Intertidal ecosystems include mudflats, rocky shores, and intertidal seagrass and algal assemblages
[d] Other benthic ecosystems included all marine ecosystems shallower than 30m for which more specific spatial information was not available across Fiji (e.g. seagrass, soft-bottom lagoons, sandy or rubble bottom). The connections between these and the two listed priority ecosystems are: 1. they include the priority ecosystems, as well as others; 2. they occur across similar depths; and 3. they are distinct from the other mapped ecosystems.

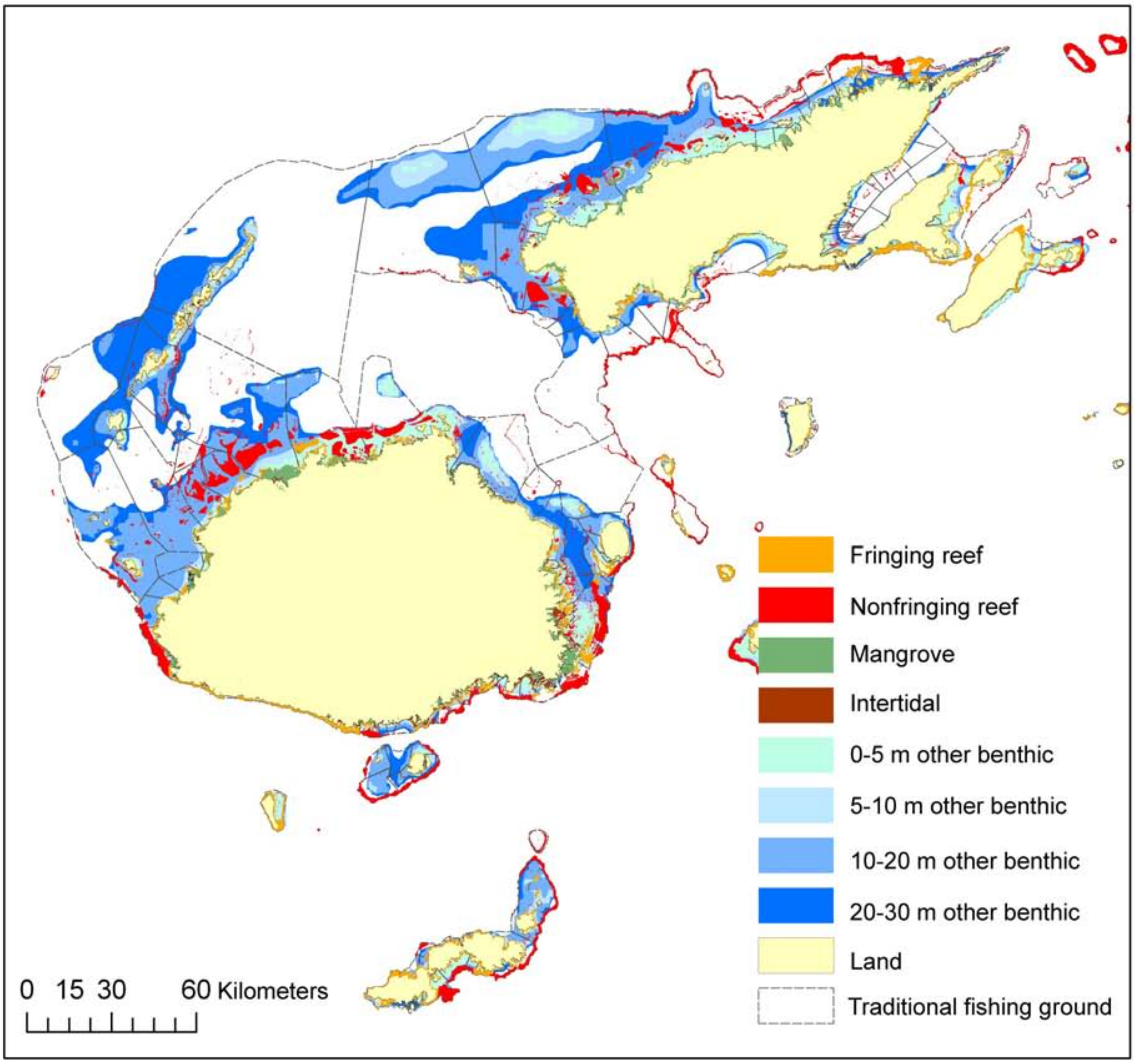

Figure 3.3. Fiji's inshore marine ecosystems.

3.3.3. Conservation objectives and effectiveness of management

I held a workshop in Suva, Fiji, in March 2010 with 12 experts in local flora and fauna and extensive experience with local management (Table 3.2 has additional information details on experts). The workshop had 2 main purposes: identification of ecosystem-specific conservation objectives on the basis of the national government's goal of managing 30% of inshore waters and assignment, by expert participants, of values of ecological effectiveness to selected species groups in each ecosystem (Jupiter et al*.* 2010). I call the assigned values (Table 3.3) ecological effectiveness scores. Experts selected species groups that they considered of national importance (e.g., fish). Identification of species groups allowed experts to more easily estimate the potential effects of different forms of management, effects that vary depending on, for example, the species' probability of being harvested by the fishing gear used in an ecosystem (Table 3.3).

Empirical data on ecological effectiveness in Fiji are unavailable at a national level. Expert opinion was therefore the only source of information. Experts are often consulted on the effectiveness of different forms of management because it is difficult to collect empirical data (Pomeroy et al. 1997; McClanahan et al*.* 2005a; Martin et al. 2005). Elicitation of expert opinion can be undertaken with different levels of quantitative rigor, depending on the amount of data available to support expert judgments (e.g., Martin et al. 2005).

Given the diversity of backgrounds of the experts, I initiated the workshop through dialectic inquiry (Mitroff et al. 1979; Schweiger 1986), in which opposing views on ecological effectiveness for different species groups were presented and discussed. I considered this preferable to surveying each expert (Pomeroy et al*.* 1997; McClanahan et al*.* 2005a; McClanahan et al*.* 2005b) because it allowed evaluation of the information and assumptions by all experts and because group participation and discussion is critical for the acceptance of results and commitment to acting on them (Schweiger et al*.* 1986). This approach also helped link this study to practical outcomes.

I described the concept of ecological effectiveness to the experts and asked them to discuss and estimate effectiveness scores from 0 (ineffective) to 1 (maximum effectiveness) at 0.05 increments for the different forms of management within LMMAs. Final scores represented the consensus on ecological effectiveness among the experts (Table 3.3). Scores were based on the response to fishing and mobility of different species within species groups, limitations and selectivity of fishing gear, changes to species' habitats associated with existing fishing practices, and accessibility of ecosystems to fishers. Permanent closures were given a score of 1. A score of 0.5 indicated that, per unit area, a form of management would maintain populations at half the densities in permanent closures, averaged over time.

These scores were based on an assumption of full compliance with management because I lacked spatial data on compliance. I believe that a high level of compliance is likely because the forms of management are community driven (Johannes 2002). However, full compliance is unlikely to be achieved consistently.

Table 3.2. Institutions represented at the workshop held in Suva, Fiji, in March 2010.

Title	Organisation	Expertise
Director/Conservation Scientist [a]	Wildlife Conservation Society Fiji Program	Coral reef and mangrove ecology; systematic planning; community-based management
Marine Biologist [a]	Wildlife Conservation Society Fiji Program	Fish and invertebrate ecology; community-based management
Government Liaison Officer [a]	Wildlife Conservation Society Fiji Program	Fish and invertebrate ecology; community-based management
Senior Manager [b]	Wetland International-Oceania	Fish ecology and taxonomy; community-based management
Senior Research Officer [a]	Institute of Applied Sciences, University of the South Pacific	Coral reef ecology; systematic planning; community-based management
Senior Research Officer [a]	Institute of Applied Sciences, University of the South Pacific	Coral reef ecology; systematic planning; community-based management
Senior Fisheries Officer [b]	Fiji Department of Fisheries	Marine species conservation; spawning aggregations; fisheries management
Fisheries Officer	Fiji Department of Fisheries	Marine species conservation (cetaceans)
Fisheries Officer [a]	Fiji Department of Fisheries	Coral ecology and taxonomy
Marine Program Coordinator	International Union for Conservation of Nature	Fisheries management
Resource Management Unit Officer	Fiji Department of Environment	Management; national biodiversity targets
GIS Officer	National Trust of Fiji	Systematic planning

[a] Indicates participant is currently an active member of the Fiji Locally Managed Marine Area Network
[b] Indicates participant was formerly an active member of the Fiji Locally Managed Marine Area Network

Table 3.3. Ecological effectiveness[a] of each form of management for conservation of selected species groups in each ecosystem.

Ecosystem and species group[b]	Permanent closures	Temporary closures with controlled harvesting	Temporary closures with uncontrolled harvesting	Other management
Total area (km^2)	122	233	212	17159
Fringing reefs				
corals	1	0.80	0.50	0.40
targeted invertebrates	1	0.70	0.10	0.20
nontargeted invertebrates	1	0.90	0.60	0.45
targeted fish	1	0.80	0.15	0.20
nontargeted fish	1	0.90	0.50	0.45
coralline algae	1	0.80	0.50	0.40
Range	*1*	*0.70-0.90*	*0.10-60*	*0.20-0.45*
Nonfringing reefs				
corals	1	0.80	0.55	0.40
targeted invertebrates	1	0.70	0.10	0.20
nontargeted invertebrates	1	0.90	0.80	0.45
targeted fish	1	0.80	0.15	0.20
nontargeted fish	1	0.90	0.60	0.45
coralline algae	1	0.80	0.55	0.40
Range	*1*	*0.70-0.90*	*0.10-0.80*	*0.20-0.45*
Mangrove				
targeted invertebrates	1	0.80	0.15	0.20
nontargeted invertebrates	1	0.95	0.85	0.50
targeted fish	1	0.50	0.10	0.15
nontargeted fish	1	0.60	0.30	0.30
mangrove	1	0.95	0.85	0.25
seabirds	1	0.95	0.85	0.20
bats	1	0.95	0.85	0.25
Range	*1*	*0.50-0.95*	*0.10-0.85*	*0.15-0.50*
Intertidal				
targeted invertebrates	1	0.70	0.10	0.20
nontargeted invertebrates	1	0.90	0.80	0.45
targeted fish	1	0.80	0.50	0.20
nontargeted fish	1	0.90	0.80	0.45
seabirds	1	0.95	0.20	0.25
Range	*1*	*0.70-0.90*	*0.10-0.80*	*0.20-0.45*
Other benthic substrata[c]				
targeted invertebrates	1	0.70	0.30	0.20
nontargeted invertebrates	1	0.90	0.80	0.45
targeted fish	1	0.80	0.50	0.20
nontargeted fish	1	0.90	0.80	0.45
Range	*1*	*0.70-0.90*	*0.30-0.80*	*0.20-0.45*

[a] Effectiveness values range from 0 (management form not effective) to 1 (management form fully effective; assumed to be provided by permanent closures) and are given to the nearest 0.05.

[b] Species groups divided into targeted (i.e., species deliberately sought for subsistence or commercial purposes) and nontargeted because management of a fishing ground is likely to increase abundance of targeted species to a greater extent than nontargeted species.

[c] Other benthic substrata consists of 4 depth classes, all of which had the same selected species groups and effectiveness scores

3.3.4. Gap analyses

To assess whether objectives for representation of ecosystems set at the March 2010 workshop were achieved, I collated information on the distribution of ecosystems, different forms of management, and ecological effectiveness. I then applied 3 alternative gap analyses, each with different assumptions: (1) all parts of LMMAs (including closures and other management) conserve species and ecosystems effectively; (2) closures conserve species and ecosystems whereas areas outside of closures, open to varying levels of resource extraction, do not; and (3) different forms of management permitting different levels of resource extraction vary in their ability to conserve species and ecosystems. Assumptions 1 and 2 are typical of gap analyses (e.g., Mora et al*.* 2006; Wood et al*.* 2008; Weeks et al*.* 2010). I based assumption 3) on the consensus about ecological effectiveness attained at the 2010 workshop.

Spatial data were available for 4 forms of management: permanent closures, temporary closures with controlled harvesting, temporary closures with uncontrolled harvesting, and the combination of other forms of management in parts of LMMAs outside mapped closures. I updated the boundaries of LMMAs and closures presented in Govan et al. (2009), which resulted in a total of 149 LMMAs and 216 closures (Figure 3.1). In total the LMMAs and closures covered, respectively, about 60% (~17,726 km^2) and 2% (~567 km^2) of the total extent of traditional fishing grounds. I overlaid LMMAs, closure maps, and the ecosystem map in ArcInfo (version 9.3) and calculated the area of each ecosystem subject to each form of management (Appendix 8.2).

To apply assumption 1, I calculated the area of each ecosystem type covered by LMMAs (e.g., 1 km^2 of mangrove within an LMMA counted as 1 km^2 of effectively managed mangrove). To apply assumption 2, I calculated the area of each ecosystem type covered by closures (e.g., 1 km^2 of mangrove within a closure counted as 1 km^2 of effectively managed mangrove). To apply assumption 3, I used different scores of ecological effectiveness for different species groups within the mapped ecosystems (Table 3.3). I calculated the areas that were effectively managed within each ecosystem type by multiplying the percent area of each ecosystem under each form of management by the effectiveness scores attributed to each species group within that ecosystem:

$$E = \sum_{A \to D} \frac{(S_A)(t_A)}{T} X100 \qquad (1)$$

where E is the percentage of effectively managed area for the selected species group, $A \to D$ is the different forms of management (Table 3.3), S is the effectiveness score attributed to different forms of management for the species group (Table 3.3), t is the area of the ecosystem covered by each form of management, and T is the total area of the ecosystem within the Fijian traditional fishing grounds. I then identified the highest and lowest E (i.e., the maximum and minimum percent areas of each ecosystem type effectively protected across all species groups).

3.4. Results

3.4.1. Conservation objectives and ecological effectiveness

Ecological effectiveness scores varied from 0.10 to 1 (Table 3.3). Temporary closures with controlled harvesting had relatively high scores (0.50-0.95). Temporary closures with uncontrolled harvesting and other management areas had scores of 0.10-0.85 and 0.15-0.50, respectively.

Experts provided 4 broad statements of opinion to support their effectiveness scores. First, temporary closures with uncontrolled harvesting are less effective at protecting targeted invertebrates than targeted fish because fish rapidly learn to avoid highly fished areas. Second, targeted invertebrates and fish are more effectively protected within managed areas outside closures and on nonreef substrata than on reefs protected by temporary closures with uncontrolled harvesting. This difference in effectiveness was attributed to intense concentration of fishing effort within temporary closures with uncontrolled harvesting during openings. Third, the effectiveness of temporary closures with controlled and uncontrolled harvesting in mangroves is similar for targeted and non-targeted fish because use of gill nets, a relatively unselective gear type, is high. Fourth, differences in ecological effectiveness between fringing and nonfringing reefs are due to the greater accessibility of fringing reefs and to greater effects from trampling during fishing activities.

3.4.2. Gap analyses

The LMMAs ranged in size from 0.01 to 4,168 km^2 (mean=119 km^2, median=11 km^2). When all parts of LMMAs were assumed to conserve species and ecosystems effectively (assumption 1), conservation objectives for all ecosystems were exceeded (Figure 3.4) and coverage of all ecosystems was >40%. The highest coverage was for other benthic substrata at depths of 0-5 m and 5-10 m (59% and 60% respectively).

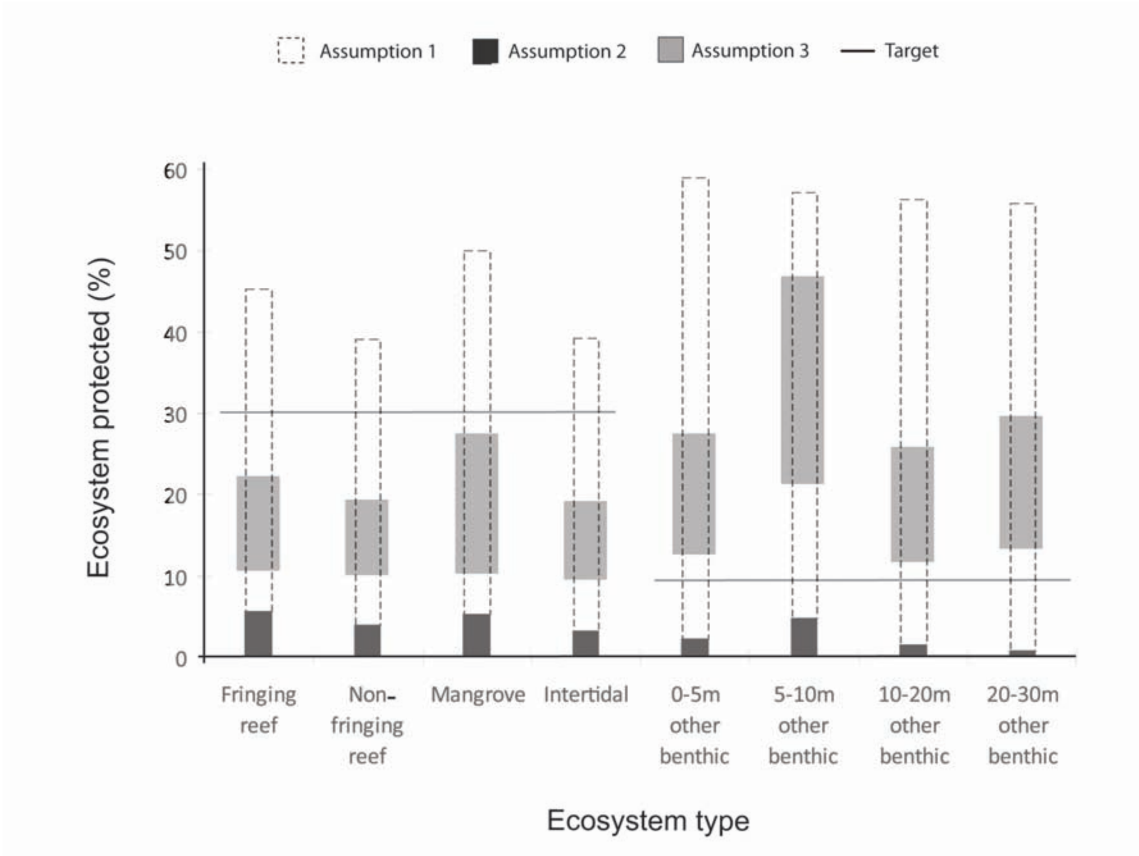

Figure 3.4. Percentages of 8 ecosystems protected effectively on the basis of 3 assumptions about ecological effectiveness of the various forms of management: assumption 1, all parts of locally managed marine areas (including closures and other management) conserve species and ecosystems effectively; assumption 2, closures conserve species and ecosystems, whereas areas outside closures, open to varying levels of resource extraction, do not; assumption 3, different forms of management permitting different levels of resource extraction vary in their ability to conserve species and ecosystems. Ranges of percentages for assumption 3 are based on upper and lower effectiveness scores in Table 3.3. Grey horizontal lines indicate objective for each ecosystem.

Closures ranged from 0.01 km^2 to 66 km^2 (mean=3 km^2, median=0.73 km^2). When closures were assumed to conserve species and ecosystems, but areas outside closures were assumed to offer no protection (assumption 2), none of Fiji's conservation objectives were met (Figure 3.4). Coverage ranged from a maximum of 6% for fringing reefs to 1% for intertidal ecosystems.

When different forms of management were assumed to vary in their ability to conserve species and ecosystems (assumption 3), Fiji met or exceeded its conservation objectives only for other benthic substrata in all depth classes (Figure 3.4 and Appendix 8.2). Additional coverage of between 10% and 20% of fringing reef, nonfringing reef, mangrove, and intertidal ecosystems was still required to meet objectives. For fringing reefs, one of the most heavily fished ecosystems, to meet the objective required the addition of either 402 km^2 of permanent closures, 574 km^2 of temporary closures with controlled harvesting, 1340 km^2 of temporary closures with uncontrolled harvesting, or 2010 km^2 of other management. The extent of unmanaged fringing reef in Fiji was 867 km^2.

3.5. Discussion

This study was designed to inform an impending policy commitment by the Government of Fiji to complete a national marine gap analysis, but the approach is applicable to other countries where empirical data on ecological effectiveness are limited. The 3 assumptions varied in their validity.

Assuming that all parts of LMMAs effectively conserve species and ecosystems leads to inferring that all conservation objectives were achieved. However, despite the rapid increase in the number of community-based conservation initiatives, the limited data available suggest the abundances of species harvested on Fijian inshore reefs are declining, with many harvested invertebrates already at low abundances (Teh et al*.* 2009). Although Fijian LMMAs span large areas and LMMAs have been implemented by communities across the Pacific, declining resources indicate existing management measures within LMMAs may not be sufficient to ensure long-term sustainability of inshore fisheries.

Assuming closures conserve species and ecosystems and areas outside closures open to varying levels of resource extraction do not, Fiji's conservation objectives were not achieved. Management through closures has 3 related limitations in Fiji. First, over 99% of the closures were extremely small (median size 0.73 km^2), similar to other parts of the Asia-Pacific region (Bartlett et al*.* 2009; Weeks et al*.* 2010). On the basis of larval and adult dispersal and the size of self-sustaining populations of benthic species, closures of 10-100 km^2 are recommended to protect most species associated with benthic ecosystems (Halpern and Warner 2003). The second limitation of closures was that communities in Fiji are unlikely to close 30% of their traditional fishing grounds temporarily or permanently (Agardy et al*.* 2003). Third, communities are unlikely to distribute closures evenly across ecosystems because they prefer locations within view of villages to improve compliance (e.g., Aswani and Hamilton 2004a). Communities could, however, be encouraged to adopt complementary forms of management (e.g., gear or species restrictions) that contribute to conservation objectives and are more socially acceptable (Johannes 2002).

The assumption that different forms of management vary in their effectiveness recognizes species- and ecosystem-specific variation in ecological effectiveness in Fiji, including the adverse effects of temporary closures with uncontrolled harvesting and the partial protection offered by management operating outside closures but within LMMAs. On the basis of this assumption, Fiji still did not achieve its conservation objectives, but considerable progress toward them was made. To meet conservation objectives, I recommend a combination of larger and more numerous permanent closures or temporary closures with controlled harvest within LMMAs and land-based management that mitigate pollution and nutrient runoff. These recommendations were recently presented to administrators from Fiji's 14 provinces to identify candidate sites for protection and management that could fill the gaps in the representation of ecosystems while meeting both local and national conservation goals (Jupiter et al. 2011).

The greatest challenge to incorporating ecological effectiveness into gap analyses or systematic planning exercises is the paucity of empirical data (Agardy et al*.* 2003;

Edwards et al. 2010). Within most of the scientific literature, conclusions have been drawn from observations made inside and outside permanent closures (Russ 2002; Lester et al. 2009), although research on the relative effectiveness of other forms of management is emerging (e.g., Cinner et al. 2005; Bartlett et al. 2009). In this context, expert opinion is essential, but has limitations. First, experts are unlikely to have full understanding of all ecosystems and management approaches. Even if information is available, people have limited ability to access and process it (Einhorn et al. 1977). Second, not all individuals with knowledge of the effects of management on different species can be included in a participatory process. Third, perceived effectiveness will be influenced by individuals' social and economic background, such as ethnicity or employment (e.g., McClanahan et al. 2005b). Finally, the opinions of individuals are likely to change given social pressures within a workshop or a community (Einhorn et al. 1977).

I suggest an adaptive-management approach where few data on ecological effectiveness are available, whereby scores are elicited from experts and the results are then tested through field surveys and refined as data accumulate (Salafsky et al. 2002). Expert elicitation can provide impetus for collecting empirical data. Results from this study are already helping garner funds for field experiments on the ecological effectiveness of Fiji's management.

I concentrated on ecological effectiveness in relation to mapped ecosystems, but recognize that the effectiveness of management depends ultimately on other factors, such as the productivity of ecosystems, protection of biological processes, the social and economic characteristics of managed and surrounding areas, and compliance (Hockings et al. 2006). Barriers to compliance in Fiji include conflict between customary management rules and both national legal frameworks and incentives to fish from growing global markets (Clarke and Jupiter 2010). For example, the Fisheries Act does not grant authority to those with traditional fishing grounds to legally enforce customary management (Clarke and Jupiter 2010). Imminent legislative reform seeks to rectify this.

Considering the varying ecological contributions of different forms of management is important for 2 reasons. First, achievement of conservation objectives can be evaluated in countries where large permanent closures are not feasible, as in many Pacific island nations (Johannes 2002). Second, it facilitates the design of complementary forms of management for particular social and ecological contexts. Attempts to achieve all marine conservation objectives through permanent closures are likely to create unnecessary conflict (Agardy et al. 2003).

Chapter 4

Where do national and local conservation actions meet? Simulating the expansion of opportunistic and systematic approaches to conservation into the future[3]

[3] Mills M., Adams V.M., Ban N.C., Jupiter S.D. and Pressey R.L. Where do national and local conservation actions meet? Simulating the expansion of systematic and opportunistic approaches to conservation in the future. Conservation Letters. In review.

4.1. Abstract

The marginal benefits of systematic over opportunistic selection of protected areas are rarely measured, even though this information is crucial to investing limited conservation resources effectively. I developed a method to predict the marginal benefits of systematic over opportunistic approaches to conservation over time. I tested it in Fiji, where ambitious national conservation goals for inshore marine waters rely on communities for implementing the required management. I used Maxent to develop a suitability layer for different forms of marine management based on predictor variables derived from interviews with key informants. This suitability layer, together with data on established MPAs and the software Marxan with Zones, informed simulations of the expansion of both opportunistic and systematic approaches to conservation. Within constraints on the additional extent of MPAs, the opportunistic approach achieved quantitative conservation objectives for half of the ecosystems, while all objectives were achieved or nearly achieved with the systematic approach. By defining the likely upper and lower bounds of plausible futures given different decisions about conservation investments, this work was designed to guide conservation strategies and conservation action in Fiji. This work is currently influencing the development of policies in Fiji to promote a more strategic use of limited conservation resources.

4.2. Introduction

Ongoing biodiversity loss and limited resources for conservation require effective and cost-efficient conservation actions (as discussed in Chapter 1, section 1.1.4). Most conservation actions have been opportunistic (Pressey et al. 2000; Scott et al. 2001; Rouget et al. 2003). Opportunistic actions have been implemented effectively even where central governments have weak capacity for coordination when they focus on local objectives, integrate local knowledge and customs, and involve local resource users (e.g. Johannes 2002; Lane 2008). Theoretically, opportunistic actions can coalesce into ecologically and socially functional MPA networks. However, they can also waste limited resources by focusing on areas that contribute marginally to regional-scale objectives (Pressey and Tully 1994).

An alternative approach is systematic planning (described in Chapter 1, section 1.1.5) (Margules and Pressey 2000), characterized by explicit objectives and consideration of spatial context, through complementarity and connectivity, to guide selection of conservation areas. Intuitively, this more strategic approach should allocate conservation actions more efficiently to achieve objectives, and many millions of dollars have been invested in its application. However, few spatial prioritisations have guided extensive on-ground implementation (Knight et al. 2006), and the exceptions (e.g., Pressey et al. 2009; McCook et al. 2010) were facilitated by atypical circumstances, such as lack of private tenure and uncomplicated governance. The marginal benefits of systematic over opportunistic approaches have rarely been measured and are likely to be context-specific, and there is debate over the most effective blend of the two (Knight and Cowling 2007; Pressey and Bottrill 2008). Such comparisons, and better understanding of the respective strengths and limitations of systematic and opportunistic approaches, are vitally important if we are to invest limited resources effectively.

In this thesis, I have defined systematic and opportunistic approaches as mutually exclusive. My main reason was to understand the bounds on the decision space, the potential distribution of outcomes from each conservation decision, within which systematic planners are working. There are existing attempts, both in the literature and in practice, to integrate opportunistic and systematic approaches (e.g. by scaling up conservation actions and scaling down systematic plans; see Chapter 1, section 1.1.5; Armada et al. 2009; Henson et al. 2009; Ban et al. 2009a; Pressey and Bottrill 2009; Knight et al. 2010; Game et al. 2010), but these attempts are likely to produce results within the decision space that I define here.

Of the four published studies that estimated the marginal benefits of systematic over opportunistic approaches, three were retrospective and one was predictive. The retrospective studies (Rebelo and Siegfried 1992; Pressey and Taffs 2001; Hansen et al. 2011) compared the observed, at least partly opportunistic, representation of ecosystems within protected areas to the potential representation had more systematic approaches been taken. The predictive study (Pressey and Tully 1994) projected potential representation of ecosystems from expansion of protected areas with both opportunistic and systematic approaches. Retrospective assessments provide lessons for the future, while the potential of predictive comparisons, which has barely been explored, is to construct alternative futures arising from different policy settings, thereby informing decision makers about the consequences of proceeding in alternative ways. Here, I develop a predictive comparison of opportunistic and systematic approaches that builds on the existing literature in three ways. First, this is the first marine case study. I apply these method to the expansion of community-based MPAs in Fiji. Second, whereas Pressey and Tully's (1994) opportunistic scenario came from specific areas proposed for reservation by an agency, ours required the development of new methods, including modelling suitability for different types of community-based MPAs and emulating the expansion of opportunistic MPAs with a decision tree linked to spatial data. These innovations are likely to be broadly applicable to community-based conservation. Third, this study is closely associated with policy and practice through the Fiji National Protected Area Committee, engaging them in a dialogue about strategies for achieving conservation goals. This is a different type of engagement to that of Pressey and Tully (1994).

This chapter has two aims. The first is to predict the marginal benefits for representation of ecosystems of systematic over opportunistic approaches to conservation and to test this method in inshore marine waters in Fiji. I address three questions: (1) given current trends in opportunistic actions, will their expansion achieve national conservation objectives by 2020? (2) considering realistic constraints on conservation actions in Fiji, would systematic allocation of closures achieve national conservation objectives by 2020? and (3) what is the difference in achievement of objectives between the opportunistic and systematic approaches?

The second aim is to inform policy initiatives in Fiji. This work has full support from the Fiji National Protected Area Committee. Simulation of expanding MPAs is a priority of the Fijian Department of Environment for 2011 and is seen as contributing to the Fiji National Biodiversity Strategy and Action Plan under the Inshore Fisheries thematic section. This work will therefore have direct impact on policy and allocation of conservation resources.

4.3. Methods

4.3.1. Planning region and policy context

The study region for this chapter is the same as the study region in Chapter 3 (see Chapter 3, section 3.3.1 for details). Fiji's national government has committed to protecting 30% of its inshore waters within MPAs by 2020 (Jupiter et al. 2010). Progress towards this commitment has mostly been through opportunistic implementation of community-based MPAs referred to as locally managed marine areas (LMMAs). The Fiji LMMA network of resource managers shares knowledge and experience and supports the national government's 30% commitment (Jupiter et al. 2010).

4.3.2. Data and conservation objectives

Here, I used the same data and conservation objectives as in Chapter 3. I used all available national-scale data on the spatial distribution of the following marine ecosystems: fringing reefs, non-fringing reefs, mangroves, intertidal and 'other benthic substrata'. The latter included soft-bottomed lagoons and seagrass and was divided into four depth classes (0-5m, 5-10m, 10-20m, 20-30m; see Table 3.1 for details). The ecosystem-specific conservation objectives and scores for management effectiveness are also from Chapter 3 (see section 3.3.3 for details). The objectives were 10% representation of other benthic substrata in all depth classes and 30% for other ecosystems. The 10% objectives have been achieved with existing management, so I used the 30% objectives for the systematic scenario, below. The scores for management effectiveness were a range of relative per-unit-area contributions to objectives of each form of management. I used these relative contributions in both the opportunistic and systematic scenarios. For each form of management, I chose the most common minimum contribution to focal species groups across ecosystems. One reason was that Marxan with Zones uses only one value of contribution per form of management. Another was my preference for conservative estimates of relative contributions to acknowledge the requirements of most focal species groups. I subdivided the planning region into planning units for assessment and comparison as potential future closures. These planning units were mostly grids, trimmed at closure boundaries, the coastline, boundaries of fishing grounds (so each unit was associated with only one fishing ground), and the outer bounds of the study region, with modal size of 0.5 km^2.

4.3.3. Suitability layer for closures

For both the opportunistic and systematic scenarios, I modelled the suitability of planning units outside existing closures for the establishment of new closures. First, I conducted 11 semi-structured interviews with key informants to identify factors that influence opportunities for, and constraints on, implementing closures in Fiji (survey provided in Appendix 8.3). From these, I identified spatial predictors of existing closures (Table 4.1). I used snowball sampling to select key informants with more than 2 years of experience in establishing LMMAs and who were members of partner organizations within the Fiji LMMA network. Key informants belonged to the Institute of Applied Sciences at the University of South Pacific, Wildlife Conservation Society, and Wetlands International-Oceania, all members of the Fiji LMMA network (for additional information see Table 3.2). The spatial predictors were: distance from another closure; proportion of inshore fishing ground (<3 km from the coast) within closures; distance from nearest road; distance from nearest village; presence of a provincial resource management support team; and ecosystem type (see Table 4.1 for rationale). With these predictors, I used Maxent (Phillips et al. 2006) to develop maps of suitability for new closures within fishing grounds. I used Maxent because it is robust to the limitations of presence-only data that indicate where features of interest have been observed but not where they have been looked for and not observed (Phillips et al. 2006). I interpreted data on the distribution of closures as presence-only because I did not know which areas outside existing closures might have been considered by villagers for closures but found to be unsuitable.

To develop the Maxent model, I divided the fishing grounds into grid squares of 1 ha, allowing relatively precise estimates of distance for some of the predictors. I developed four suitability maps, one for each type of closure (permanent, controlled and uncontrolled) and one for all closures combined. To train the model, I associated the centroid of each grid square within existing closures (n = 2153) with the six predictors. Background points, selected from outside closures but only in fishing grounds with closures, informed the model about variations in values of the six predictors within the Fijian seascape (Elith et al. 2011). A random selection of background points across all fishing grounds would have identified fishing grounds engaged with the Fiji LMMA network themselves as an important influence on suitability for closures. This would have incorrectly obscured the signal from the six predictors in fishing grounds where communities have not begun to collaborate with the Fiji LMMA network. I tested model performance using the area under the receiver operator curve (AUC), where 1 indicates that the model reflects the current distribution of closures perfectly and 0.5 indicates a model no better than random at predicting the distribution of closures (Phillips et al. 2006).

Table 4.1. Factors identified by key interviewees as important in determining the presence, size and location of closures in Fiji, and spatial predictors used in Maxent.

Factors influencing presence, size and location of closures, identified by interviewees	Spatial predictors used in Maxent model	Rationale provided by interviewees and/or scientific literature supporting use of spatial predictor
1) Perceived benefits of being associated with an international conservation NGO	Data not available	Villagers are attracted to conservation projects by the direct benefits received from NGOs (e.g., employment opportunities) or indirect benefits of being associated with them (e.g., help with leveraging funds from other organizations, improvement of village status relative to surrounding villages) (Foale 2001).
2) Establishment of closures by adjacent villages	Distance from nearest other closure within any fishing ground [a]	After a village joins the LMMA network, the villagers present their work at provincial meetings (LMMA), initiating interest from other villages in the same province (USP 2007; WRI 2008). In other regions, such as the Philippines, communities also become interested in establishing MPAs after hearing from others about their potential benefits (Alcala and Russ 2006).
3) Perception of resource decline	Data not available on a national-scale	Villagers will manage their natural resources when they see them as threatened or in decline (Johannes 2002).
4) Need for access to traditional fishing grounds	Proportion of inshore traditional fishing ground (<3 km from coast) within closures [a, b]	Villagers are unlikely to change their preferred fishing areas and abide with new resource regulations if they do not have suitable alternative fishing areas (Abernethy et al. 2007; Daw 2007). Because few villagers have access to motorboats, most people are restricted to fishing and collecting marine resources within approximately 3 km of their villages (Adams et al. 2011). Additionally, the scope for fishers to change fishing locations in Fiji is restricted by the limits of traditional fishing grounds and use rights based on their lineages.
5) Accessibility and visibility from village/ability to enforce and monitor resource regulations	Distance from nearest road[2]	The spatial mobility of fishers is limited by transport and fuel costs (Begossi 2001). Fishing grounds closer to roads are more accessible.
	Distance from nearest village[2]	Enforcement of regulations is highly reliant on vigilance by members of the village. Managed areas must be visible from villages to allow effective enforcement (Aswani & Hamilton 2004; Leisher *et al.* 2007).
	Presence of a provincial resource management support team[1]	One of the responsibilities of the provincial resource management support teams is to integrate rules from LMMAs into provincial legislation (WRI 2008). When regulations are so integrated, they can be legally enforced (Tawake 2007).

6) Ecosystem health, productivity and type	Ecosystem type [c]	Villagers choose to protect either: (1) the most productive (e.g., coral reefs) and healthy ecosystems to get the maximum benefit from management; or (2) degraded ecosystems to promote recovery. Information on ecosystem health and productivity was not available. However, ecosystem type could be a useful surrogate because some are more productive than others (e.g., mangroves and reefs are more productive than other benthic substrata, Mumby et al. 2004; Mason et al. 2005).

[a] Map created with data provided by the Fiji Locally Managed Marine Area network

[b] Map created with data provided by the Fijian National Government

[c] Map from Chapter 3

4.3.4. Opportunistic scenario

I simulated the expansion of opportunistic closures to estimate the extent to which they might achieve national conservation objectives by 2020. I simulated the possible future expansion of closures in Fiji in a way that reflected past opportunistic decisions. The simulation emulated the approach of the Fiji LMMA network to encouraging management. I based the simulation on Fiji LMMA reports (e.g., LMMA 2003), discussion with LMMA members, and the suitability models from Maxent. I included existing closures in every simulation as starting points for expansion. Because there was a stochastic element in the simulation model (Figure 4.1), I ran 100 repeat simulations, each for ten annual time steps (2011-2020), to produce 100 maps of potential future closures.

The simulation steps (detailed in 'simulation steps and assumptions' and in Figure 4.1) were repeated iteratively within each yearly time step. Due to the simulation's stochastic selection of percentages of traditional fishing grounds to be closed and areas of individual closures, the exact area closed in a yearly time step could not be specified. Therefore, the simulation was calibrated to select on average 90 km^2 each year, reflecting the average annual area closed when the expansion of locally managed marine areas was peaking (between 2002-2004). This rate gave the most optimistic picture of achievement of conservation objectives by opportunistic expansion, allowing us to simulate the maximum potential achievements of opportunistic actions. After closures had been allocated to fishing grounds they were classified as permanent, controlled or uncontrolled (details in 'classification of closures'). All other planning units within fishing grounds containing closures were classified as 'other management', reflecting practices by the Fiji LMMA network (Jupiter at el. 2010).

Simulation steps and assumptions

In designing the simulation, I made two main assumptions. First, I assumed that once a traditional fishing ground had 30% or more of its area in closures, the Fiji Locally Managed Marine Area network would not approach additional villages within that traditional fishing ground for further expansion of closures. This was based on the Fiji Locally Managed Marine Area Network's stated commitment to help the government achieve its objective of 30% representation of inshore waters under protection (Jupiter et al. 2010), while aiming to distribute costs and benefits of conservation equitably among villages. I therefore restricted the simulation to eligible traditional fishing grounds, defined as those with less than 30% already closed, and split these traditional fishing grounds into two groups. The first group was *Empty*, meaning that there were currently no closures within that fishing ground. The second group was *Locally Managed Marine Area (LMMA)*, meaning that the fishing ground was an existing *LMMA* and already contained some closures. In this simulation, *LMMA* fishing grounds were only engaged one additional time over the 10-year period, with closures added before the fishing grounds were removed from the eligible list. *Empty* fishing grounds followed the same engagement rule: if, after their first engagement, they had less than 30% closed, then they were engaged at most once more in this simulation. To date most LMMAs have 10-20% of their areas in closures (LMMA 2005).

The second assumption was that *Empty* and eligible *LMMA* fishing grounds would be engaged by the Fiji Locally Managed Marine Area network equally often. Therefore, for any iteration in the simulation, I selected from the two groups based on a random coin toss (Bernoulli random variable with $p = 0.5$).

I coded planning units as protected if they were already closures. I coded planning units as available for establishment of closures if they were open and at least 1 km away from other closures. This followed the practice of avoiding close aggregation of closures and reflected the current minimum distance between existing closures.

To inform the simulation about how much area to close within each fishing ground and the size of each closure, I developed probability distributions of these variables. I used the frequency distribution of closure size to estimate the probability distribution for size of closures using maximum likelihood methods in Matlab. The best fit model was a lognormal distribution (fitted parameters of μ =-0.3269 and σ = 1.6617). The chi-square goodness of fit test indicated that the fit was adequate ($\chi2$ = 14.5593, df = 11, p = 0.205). I used the frequency distribution of percentage of fishing ground closed to estimate in Matlab the probability distribution of percentages. The best fit model was a beta distribution (fitted parameters of $\alpha = 0.4077$ and $\beta = 1.3707$). The chi-square goodness of fit test indicated that the fit was adequate ($\chi2 = 0.61$, df = 14, $p = 0.717$). For the simulation, areas to close within each fishing ground and sizes of closures were picked randomly from these frequency distributions (Figure 4.1).

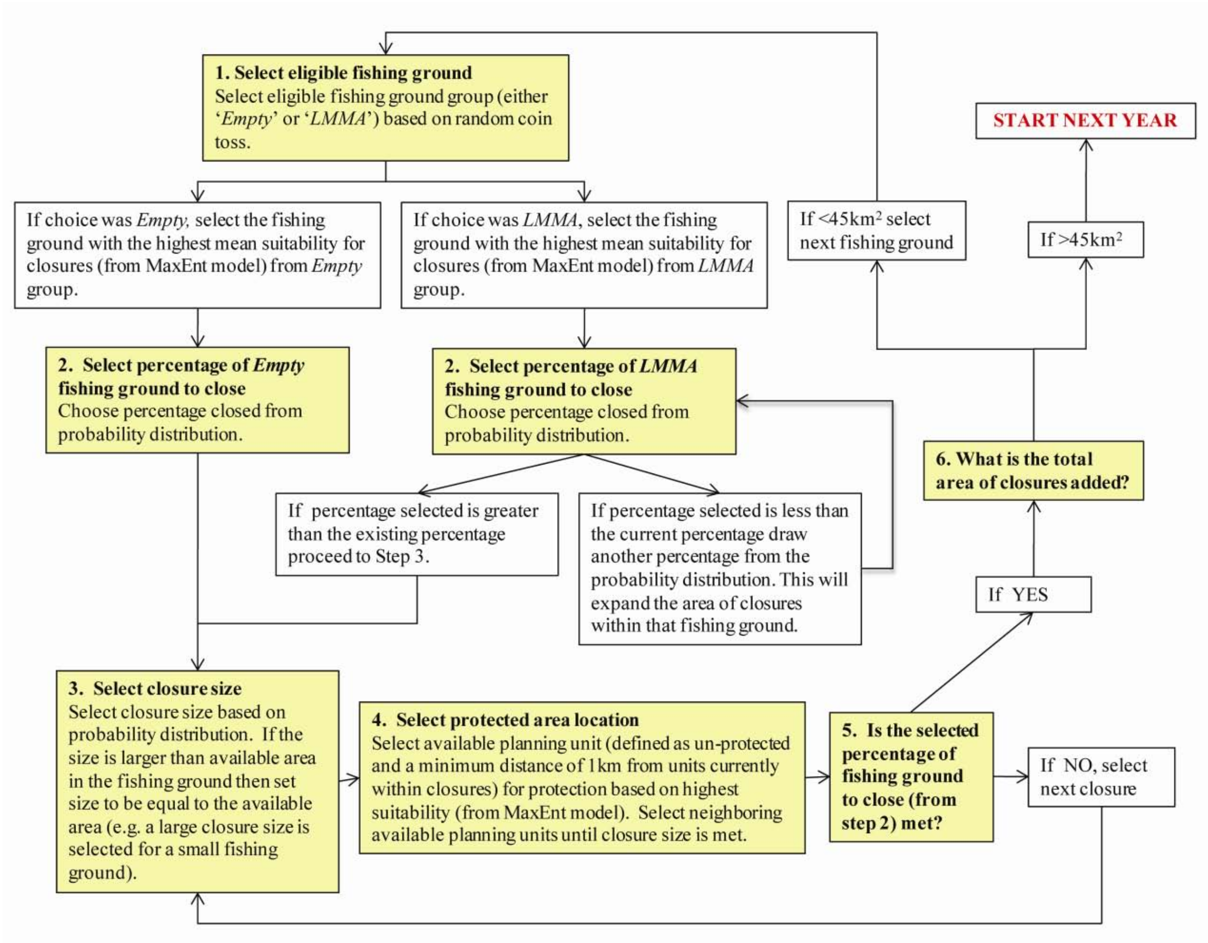

Figure 4.1. Flow diagram of simulation steps. Yellow boxes represent steps. White boxes represent alternative routes within those steps. The steps were applied multiple times within each year until an annual average area of 90 km^2 was placed in closures. Each simulation ran for 10 years. The simulation was repeated 100 times to produce 100 alternative configurations of opportunistically established closures.

Classification of closures

Each simulated future closure was assigned a closure status (permanent, controlled or uncontrolled) based on the unit's distance from the nearest road and type of nearest existing closure (Figure 4.2). Distance from nearest road (strongly correlated with distance from nearest village and distance from nearest neighbouring closure) was selected to assign closure types to planning units for two reasons. First, Maxent modelling showed this was the most important predictor of the locations of each of the three types of closures within fishing grounds (Table 4.2). Second, based on the different distributions of distances in Figure 4.2a, distance to nearest road was also a strong predictor of closure type.

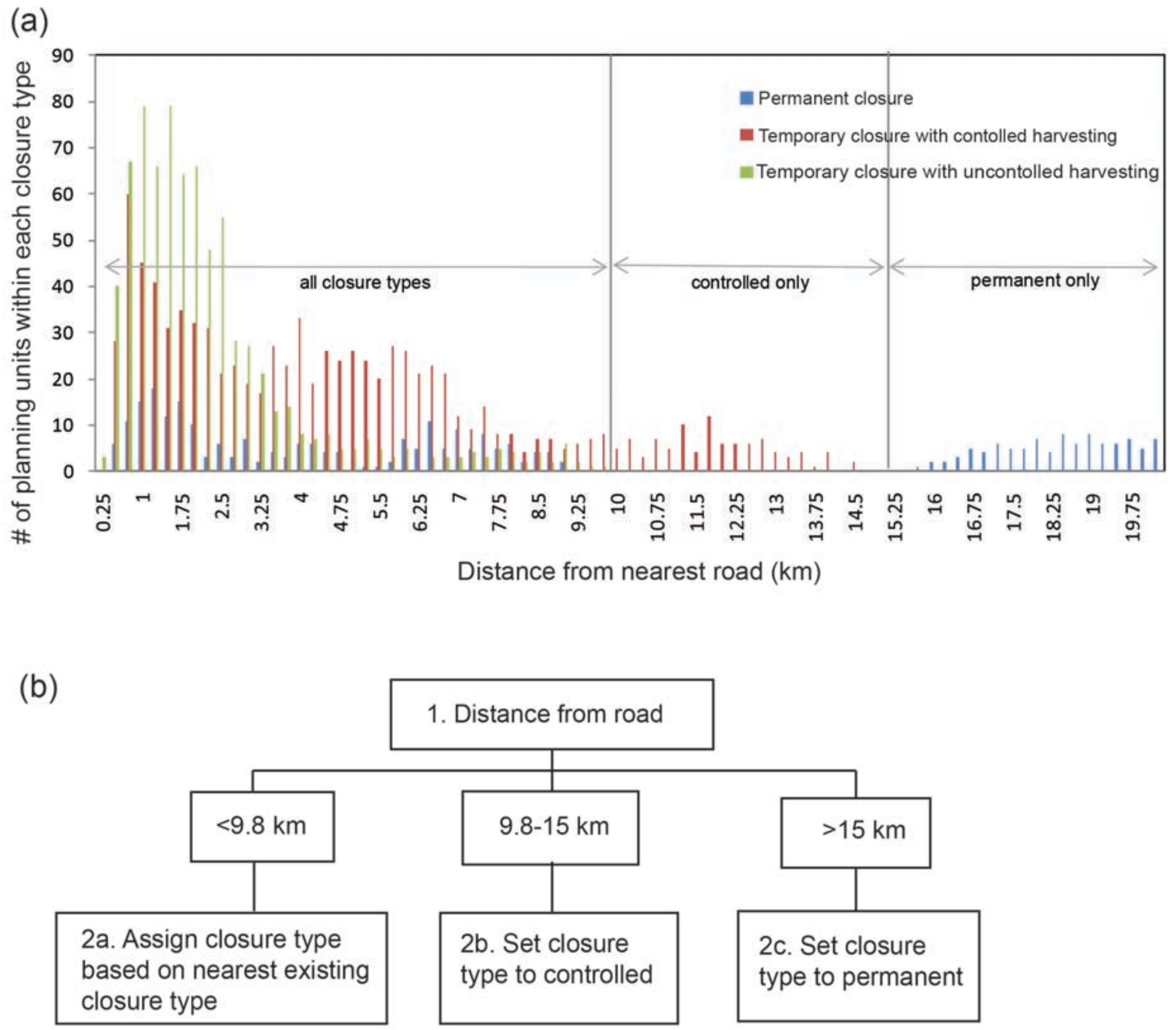

Figure 4.2. Types of closures in relation to distance from nearest road. (a) Observed distances of three closure types from the nearest road. (b) Rules for assigning closure types in the opportunistic simulation. Permanent and temporary closures with controlled harvesting were assigned according to distance from nearest road. If the distance from nearest road was less than 9.8 km, then the closure was assigned to the type of the nearest existing closure.

Table 4.2. The estimated percentage contributions of each predictor to the Maxent models for permanent, temporary closures with controlled and uncontrolled harvesting and all closures combined.

	Permanent closure	Closure with controlled harvesting	Closure with uncontrolled harvesting	All closures
Presence of provincial resource management support team [a]	0.1	28.3	3	0.3
Proportion of inshore fishing ground closed[a]	72.6	27.8	28.2	48.8
Distance from nearest road, village and closure [b, c]	24.2	34.4	66.5	40.4
Ecosystem type [b]	3.1	9.6	2.3	10.5

[a] Predictors between fishing grounds

[b] Predictors within fishing grounds

[c] Contribution of predictors cannot be separated because they are significantly correlated

4.3.5. Systematic scenario

I used Marxan with Zones for a spatial prioritisation that identified closure configurations to achieve conservation objectives. Because part of the software's selection method is stochastic, I ran the selection process 100 times. I minimized impacts of closures on villages by preferentially selecting planning units most suitable (from Maxent) for each closure type, with more suitable planning units having lower cost. Marxan with Zones selected planning units for different forms of management (zones) based on their relative costs and contributions to conservation objectives (Watts et al. 2009). I included existing closures in all configurations and counted their contributions to objectives. Planning units outside existing closures could be assigned to one of three zones: permanent, controlled, or no management. Uncontrolled closures were not considered because accrued benefits can be rapidly reversed during intensive harvests (Foale and Manele 2004). I made objectives proportional across fishing grounds so, for example, 30% of the mangroves within each had to be represented. I adjusted the relative costs of the forms of management so that selected permanent and controlled closures were in the same ratio (1:4) as existing ones. I ran the analyses to maximize achievement of objectives within the constraint of adding an average of 90 km^2 of closures per year, the same as in the opportunistic scenario. Selected closures were attributed to individual years between 2011 and 2020, assuming that closures with highest suitability would be added first. After closures had been allocated to fishing grounds, to match the opportunistic scenario, all other planning units within fishing grounds containing closures were classified as 'other management'.

4.3.6. Comparing opportunistic and systematic scenarios

After the different forms of management were allocated to fishing grounds in the opportunistic and systematic scenarios, I averaged the percentage achievement of each ecosystem's objective across the 100 simulations (opportunistic) or 100 repeat runs (systematic) for each annual time step over the 10 years. I also averaged yearly achievement of objectives across the 100 replicates and across all ecosystems to give a single parameter for comparing opportunistic and systematic scenarios over time.

4.4. Results

The four Maxent models, predicting suitability for the different types of closures and all closures combined, produced good fits to the existing data (cross-validated AUC at least 0.98). The most important predictors were: distance from nearest road (correlated with distance from nearest village and distance from nearest closure); proportion of inshore fishing ground already closed; and presence of a provincial resource management support team (Figure 4.3).

Simulations showed that neither the opportunistic nor systematic scenario achieved all conservation objectives by 2020, although the systematic approach was more successful (Figure 4.4). In the opportunistic scenario, fringing and non-fringing reefs, mangroves and intertidal ecosystems missed their objectives by at least 12-17%, while objectives for ecosystems of other benthic substrata were exceeded (Figure 4.4a,c). In the systematic scenario, non-fringing reefs and mangroves missed their objectives by 2-5%, but all other objectives were exceeded (Figure 4.4b,d). In the systematic scenario, high selection frequencies were concentrated on targeted ecosystems (Figure 4.5). Selection frequencies in the opportunistic scenario were unaffected by ecosystem type and consequently lower and more evenly spread across fishing grounds, indicating higher flexibility in the choice of planning units, but also lower efficiency in achieving all objectives within the constraint of 90 km^2 of closures per year. Opportunistically selected closures were confined to a small set (31% on average) of fishing grounds because of their higher suitability nationally. Systematically selected closures were spread across most (95% on average) of the fishing grounds.

The overall achievement of objectives by the opportunistic and systematic scenarios, averaged across 100 selection processes and across all ecosystems, was similar for the first four years (Figure 4.4e). After 2013, achievement of objectives by the systematic scenario increased much more quickly.

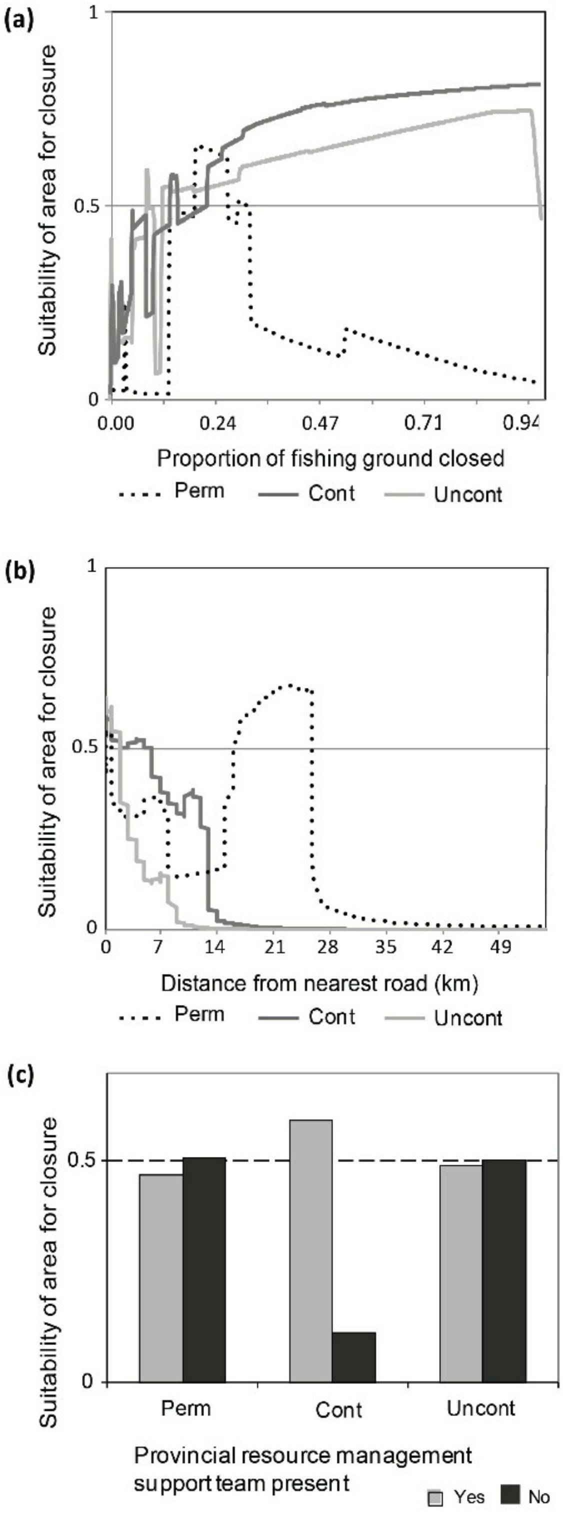

Figure 4.3. The 3 most important predictors describing the distribution of closures within fishing grounds in Fiji (see Table 4.1), based on Maxent models. The response curves show how the suitability of areas within fishing grounds for each type of closure is related to each predictor. These curves do not incorporate interactions between the predictors. Perm - permanent closures; Cont – controlled closures; Uncont – uncontrolled closures. (a) Suitability for closures in relation to the proportion of inshore fishing ground already closed. The effect of this predictor was greatest for permanent closures (contributing to 73% of the model). (b) Suitability for closures in relation to distance from nearest road. This predictor was significantly correlated with both distance from nearest village and distance from nearest closure. Attribution related to the three predictors has been combined, so single predictors should not be interpreted in isolation. The effect of distance from nearest road was greatest for uncontrolled closures (contributing to 67% of the model) and controlled closures (contributing to 34% of the model). (c) Suitability for closures in relation to the presence of a provincial resource management support team. The effect was greatest for controlled closures (contributing to 28% of the model). The effects on other closure types were negligible.

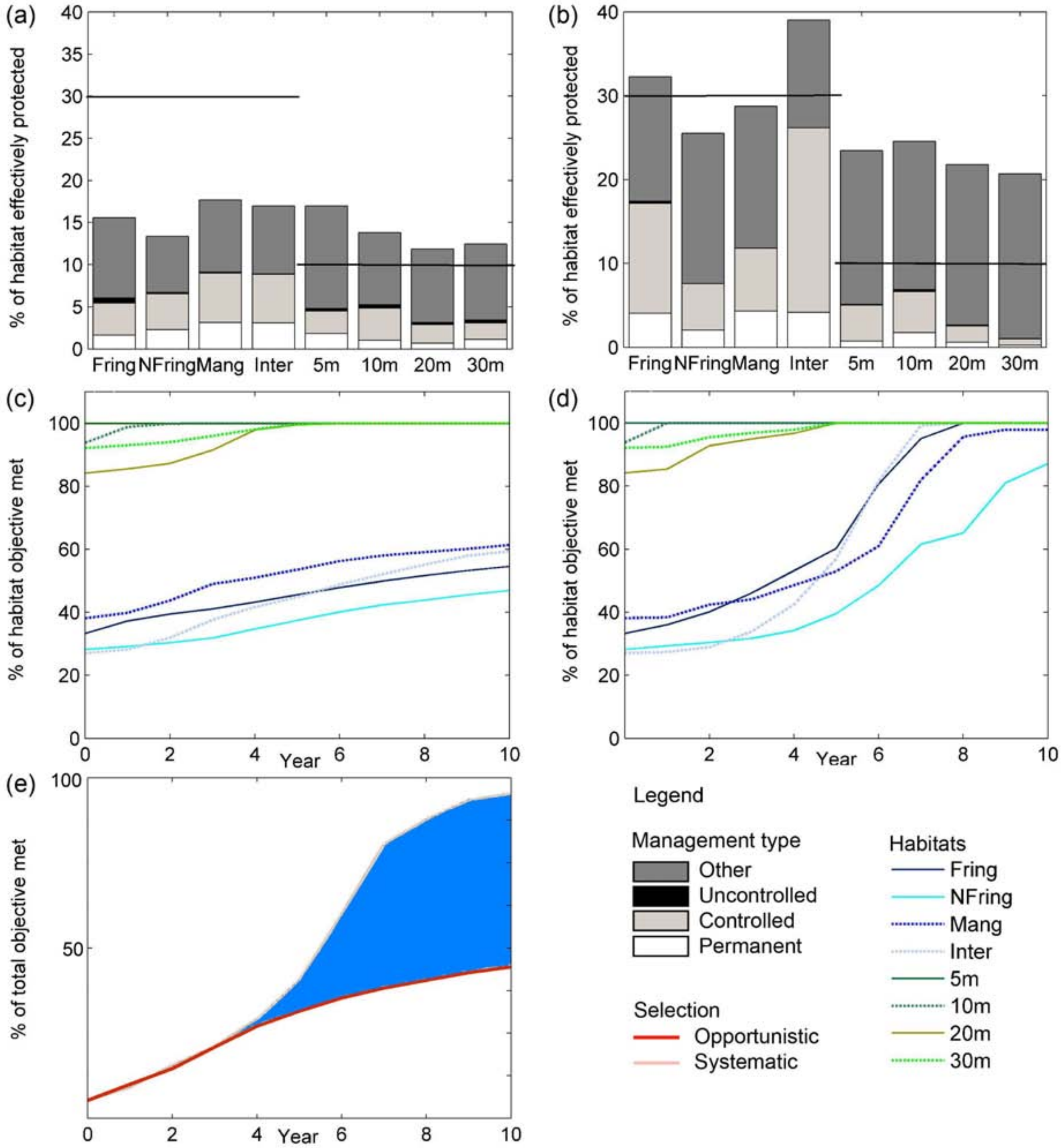

Figure 4.4. Achievement of objectives by the opportunistic and systematic scenarios. Fring - fringing reef; NFring - non-fringing reef; Mang - mangroves; Inter - intertidal; 5 m - other benthic substrata, 0-5 m depth; 10 m - other benthic substrata, 5-10 m depth; 20 m - other benthic substrata, 10-20 m depth; 30 m - other benthic substrata, 20-30 m depth. (a) Representation of ecosystems by 2020 in the

opportunistic scenario. Horizontal lines indicate conservation objectives. (b) Representation of ecosystems by 2020 in the systematic scenario. Horizontal lines indicate conservation objectives. In all parts of this figure, the percentage of ecosystem effectively protected considered the relative per-unit-area contribution of each management type: 100% contribution by permanent closures; 70% contribution by temporary closures with controlled harvesting; 10% contribution by temporary closures with uncontrolled harvesting; and 20% contribution by other management. The effectiveness of different forms of management in protecting each ecosystem was based on Chapter 3. (c) For the opportunistic scenario, the increase in representation of each ecosystem over the ten years to 2020, averaged across 100 simulations. (d) For the systematic scenario, the increase in representation of each ecosystem over the ten years to 2020, averaged across 100 runs. (e) Achievement of objectives over the ten years to 2020 in the opportunistic and systematic scenarios, averaged across 100 selection processes and across all ecosystems. The area shaded in blue bounds the potential achievement of objectives. Intermediate achievements are likely with scaling down of the systematic design, perhaps compromising achievement of some objectives, and with scaling up opportunistic action by coordination between villages and fishing grounds.

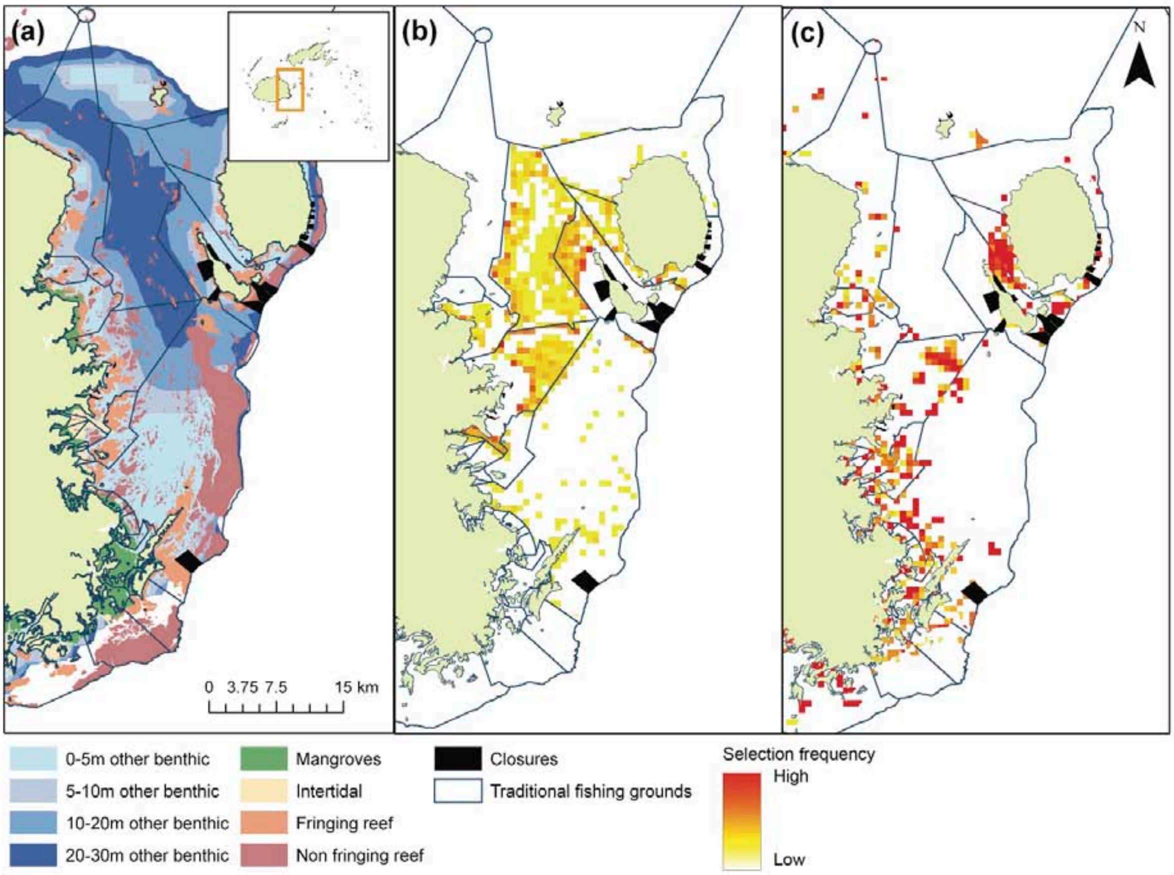

Figure 4.5. Ecosystems and selection frequencies of areas in fishing grounds on the east coast of Viti Levu. Inset shows the Fiji Islands, with the main figure focused on inshore waters in portions of Tailevu, Lomaiviti and Rewa Provinces. (a) The eight ecosystem types with spatial information nationally. (b) Selection frequencies of areas for the opportunistic scenario, measured across 100 simulations. (c) Selection frequencies of areas for the systematic scenario, measured across 100 runs. Areas selected multiple times across the different scenarios are represented by warmer colours. In (c), areas with high selection frequencies are those with few spatial options to achieve conservation objectives.

4.5. Discussion

This study aimed to inform decision-makers about potential future outcomes of different approaches to locating conservation actions. In this way, I hope to encourage strategic thinking about local conservation investments. For Fiji, this study is embedded in the policy process surrounding the expansion of MPAs. For other regions, this study could be adapted easily, recognizing the need for context-specific models of suitability and simulation rules for opportunistic decisions about conservation management.

This simulations defined the likely upper and lower bounds of plausible futures given different decisions about conservation investments. I found that, in Fiji, given constraints on the annual expansion of closures, a systematic approach, if it could be implemented, would lead to better achievement of national conservation objectives than an opportunistic approach. This was not surprising given that the systematic approach is specifically designed to achieve these objectives. However, although this cost layer led to preferential selection of more suitable areas in the systematic scenario, areas classified as ‘unsuitable’ were still available for selection and some were included in the MPA network. Realistically, though, not all areas will be available for conservation. This systematic analysis also assumed the data available were accurate on the ground, although all regional data are likely to have errors that can only be corrected with local insight. As with any systematic plan, scaling down will require adjustments of design to accommodate availability of areas and errors in data, probably losing of some of the theoretical marginal benefit of spatial prioritisations (Figure 4.2e).

Systematic plans that have been effectively scaled down and implemented have relied on a strong government leadership and incorporated fine-resolution information through extensive stakeholder engagement (e.g., McCook et al. 2010). Given limited government resources in Fiji (Lane 2008), much of the planning for management is outsourced through the Fiji LMMA network which encourages opportunistic actions, and is now moving to coordinate these opportunistic actions (WRI 2008). In this context, and with limited resources for conservation and little enforcement by the national government, scaling up opportunistic actions will probably be more effective in Fiji than scaling down a systematic national plan. My simulations point the way to identifying approaches to coordination that make opportunistic, community-based MPAs more strategic and effective. The Fiji Government and the Fiji LMMA network have already indicated a willingness to consider the results of this study in future conservation decisions (S. Jupiter, personal communication).

I illustrated the feasibility of modelling the suitability of areas for different kinds of closures as a proxy for likelihood of implementation or conservation opportunity. Data on opportunities inform managers about where to work and what tools to use (Knight et al. 2010). Opportunities for applying different forms of management will depend on the social context, including communities' willingness to engage in conservation. Previous studies of opportunity have collected and mapped data on characteristics of private landholders or communities and related these data to willingness to engage. Examples are community preferences and values (Game et al. 2011), local knowledge on biodiversity attributes (Seddon et al. 2010), knowledge of conservation (Knight et al. 2010), and the ability to adapt to environmental change (Sexton et al. 2010). Previous mapping of opportunities for management has been limited to small study areas (e.g., 146 660 hectares; Knight et al. 2010) because collection of data on opportunities is demanding of time and other resources. In comparison to biological data, the funding allocated to the collection of social data is generally small. Ideally opportunity would be predicted locally with remotely collected data. This has been trialled by Guerrero et al. (2010), who found that characteristics related to willingness to sell could be predicted using census data. Data on the human and social characteristics influencing opportunity for conservation were not available nationally in Fiji to test this approach. Instead, I used Maxent to model suitability for closures based on the spatial relationships between existing closures and predictors, assuming that suitability was correlated with opportunity. Models such as this, covering multiple forms of management, could inform spatial prioritizations across local or larger extents.

While this study provides a broad perspective on conservation opportunity in Fiji, numerous interconnected social, economic and political factors will also influence whether villages within a fishing ground will undertake conservation (Ostrom 2007; Weible and Sabatier 2007). These factors operate locally and are seldom depicted spatially. My suitability models, though supported by the resource management literature for the Pacific, are therefore only a starting point for discussion with communities about the potential reductions in conservation achievements with an uncoordinated approach to establishing conservation actions. Other factors which will influence the spatial distribution of opportunity include policies, markets, characteristics of resource users, incentives, cultural values, and governance (Berkes 2007; Ostrom 2007). Incorporating insights into these factors will be critical to guiding initiatives to scale up local conservation action.

I have concentrated this study on representation objectives but, in reality, planning processes involve multiple social and ecological objectives that vary between local and wider contexts. A more complete understanding of the benefits of systematic over opportunistic approaches will come from consideration of multiple objectives, the trade-offs between them given constraints on conservation resources, and the respective likelihoods of implementing management designed systematically and opportunistically.

Chapter 5

First steps to planning with opportunity: Defining conservation opportunity in a common pool marine resource governance system[4]

[4] Mills M., Pressey R.L., Foale S., Knight A.T., Ban N. and Aswani S. Defining Conservation opportunity in a common pool marine resource governance system. Conservation Letters. In review.

5.1. Abstract

Effective conservation requires people to make choices about how they interact with the environment to ensure its sustainability. Although it is commonly acknowledged that human and social characteristics influence the likelihood of establishing long-term conservation actions with strong compliance (hereafter conservation opportunity), these characteristics are rarely considered in systematic planning. Ours is the first study to: explicitly test the human and social characteristics of conservation opportunity in a planning region that influence the presence and form of management; and, to compare the performance of data at different resolutions to inform conservation opportunity. My method uses the social-ecological systems framework (Ostrom 2007) and literature on management in Melanesia to create a Melanesia-specific social-ecological systems framework to identify opportunity for the presence and different forms of management. I then apply this Melanesia-specific socio-ecological systems framework to test for associations between the presence and form of management on one hand and human and social characteristics on the other, using data collected at different resolutions. For Melanesia, I found that characteristics of the governance system, users and the social, economic and political setting influenced conservation opportunity. Data at both household and village resolutions characterized villages by similar human and social characteristics, with villages having more that a single management regime being more similar to each other than to villages without management. The human and social characteristics identified from household interviews accounted for over double the variation in the form and presence of management compared to data at the resolution of villages. Household data are therefore needed to predict conservation opportunity effectively. I propose that the socio-ecological systems framework be used to guide systematic planning and that future planning processes adapt my methods to gain insight into conservation opportunity and the types of data required to predict it.

5.2. Introduction

Although it is commonly acknowledged that human and social characteristics influence the likelihood of establishing long-term conservation actions (e.g. resource management) with strong compliance (hereafter 'conservation opportunities') (Mascia 2003), these characteristics are rarely considered in systematic planning (Cowling et al. 2004; Polasky 2008). The spatial prioritisation component of systematic planning guides spatial and temporal decisions about conservation actions that achieve conservation goals cost-effectively (Pressey and Bottrill 2009). To date, social context has mostly been incorporated into spatial prioritisations as threats or costs. For threats, areas at risk of degradation or the most "pristine" areas are prioritized, depending on the approach. When costs are incorporated, areas that provide the largest biodiversity benefits for the minimum cost (e.g., acquisition, opportunity or management costs) are favoured (Ando et al. 1998; Ban and Klein 2009; Margules and Pressey 2000; Wilson et al. 2007).

A more nuanced approach to planning with social characteristics can also identify areas with conservation opportunities, potentially reducing misspending on areas where human and social characteristics will inhibit effective conservation action, for example through low capacity or willingness to implement conservation, or high levels of corruption (Game et al. 2011; Knight and Cowling 2007; Knight et al. 2010). Information on conservation opportunity could be used in parallel with data on cost and threat, identifying areas where management can be implemented and sustained, as well as being cost-effective and intervening effectively in mitigating threats. In this study, I borrow from the social science literature a diagnostic framework for identifying characteristics of effective resource governance within social-ecological systems. I use this framework to inform a new method of identifying regionally-specific human and social characteristics of conservation opportunity. I apply this method to a case study in the Solomon Islands, and assess the resolution of data necessary to understand conservation opportunity in that country. I suggest future planning processes could adapt the method I develop, making studies on conservation opportunities comparable so it is possible to learn what drives conservation opportunity in different socio-political contexts and what resolution of data is needed in different social and ecological systems.

Conservation opportunities can be included in planning in several ways: 1) by involving stakeholders throughout the planning process, encouraging them to highlight opportunities for or constraints on conservation actions (Knight et al. 2006; Pressey and Bottrill 2009); 2) undertaking spatial prioritisations based on information on conservation value, cost, and threat, and then ensuring institutional processes support “informed opportunism” so that strategic priorities for conservation action are balanced with consideration of opportunities (Game et al. 2011; Knight and Cowling 2007; Noss et al. 2002; Pressey and Bottrill 2008); and 3) by analysing the human and social characteristics that facilitate or inhibit implementation of conservation actions, and incorporating that information into planning (Cowling et al. 2004; Knight et al. 2006). The first of these approaches, although vital, is limited by the number of people that can be involved in a planning process. The second approach is also essential, given the unforeseen circumstances faced by planners, especially when regional-scale data fail to reveal local-scale opportunities. Consequently, the third approach of analysing the social dimensions of opportunity is both complementary to the first two and crucial for understanding the drivers of conservation opportunity, and their spatial distribution.

Several studies have attempted to map conservation opportunities to direct conservation actions across a variety of spatial extents. A global spatial prioritisation included country-specific governance indicators such as political stability, government effectiveness, and control of corruption (O'Connor et al. 2003). For individual countries, Stephenson and Mascia (2010) developed an approach to mapping social well-being to inform planners about the spatial distribution of human needs and capacity to manage natural resources. Also nationally, Sexton et al. (2010) investigated the extent to which culture can adapt to environmental change, and discussed how to integrate the results into planning. For small study areas in which actual management units are defined, the willingness-to-sell of private land owners has been mapped to prioritise conservation actions (Knight et al. 2010) and modelled spatially as an alternative to time-consuming surveys (Guerrero et al. 2010). Additionally, in Chapter 4 I mapped conservation opportunity based on

Maxent modelling. The human and social characteristics defining managers' willingness and capacity to engage in stewardship programmes have also been investigated (Guerrero et al. 2010; Knight et al. 2010). Curran et al. (2011) mapped opportunity for involving land managers in a restoration program. These studies, although few, provide a foundation for further work on mapping opportunity for conservation action. However, the local studies have independently selected human and social characteristics believed to facilitate management and have not used or proposed a common framework to structure their assessments. A common framework has two important advantages: it allows comparison of studies on opportunity and identification of common characteristics associated with forms of management across different social and ecological contexts. Consequently, much scope remains to improve current approaches to mapping conservation opportunity for conservation action, and to improve the robustness of recommendations about what factors shape conservation opportunity and what human and social data are needed to predict opportunity spatially.

Although mapping conservation opportunities is relatively new to systematic planning, a large body of literature within the social sciences has investigated conditions for effective resource governance, a precondition for conservation opportunity. Ostrom (1990) identified eight principles defining robust governance of common-pool resources, including well defined resource boundaries, and collective-choice arrangements. Agrawal (2001) identified more than 30 human and social characteristics influencing sustainability of resource use by facilitating self-organization of communities and implementation of conservation actions. This body of work, in tandem with preceding studies on human-environmental interactions (e.g., McCay 1978; Rappaport 1968), gave rise to the socio-ecological systems framework (e.g., Ostrom 2007). This framework analyses characteristics of six components: (1) the natural resource users (e.g., fishermen), (2) the governance system (e.g., property rights), (3) the resource system (e.g., coral reef ecosystem), (4) the resource units (e.g., fish), (5) the related ecosystems, and (6) the broader social, economic and political context. These components of the socio-ecological system will interact to shape outcomes that can help or inhibit effective resource governance. The studies that have contributed to this framework emphasise that human and social characteristics that influence effective resource governance are context-specific. However, the repeated application of the socio-ecological systems framework will build understanding of both common and idiosyncratic characteristics of effective resource governance in different socio-ecological systems.

The Advocacy Coalition framework provides a different perspective, stating that people's beliefs need to align with new policy for it to be effectively implemented. Furthermore, this framework states that effective implementation of new policy depends on external shocks (e.g., changes in socio-economic characteristics), accumulation of knowledge (e.g., scientific), and/or dissatisfaction with the status quo and willingness to compromise to promote change (Weible and Sabatier 2007). Both the socio-ecological systems and Coalition Advocacy framework can inform conservation science (Berkes 2007; Weible 2007)**.**

Effective systematic planning depends on ascertaining which social data are most useful for defining conservation opportunities. Here I identify human and social characteristics that influence conservation opportunity for resources embedded in common property systems in the Solomon Islands. The methods I develop are applicable to understanding conservation opportunity in common property systems in other terrestrial and marine regions. I use Ostrom's (2007) social-ecological systems framework to organise the human and social characteristics that influence conservation opportunities in Melanesia, and assess whether these characteristics mirror Ostrom's principles for effective governance (Ostrom 1990). I also determine which human and social characteristics of villages are associated with different arrangements for management of marine resources, and compare analyses of social data from a national village survey to those from household interviews. Two research questions underpin this study: (1) What human and social characteristics explain conservation opportunities? And, (2) at what resolution are data needed to define conservation opportunities?

5.3. Methods

5.3.1. Study region

The Solomon Islands, located south-east of Papua New Guinea, comprises 6 main islands and hundreds of adjacent smaller islands. Most islands are surrounded by steeply sloping fringing coral reefs, with globally significant marine biodiversity (Green et al. 2006). The population of the Solomon Islands was estimated to be 515,870 at the last national census in November 2009. Like neighbouring Papua New Guinea, it is highly fragmented politically and diverse culturally and linguistically (Tryon and Hackman 1983). The population has a very high dependence on subsistence farming, primarily slash and burn, and fishing. While the population density is low (about 18 people/km^2) compared with Southeast Asia, growth is rapid (2.3% per year). This growth, coupled with expanding domestic and export markets for fish and marine invertebrates and extensive industrial logging, has increased sedimentation, nutrient runoff and fishing pressure (Albert et al. 2008), all of which threaten marine biodiversity. Solomon Islands is part of Melanesia, a region within the Pacific Ocean known for complex arrangements for customary ownership of marine resources (Hviding 1998).

Customary law is the primary institution regulating management of marine and terrestrial resources in the Solomon Islands (Hviding 1998). Conservation actions implemented by villages include different forms of management. For example, permanent closures are areas where resource extraction is prohibited. Temporary closures are areas where harvesting is allowed temporarily (e.g., for feasts). Some villages also implement quotas and restrictions on species and gear. The national government is thought to have insufficient expertise or resources to meet the challenges of management (Lane 2008).

This study covers two spatial resolutions. First, I used data from government surveys of villages from around the country. I also collected new data for individual households in villages within the Roviana, Vonavona, and Morovo Lagoons in the Western Province. I expected household data to reveal variation in human and social characteristics within villages and, because of the survey design for households, to provide more detailed information on the drivers and challenges of management. Of this second group of villages, those that have implemented management are part of a Resource Management Program established by one of the authors (S Aswani) in 1999. As of 2011, this program has helped set up 32 permanent and temporary closures. The objectives of these closures are to: (1) improve local fisheries; (2) protect spawning and nursery areas; (3) protect vulnerable species and habitats; (4) reinforce customary sea tenure; and (5) build indigenous ecological knowledge (Aswani et al. 2007).

5.3.2. Defining conservation opportunity

I reviewed the scientific and "gray" literature on resource governance in Melanesia to identify human and social characteristics that potentially define conservation opportunities in the Solomon Islands. I used the ISI Web of Science and Google Scholar search engines to find literature and sought additional studies by scanning reference lists and contacting academics working in the region. Using the socio-ecological systems framework, I identified whether human and social characteristics identified in the literature aligned with one or more of three components: 1. governance system, 2. users, or 3. social, economic and political setting. I also noted if the human and social characteristics overlapped with Ostrom's (1990) principles for effective governance. I assumed that the resource system, resource units, and related ecosystems were equivalent for all villages, because most villages for which data were available were coastal with fringing coral reefs.

I used data from a village resource survey conducted by the Solomon Islands' National Statistics Office (hereafter 'national village data') between 2007-2008 to assess whether data at the resolution of whole villages explained implementation opportunity. This survey involved one interview per village (details at http://www.spc.int/prism/country/sb/stats/). I selected human and social characteristics from the village data that matched those identified from the literature review. Data for only five human and social characteristics were both relevant and complete for most (1269) villages. These were: (1) interest in establishing locally managed marine areas (to gauge interest in management, generally following perceptions of resource decline); (2) frequency of village meetings; (3) collective efforts to clean the surrounding environment or rebuild village infrastructure (both surrogates for social capital); (4) perceived status of the fishery (associated with incentives for participating in management); and (5) existing land-sea ownership conflicts (as a surrogate for definition of resource ownership).

I used canonical correspondence analysis (CCA) to examine the relationship between the presence and form of management (including permanent closures, temporary closures, gear restrictions, species restrictions, and no management) and the selected human and social characteristics. CCA is an ordination technique that allows for direct comparison of two data matrices, in this case management forms and human and social characteristics, and can use categorical, ordinal and continuous data simultaneously (Legendre and Legendre 1998).

I also assessed whether data from household interviews could be used to explain conservation opportunities. I undertook 140 randomly selected household interviews in 10 villages using closed- and open-ended questions, investigating whether the human and social characteristics identified from the literature review to define conservation opportunities were associated with the presence and form of management (survey provided in Appendix 8.5). I sought to interview the head of the household, usually male, and, if unavailable, the next in authority. Questions also aimed to determine households' dependence on fishing for income, consumption patterns, distance travelled to fishing grounds, support for closures, and household priorities for village management (e.g., school, resource management, church). Interviews took between 30 minutes to 1 hour. Villages varied from 10 to 300 households. Between 6 and 90 percent of households were surveyed per village, with high percentages interviewed in villages with 10-20 households and low percentages in villages with hundreds of households.

I undertook CCAs on different combinations of variables from the surveys to reduce the number of human and social characteristics to those clearly relevant to conservation opportunity. This gave us a final list of 5 variables from the national survey and 14 variables from the household surveys.

5.4. Results

5.4.1. Socio-ecological systems framework for Melanesia

Based on the literature review, I identified 11 human and social characteristics potentially related to effective management in Melanesia. I associated these human and social characteristics with Ostrom's (2007) governance system, resource users, and social, economic and political settings (Figure 5.1). Five of the 11 human and social characteristics overlapped with 5 of Ostrom's (1990) 8 principles for effective governance (Table 5.1). These characteristics, all within the governance system (Figure 5.1), were: 1) clearly defined resource use rights, which help to reduce conflicts over ownership and facilitate management arrangements (Foale and Macintyre 2000; Macintyre and Foale 2007); 2) designing and explaining management while incorporating local and/or traditional knowledge and existing management systems (Govan et al. 2009; Johannes et al. 2000), thereby aligning rules and local conditions and engendering support; 3) monitoring of resources and resource users, important for effective governance (Aswani 2005); 4) mechanisms for resolution of conflicts over resource ownership that often arise with economic opportunities for use of resources (Foale and Macintyre 2000); and 5) linkages between the different levels of governance (nested enterprises) that integrate local management with legislation, promoting the sustainability of management (Schoeffel 1997).

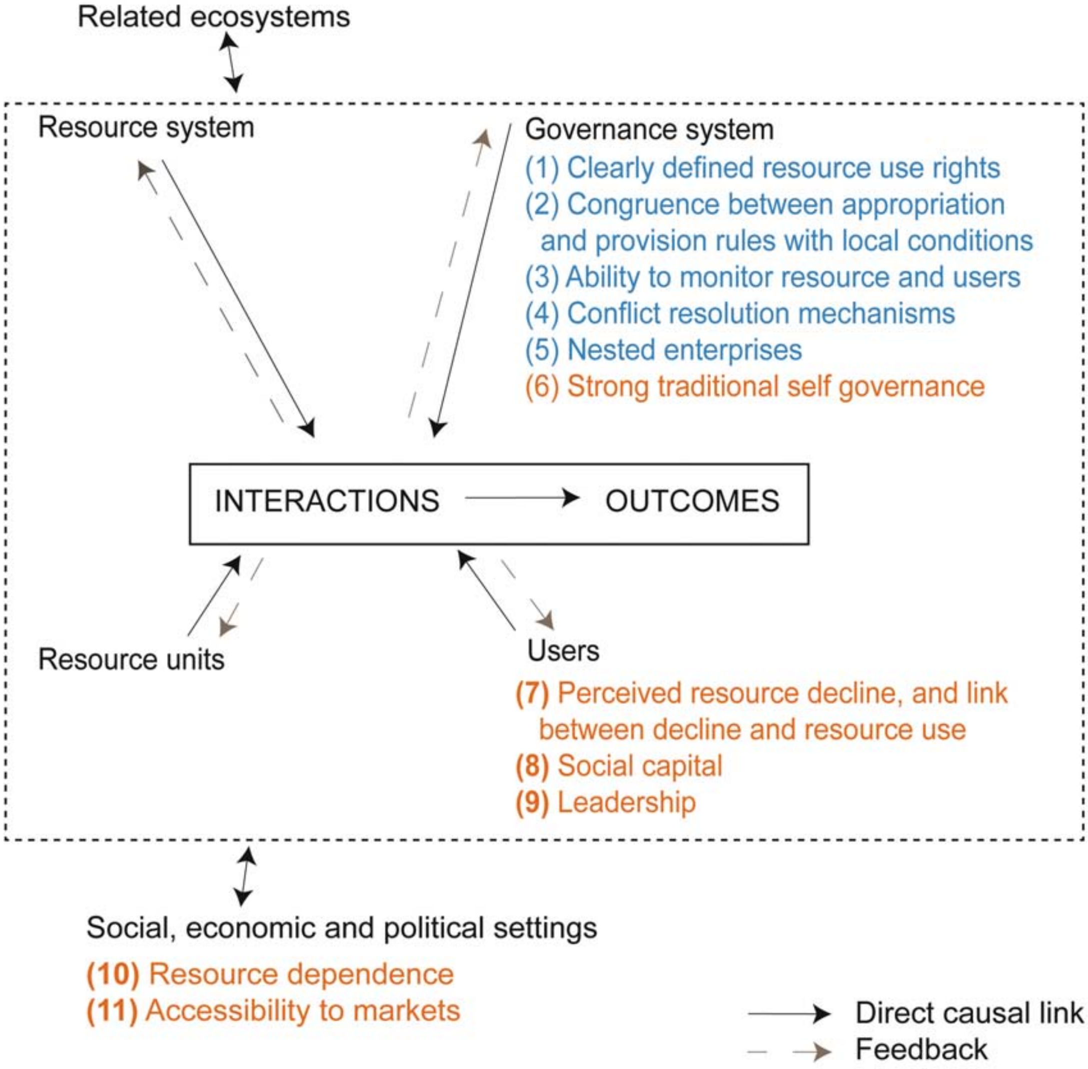

Figure 5.1. The 11 human and social characteristics organized within Ostrom's (2007) socio-ecological systems framework. Characteristics in blue overlap with Ostrom's (1990) principles for effective governance. Characteristics in orange do not coincide with Ostrom's principles.

Table 5.1. Ostrom's eight principles for effective governance (2) and literature on human and social characteristics that supports the relevance of these principles in Melanesia.

Ostrom's principles	Description	Experience in Melanesia
1	Well defined boundaries	Aswani (2005) found that clearly defined resource-use rights helped to reduce conflicts, facilitating the implementation of resource management. However, clearly defined resource-use rights are uncommon in the Solomon Islands and conflicts over ownership often emerge only when there is economic opportunity associated with resources (Cowling et al. 2004; Polasky 2008; Pressey and Bottrill 2009)
2	Congruence between appropriation and provision rules and local conditions	It has been suggested that temporary closures that allow for periodic harvest for cultural festivals have greater affinity with cultural traditions (Macintyre and Foale 2007; Foale 2008a) Explaining causes of resource decline and the benefits of resource management with due consideration of local and/or traditional knowledge was critical for the success of conservation projects undertaken in the Morovo, Roviana and Vonovana parts of New Georgia (Macintyre and Foale 2007; Foale 2008b; Laffoley 2008; Otto 1998; Babbie and Mouton 2001; Cox et al. 2010) Spiritually significant areas have been targeted for conservation. An example is Tetepare Island where there is some congruence between spiritual and conservation values. However, in Tetepare, the prohibition of resource use by The Friends of Tetepare (FOT) landowner organization was closely followed by internal conflict over resources (Aswani and Hamilton 2004a)
3	Collective-choice arrangements	No information available
4	Monitoring	A lack of support from government for resource management means that communities themselves will be responsible for enforcement of resource-use regulations. Resource use is more easily regulated by communities if the resources to be managed are adjacent to the community (Weible 2007; Foale and Manele 2004)
5	Graduated sanctions	No information available
6	Conflict-resolution mechanisms	Both the commodification of resources and resource management will often cause conflicts over ownership of resources (Aswani et al. 2007). Effective mechanisms for conflict resolution, whether formal or informal, will be required.
7	Minimal recognition of rights	No information available
8	Nested enterprises	If resource management is recognized within legislation, it is more likely to be sustained (Johannes et al. 2000)

Other human and social characteristics that appear to influence conservation opportunity did not overlap with Ostrom's principles, but did coincide with components of the socio-ecological systems framework (Figure 5.1). Under the governance system, I added 6) strong traditional self-governance, thought to influence the enforceability of management institutions (Aswani 2005). Under users, I included: 7) perceived decline in resources that motivates villagers to participate in management (Sabetian and Foale 2006); 8) social capital, facilitating the success of alternative livelihood projects to compensate for restrictions on resource use (Foale 2001); and 9) strong leadership to motivate and unite chiefs and villagers in support of management (Aswani and Hamilton 2004b; Aswani and Lauer 2006; Foale 2001; Laffoley 2008; Muehlig-Hofmann 2007) (Table 5.1). Under social, economic and political settings, I included 10) resource dependence of villages in combination with 11) the accessibility of markets. Increased pressure on resources makes the implementation of management more difficult (Van Helden 1998; Table 5.1). However, low population densities mean that pressure on resources can be low until they are commodified (e.g., beche de mer; Foale 2008b; Otto 1998; Sabetian and Foale 2006).

5.4.2. Human and social characteristics from national survey

Data at the resolution of villages from the national survey provided an overview of human and social characteristics across villages in the Solomon Islands. Of the 1269 villages surveyed, 31% had implemented some form of management including: temporary closures (24%), species restrictions (13%), quota restrictions (10%), gear restrictions (10%), and permanent closures (5%). Of the villages with management, 49% had multiple forms of management. Of all villages, 23% undertook collective efforts to clean the surrounding environment or rebuild village infrastructure, 5% had ownership disputes, and 39% were considering establishing locally managed marine areas. There was a significant difference in perceived changes in fishery conditions between villages with and without management (Pearson's chi squared test, $P < 0.000$). Those with management identified improved resource conditions more frequently (10% of villages) than those without management (1% of the villages). There was also a significant difference in frequency of village meetings between villages with and without management (Pearson’s chi squared test, P <0.000). Villages without management had fewer meetings.

5.4.3. Human and social characteristics from household interviews

Household interviews, in Western Province, formulated around the social-ecological systems framework for Melanesia, indicated that most households (45%) relied on gardening for their primary income and that fishing was the primary income for 21% of households. Distances travelled to subsistence fishing grounds were positively correlated with those travelled to commercial fishing grounds (Pearson’s correlation of 0.665, P<0.000), likely reflecting the fact that fishermen frequent the same fishing grounds and sell any additional catch. Interviewees mostly chose to use the fish they caught themselves. Less than 1% of all interviewees had sold all their catch from their last fishing trip and only 18% had sold more than half. Interviewees mostly fished close to home: 77% claimed to paddle less than 30 minutes to their favourite subsistence fishing ground and 32% paddled for less than 10 minutes. Just under half the interviewees perceived a decrease in the number (47%) and size (46%) of fish within their fishing ground (Figure 5.2a). Negative changes to fishery conditions were thought to be associated with increased fishing pressure (64%) and destructive fishing gears (22%). Those interviewees who believed there were more and larger fish (10% and 7%, respectively) attributed these changes mainly (>90%) to the presence of closures.

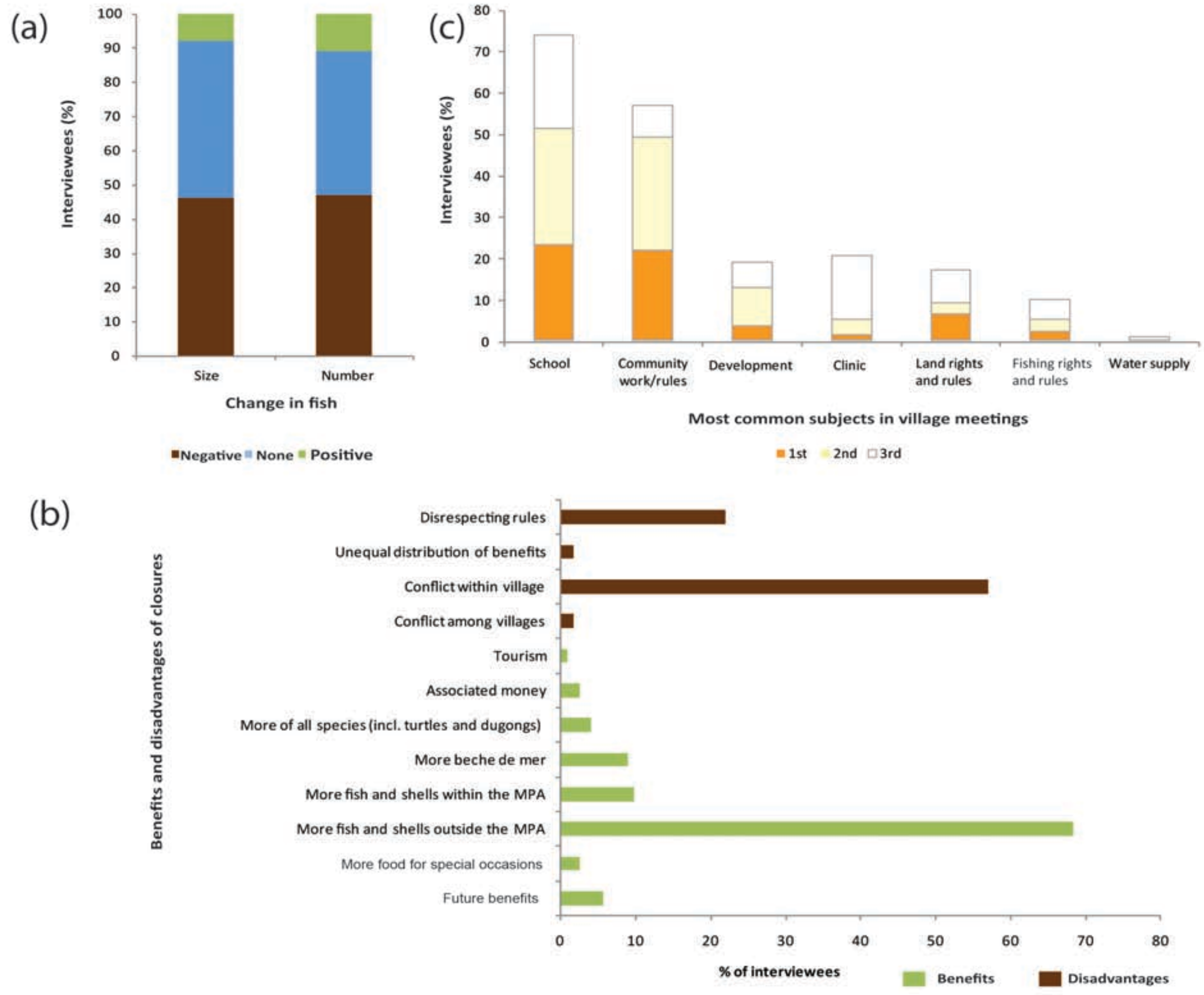

Figure 5.2. Results from household interviews. (a) Perceived changes in fish size and number. (b) Opinions about the benefits (green) and disadvantages (brown) of closures, all of which are aspects of fishing rights and rules (sixth column c). (c) Subjects discussed in village meetings.

Interviewees (n=122) identified both advantages and disadvantages of closures (Figure 5.2b). Most (89%) believed there were present benefits (83%) or would be future benefits (6%) from closures, including: more fish and/or invertebrates (inside and/or outside closures) (82%), tourism (1%), and other forms of monetary benefits (2%) such as schools or churches built from proceeds of organized periodic fishing within closures. At the same time, 50% of interviewees believed that closures had disadvantages, including conflicts associated with lack of compliance and loss of fishing grounds. Conflict was identified in all communities with management.

Most interviewees did not feel they had much role in establishing closures, with only 38% of villagers stating that the community participated in making the fishing rules. Most interviewees identified the Resource Management Committee (66%) and the chief, elders and/or pastors (74%) as having roles in establishing the rules. Resource Management Committees, comprised of selected individuals (usually elders) in each village, are established when management is implemented and are the points of contact for NGOs and academics working in the region. Most villagers (78%) did not believe they had much role as individuals in enforcing the rules. About 39% thought the chief, elders or pastor had a role in enforcement, and 69% thought enforcement was the role of the Resource Management Committee.

To understand village priorities, I asked interviewees to list and rank the three most frequently discussed issues in village meetings. School, church and community rules were the most topical issues (Figure 5.2c). Reefs were only identified as a frequent issue at one site (Nusa Hope), and by 7% of the interviewees.

5.4.4. Understanding conservation opportunity

With data from the national, village-resolution survey, human and social characteristics explained only 24% of the variation in the presence and form of management (Monte Carlo test of all canonical axis, F = 32.5, P = 0.002). Within this 24% variation, the *x*-axis captured 83% of the relationship between forms of management and human and social characteristics, separating the villages into two main groups; those with and without management (Figure 5.3a). Villages associated with management were characterized by higher social capital, land-sea ownership disputes, and incentives for participation in management.

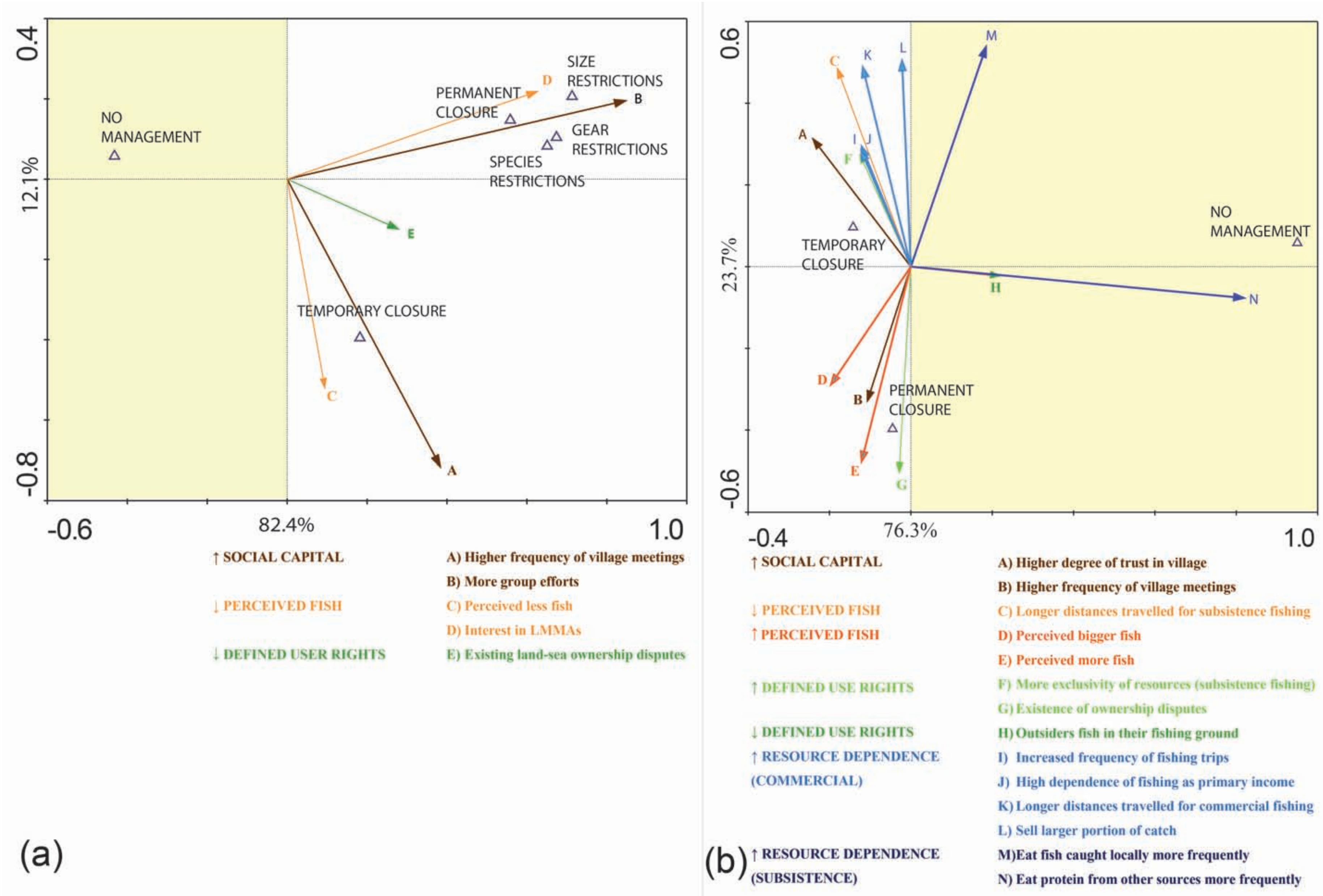
0.4
-0.8
12.1%
-0.6
82.4%
1.0
NO MANAGEMENT
PERMANENT CLOSURE
SIZE RESTRICTIONS
GEAR RESTRICTIONS
SPECIES RESTRICTIONS
TEMPORARY CLOSURE
A
B
C
D
E
↑ SOCIAL CAPITAL
A) Higher frequency of village meetings
B) More group efforts
↓ PERCEIVED FISH
C) Perceived less fish
D) Interest in LMMAs
↓ DEFINED USER RIGHTS
E) Existing land-sea ownership disputes
(a)
0.6
-0.6
23.7%
-0.4
76.3%
1.0
NO MANAGEMENT
TEMPORARY CLOSURE
PERMANENT CLOSURE
A
B
C
D
E
F
G
H
I
J
K
L
M
N
↑ SOCIAL CAPITAL
A) Higher degree of trust in village
B) Higher frequency of village meetings
↓ PERCEIVED FISH
C) Longer distances travelled for subsistence fishing
↑ PERCEIVED FISH
D) Perceived bigger fish
E) Perceived more fish
↑ DEFINED USE RIGHTS
F) More exclusivity of resources (subsistence fishing)
G) Existence of ownership disputes
↓ DEFINED USE RIGHTS
H) Outsiders fish in their fishing ground
↑ RESOURCE DEPENDENCE (COMMERCIAL)
I) Increased frequency of fishing trips
J) High dependence of fishing as primary income
K) Longer distances travelled for commercial fishing
L) Sell larger portion of catch
(b)
↑ RESOURCE DEPENDENCE (SUBSISTENCE)
M) Eat fish caught locally more frequently
N) Eat protein from other sources more frequently

Figure 5.3. Associations between human and social characteristics of villages and different forms of management of marine resources. (A) Canonical correspondence analysis (CCA) biplot, using data on 1269 villages from the national village survey. (B) CCA biplot, using data from household interviews. The human and social characteristics hypothesized to facilitate resource management are overlaid as eigenvectors. Smaller angles indicate stronger correlations between forms of resource management (open triangles labelled with capitals) and human and social characteristics (arrows labelled with letters corresponding to keys below the figures, with individual characteristics in colour-coded categories). Human and social characteristics with longer arrows are more strongly correlated with the ordination axes and therefore better explain the variability found within the data.

With data from the household interviews, human and social characteristics accounted for the larger part (59%) of variation in presence and forms of management across the 10 surveyed villages (Monte Carlo test of all canonical axis: F = 5.818, P = 0.0020). Within this 59% variation, the *x*-axis captured 76% of the relationship between presence and form of management and human and social characteristics. Like CCA of national village data, the CCA of data from household interviews separated the villages into two main groups: those with and without management (Figure 5.3b). Relative to villages without management, villages with management were associated with high social capital, perceived change in fish status, high dependence on local marine resources for commerce, disputes over resource ownership, and high resource exclusivity. Villages with no management were associated with high use of local marine resources for subsistence, low resource exclusivity, and higher consumption of non-marine animal protein.

5.5. Discussion

Despite consensus on the need to integrate social data on conservation opportunities into planning (Cowling et al. 2004; Polasky 2008), few studies have attempted to do so (e.g., Guerrero et al. 2010; Knight et al. 2010). Previous studies that mapped conservation opportunity considered human and social characteristics hypothesised to promote conservation generally (Babbie and Mouton 2001). I used the social-ecological systems framework (Ostrom 2007) and literature on management of natural resources in Melanesia to create a context-specific framework. I then explicitly tested the human and social characteristics that apparently defined conservation opportunity for their influence on management, using social data collected at two resolutions: whole villages and individual households. The same approach could be taken to understand conservation opportunity in many other regions. This understanding is especially critical in developing countries where governments have limited reach and little capacity to implement management. My approach is a way of predicting conservation opportunities, providing insights into the informativeness of different data sets.

5.5.1. What human and social characteristics might explain conservation opportunities?

The social-ecological systems framework (Ostrom 2007) provided a useful guide for organising characteristics describing conservation opportunity in this study region. Several of the human and social characteristics found to influence conservation opportunity in my literature review overlapped with Ostrom's principles for robust systems for governance of common pool resources (Cox et al. 2010; Ostrom 1990). I found overlaps with five of the principles and human and social characteristics identified in the literature review (Table 5.1), providing further evidence that some of the principles are widely applicable. CCAs of data from household interviews showed that villages associated with management were less likely to have outsiders fish their waters and more likely to consider resource use to be exclusive, supporting Ostrom's well defined borders principle (Table 5.1). However, sea/river ownership disputes were also more often associated with villages with management than those without. Such disputes could motivate management that reinforces traditional claims over marine areas (e.g., Aswani and Hamilton 2004a). Alternatively, disputes could arise because of management (e.g., Foale and Macintyre 2000).

For several principles, my literature review found no supporting evidence of positive or negative influence on conservation opportunity in Melanesia. These were collective-choice agreements, graduated sanctions, and minimal recognition of rights (Table 5.1). Perhaps, in the context of Melanesia, collective choice agreements can be replaced by strong local governance, considered critical for effective conservation action in this region**.** Reviewing Ostrom's principles, Cox et al. (2010) found case studies that suggested graduated sanctions could be replaced by high social capital because high levels of cooperation would make people less likely to break rules. Previous research in Melanesia and this study showed that high social capital was more strongly associated with villages with management than those without (Aswani 2005; Foale 2001). National recognition of rights through recognised customary tenure is minimal in Melanesia (Hviding 1998) and therefore likely to have limited influence on conservation opportunity locally. Data on the recognition of rights and strong local governance were not available for this study and therefore not tested using my survey data.

The literature review and my results also revealed that user characteristics and external characteristics influenced effective resource governance. The user characteristics that influenced conservation opportunities (e.g., perception of declining resources) aligned with the Advocacy Coalition framework. This framework states that people's beliefs, sometimes driven by their socioeconomic conditions, motivate or prevent changes in policy, and that accumulation of knowledge can lead to new policy (e.g., Weible 2007). Through the CCA of national village data, I found a higher perceived resource decline in villages with management than those without (Aswani and Hamilton 2004b; Aswani and Lauer 2006; Foale and Manele 2004; Hviding 2006). The CCA of data from household interviews, on the other hand, associated both more and bigger fish with villages with management, but also fishermen travelling longer distances to catch their fish, potentially a sign of resource depletion. These apparently contradictory findings are likely a result of the time lag of approximately 10 years between establishing management and my surveys. Perhaps the fishery is recovering in waters with management but not yet enough to justify a return to fishing adjacent to villages.

External characteristics are those within the social, economic and political settings component of the social-ecological system. According to my literature review, the external characteristics that influence conservation opportunities in Melanesia, include resource dependence and market integration. Market integration has also been found to influence conservation opportunity in Honduras and Indonesia (Cinner and McClanahan 2006; McClanahan et al. 2006; Tucker 1999; Van Helden 1998). In line with this literature, analysis of my data from household interviews showed that increased dependence on resources for commercial purposes was associated with villages with management. Villages exploiting resources for commercial purposes are most likely to overfish (Foale 2008b; Otto 1998; Sabetian and Foale 2006), and resource decline will presumably strengthen the incentive to manage. I speculate that this implies fishermen are risk-averse with respect to income when considering management. Unless there is a strong motivation for management, fishermen are unlikely to get involved.

5.5.2. Which types of data most effectively define conservation opportunities?

I found that both national village data and the household interviews explained opportunities, but to different extents. Data collected during household interviews provided a more thorough understanding of conservation opportunities. Human and social characteristics from the national village data explained only 24% of the variation in the form and presence of management in Solomon Islands. The limited performance of the national dataset is likely due to the questions not being formulated specifically to obtain information on conservation opportunities. As well, the national data lack insights into communities because only the heads of villages were interviewed, thereby missing substantial variation in perceptions of villagers between households. In contrast, the household interviews explained a large proportion (59%) of the variation in forms and presence of management observed for villages. The better performance of these data is likely due to their specific focus on conservation opportunities and their ability to capture variation in perceptions and priorities within villages. Regarding the latter point, although village chiefs have the final say about establishment and enforcement of management, they are unlikely to act without the support of their communities. The diversity of responses within villages revealed by the household surveys therefore better relates human and social characteristics with management.

Direct interviews at the resolution of households were most informative, but also most time-consuming, in my study. This study suggests a trade-off between the accuracy with which conservation opportunity can be predicted and the extent of the planning region, given the positive correlation between the extent of planning regions and their human populations. Consequently, it is critical to identify the data resolution that provides the greatest benefit for decision-making. Stephanson and Mascia (2010) suggested that the required resolution of social data depends on the goals of the spatial prioritisation. Although national data might be sufficient to inform prioritisations among countries, ecoregional planning requires data for districts or subdistricts, and planning for management on the ground requires data for subdistricts or communities (Stephanson and Mascia 2010). More recent studies have found higher-resolution are needed to depict conservation opportunity accurately. For example, in private land in South Africa, the Knight et al. (2010) showed that conservation opportunity varied significantly between land managers across landscapes. In this study, national village data explained conservation opportunities poorly, confirming earlier findings by Knight et al. (2010). Using inaccurate representations of conservation opportunity into spatial prioritisations could do more harm by shifting conservation priorities more frequently towards areas with little promise for implementation.

Additionally, this study suggests that predicting conservation opportunity for individual forms of management will require a more detailed understanding of the human and social characteristics driving conservation opportunity than is needed for a general understanding of conservation opportunity. Both the national village data and household data showed stronger differences between villages with and without management (*x*-axes capturing 82% and 76% of the relationships between characteristics and the presence of management) than between villages with different forms of management (*y*-axes capturing 12% and 24% of the relationship between characteristics and the different forms of management). This suggests that more data are needed to accurately predict conservation opportunity for different forms of management. My results contrast with those of Knight et al. (2010) who suggested that household interviews were suitable to identify human and social characteristics associated to conservation opportunities for different forms of management. This difference can be attributed to differences between communal and private land tenure, and/or marine and terrestrial systems. When resources are governed communally, as in the Solomon Islands, a more detailed understanding of incentives and disincentives is required to spatially allocate different forms of management effectively.

Because of the costs of obtaining data from direct interviews, approaches to modelling conservation opportunity across regional extents offer a way forward. Guerrero et al. (2010) census data to model willingness to sell among private landholders, an indicator of conservation opportunity in South Africa, and suggested relevance to planning across regions. In contrast, I found that national census data for villages explained only 24% of the variation in forms of management employed in

Solomon Islands, thereby predicting conservation opportunity poorly. Potential explanations for this poor performance are: (1) there was not enough resolution in existing census-type data to reveal variation within villages of the human and social characteristics related to management; and (2) the questions were not focused sufficiently on drivers of conservation opportunity. Modelling conservation opportunity based on human and social characteristics from household data would be more accurate, but would require a significant investment of time and resources by the organisation leading the planning process.

5.5.3. Limitations

The main limitation of this study is that the data provide only a snapshot in time. Conservation opportunities are not static as depicted here, but will change in response to regional and local drivers such as changes in market conditions and local leadership. All villages in which I surveyed households had management established for at least 10 years, while the duration of management by villages in the national survey is unknown. Consequently, the human and social characteristics analysed reflect the current status of the villages, not those when management were first established. This makes it difficult to determine whether the characteristics are preconditions for conservation opportunities or consequences of management.

5.5.4. Conclusion

Indentifying conservation opportunities across a planning region is the first step to reducing misspending of conservation funds on areas where effective conservation action is unlikely. I identified human and social characteristics that were associated with the presence and form of management and therefore either indicate conservation opportunity or the consequences of management. There will be a trade-off in investing in higher-resolution and more intricate social data for a better understanding of opportunity or spending those funds on attempting to implement conservation actions. Investing in social data to predict opportunities should always be complementary to involving stakeholders in the planning process and evaluating, after initial spatial assessments, alternative local opportunities for conservation action through informed opportunism.

Chapter 6

General Discussion

6.1. Thesis summary

To halt the decline and degradation of marine resources, implementation of additional MPA networks is recommended (Pauly et al. 2002; Sala and Knowlton 2006). Systematic planning is an effective approach to design MPAs, and some results of the implementation of large-scale MPAs designed using this approach are promising (e.g., McCook et al. 2010). However, numerous factors impede the translation of systematic plans into management, including (as outlined in Chapter 1): (1) the scale mismatch between systematic planning and conservation actions (Briggs 2001); (2) limited collaboration between planners and resource managers (Knight et al. 2008); (3) lack of comprehensive social assessments (Cowling and Wilhelm-Rechmann 2007; Polasky 2008); (4) lack of institutions supporting a dynamic systematic planning processes (Grantham et al. 2010; Pressey and Bottrill 2009), and; (5) lack of understanding of the optimal suite of conservation actions and incentives to achieve conservation goals (Ferrier and Wintle 2009; Knight et al. 2007). This knowing-doing gap has led to discussion about the best investment of conservation resources between systematic planning and opportunism (Knight and Cowling 2007; Pressey and Bottrill 2008), and between conservation driven by local and regional perspectives (Noss 2010; Smith et al. 2009).

Although it can be useful to polarize these perspectives to better understand their respective strengths and limitations, many academics are exploring how they can complement one another (e.g., Game et al. 2011; Noss et al. 2002; Seddon et al. 2010). For example, local and opportunistic conservation initiatives are now being scaled up so they achieve national conservation as well as local fishery goals (e.g., Eisma-Osorio et al. 2009). Similarly, systematic plans are increasingly incorporating factors associated with opportunistic decisions (e.g., local knowledge, high levels of stakeholder participation) to ensure that the plans are useful to those working on-ground and result in actual management (e.g., Ban et al. 2009a; Pressey and Bottrill 2009).

The goal of this study was to add to the body of knowledge that explores options for integrating systematic planning with local management. I addressed this goal by examining the knowing-doing gap from both a regional (e.g., Chapter 2) and a local perspective (e.g., Chapters 3, 4 and 5). I investigated how to scale down systematic plans to increase the likelihood of implementation, and to scale up MPAs to form MPA networks. I advanced knowledge on four of the five causes of the knowing-doing gap described in Chapter 1. Below, I describe how each of the chapters contributed to the thesis objectives, and how I addressed the challenges of the knowing-doing gap.

6.1.1. Objective 1. Investigate methods for scaling down systematic plans to inform conservation action, focusing on opportunities for implementing multiple forms of management and their contribution to conservation goals.

In this thesis I contributed to knowledge on scaling down systematic plans to conservation actions in three main ways. First, I investigated the scale mismatch between systematic planning and conservation actions. This provided academics and planners with a broad overview of the progression from larger to smaller extents and resolutions (e.g., from the identification of broadly defined priority areas to identification of areas for management) when doing systematic planning. Second, I developed methods to understand and map conservation opportunity, so that it could be better incorporated into spatial prioritisations. Third, I developed a method to undertake gap analyses that incorporates the relative contributions of different forms of management encountered within a planning region, using data typically available for gap analyses. Below I expand on each of these contributions, and associate them to aspects of the knowing-doing gap that I addressed.

Addressing the scale mismatch between systematic planning and conservation actions

The framework I developed in Chapter 2 contributes to understanding the key decisions about spatial scale in systematic planning and their impacts, thus allowing planners to work within current limitations of spatial scale (e.g., political boundaries), while finding methods to overcome those limitations (e.g. by gathering information on local costs and values). A regional perspective allows planners to explore the spatial and temporal options for management, and identify spatial solutions to resource degradation that are more likely to be effective and politically and ecologically sustainable (Cash et al. 2006). However, decisions made from a regional perspective will often conflict with local priorities and values (Smith et al. 2009). The framework developed in Chapter 2 is the first to explore the key decisions about spatial scale in systematic planning and investigate the considerations that influence these key decisions (see Figure 2.1). Among these considerations are limitations of data, resources and information on opportunities and constraints. When making each decision about spatial scale, there will be trade-offs between different considerations (e.g., the extent of management can be either ecologically optimal or socially feasible), with implications for the ease of using the products from spatial prioritisations to inform management (see Table 2.2 for a summary of trade-offs).

Contribution to knowledge on comprehensive social assessments

Social, political and economic factors influence the likelihood of conservation actions being implemented and supported in any planning region. Systematic planning should therefore include comprehensive social assessments (Polasky 2008; Wilhelm-Rechmann and Cowling 2011). An important part of these social assessments is the identification of conservation opportunities, to provide insight on where management is likely to be most effective (Knight et al. 2010). However, there is very little existing research providing guidance on how to map conservation opportunity (e.g., Guerrero et al. 2010; Knight et al. 2010; Sexton et al. 2010). In this thesis, I advanced existing knowledge of conservation opportunity in two ways.

First, in Chapter 4, I used maximum entropy methods to model conservation opportunity for different forms of management at a regional scale. The few studies that have mapped conservation opportunity at regional scales used data at the resolutions of states or districts (Sexton et al. 2010; Stephanson and Mascia 2010), and did not provide the detail required for locating priority areas for individual forms of management. Guerrero et al. (2010) modelled conservation opportunity across a smaller study area (146,660 hectares, with 48 land managers) and suggested the model could be expanded to larger extents. This expansion would be valid if patterns of human and social characteristics related to opportunity that were identified locally also applied more extensively (Guerrero et al. 2010). To validate this assumption, all landholders in the larger area would have to be interviewed using methods proposed by Guerrero et al. (2010) – a task that is unlikely to be feasible. The maximum entropy model used in Chapter 4 provided two significant advances over previous studies. First, it provided the resolution required to inform management while still being based on remotely collected data that served as predictors. This is highly desirable for integrating information on conservation opportunity into spatial prioritisations because of limitations on resources and time to collect data for planning. Second, the maximum entropy method can be used to inform conservation opportunity for different forms of management, while other studies have modelled opportunity for only one type of management or for generic conservation opportunity. My method can also provide insight into the factors associated with the presence and type of management.

As a second advance in understanding conservation opportunity, in Chapter 5, I used a novel method to investigate human and social characteristics related to conservation opportunity, and to compare the performance of different resolutions of data to understand conservation opportunity. Studies on conservation opportunity have mostly selected human and social characteristics believed, without testing, to be associated with specific forms of management or using a diagnostic framework to guide their decisions (e.g., Guerrero et al. 2010; Knight et al. 2010). Factors associated with conservation opportunity are likely to be context-specific, but a common diagnostic framework allows researchers to match conservation actions to particular social-ecological systems in which they are most likely to be implemented and effective. In Chapter 5, I used the social-ecological systems framework (Ostrom 2007) to guide my investigation of conservation opportunity. The social-ecological systems framework proposed by Ostrom (2007) was developed to match effective governance of common pool resources to characteristics of social-ecological systems. The use of this framework as a diagnostic tool in Chapter 5 provides an alternative and improved method to inform spatial prioritisations about conservation opportunity for different forms of management because it allows for different case-studies to be effectively compared. The method described in Chapter 5 could facilitate the scaling down of plans by directing management where it is more likely to be supported and effective. The method can also be used to investigate the resolution of data required to inform conservation opportunity. I found that, in the Solomon Islands, village characteristics related to the presence and absence of management were more significant than those associated with different management types. I suggest other studies use the same approach to identifying characteristics of successful conservation actions in different regions so that findings can be compared and contrasted, facilitating learning about conservation opportunities and translation of spatial prioritisations into conservation actions.

Contribution to knowledge on planning for the optimal suite of conservation actions and incentives to achieve conservation goals

The most appropriate form of management will be context-specific, and scaling down systematic plans is facilitated by the inclusion of management relevant to the selected planning region. To include multiple forms of management in spatial prioritisations, their relative contributions to conservation goals must be known. In Chapter 2, I advanced the existing literature on the relative contribution of different forms of management by gathering information on ecological effectiveness using key informants and a technique known as dialectic inquiry. Dialectic inquiry is a method developed for strategic decision-making by small groups of managers attempting to solve problems with limited evidence (Mitroff et al. 1979). Such data can inform rapid spatial prioritisations, allowing for effective and scientifically defensible decisions with limited information, until better data are collected (Knight et al. 2006a). I found this to be an effective method for this study because, even though no data were available on ecological effectiveness, numerous researchers and managers had insights into the relative effectiveness of different forms of management based on their field experience. Additionally, this method stimulated interest and consensus on the importance of understanding differential effectiveness of management. The Fijian government and conservation NGOs have instigated such studies, and will gradually incorporate new information on ecological effectiveness into future assessments of conservation achievements. The methods used in Chapter 2 are relevant globally, especially in countries with very limited data to support management decisions and which rely on multiple forms of management to achieve conservation goals.

6.1.2. Objective 2. Explore considerations for scaling up conservation actions to achieve regional conservation goals.

In this thesis I contributed to knowledge of scaling up conservation actions to achieve regional conservation goals in two main ways. First, my investigation of the scale mismatch between systematic planning and conservation actions informs those intending to scale up management to MPA networks about what must be considered (e.g., difficulties in crossing political boundaries, information on large-scale ecological processes). Second, I contributed to understanding how to plan for the optimal suite of conservation actions by providing a feasible method to evaluate the relative contributions of different forms of management to conservation goals, with limited data. Acknowledging the contribution of community-based management to conservation goals is crucial to encourage communities to scale up their efforts. Additionally, I compared the future benefits of systematic planning and opportunistic action with respect to habitat representation. By estimating the shortfalls in achieving conservation objectives of uncoordinated management, we increase our understanding of the incentives needed to scale up management effectively.

Addressing the scale mismatch between systematic planning and conservation actions

The knowledge, time and resources required to implement MPA networks can exceed that of implementing an individual MPA, so managers need insights into these additional considerations. Such considerations can range from knowledge of connectivity or large-scale processes, to time, resources and governance arrangements for coordinating management. The framework developed in Chapter 2 informs resource managers intending to scale up local management about what issues will need to be considered and what trade-offs there might be between the different considerations. Additionally, the framework provides resource managers working at local scales with insights into all the decisions about spatial scale that occur in systematic planning prior to considering the boundaries of individual management areas. This understanding can stimulate dialogue and cooperation between those focusing on systematic planning (in particular the spatial prioritization part) and those focusing on the implementation of management.

Contribution to knowledge on planning for the optimal suite of conservation actions and incentives to achieve conservation goals

Considering the differential contribution of various forms of management to conservation goals is crucial to scaling up community-based management, which often does not involve full or permanent restriction of resource use (Johannes 2002). Gap analyses that incorporate the relative contributions of different forms of management to conservation goals are therefore needed to measure progress in conservation, and provide direction for future conservation action. Chapter 2 is the first study to incorporate differential effectiveness of management in a national gap analysis, and provides guidance for future community-based conservation action to develop MPA networks. The methods used can be applied in areas with limited data, and consequently can be repeated in developed and developing nations that depend on various forms of management to achieve conservation goals.

Finally, I also contributed to the existing literature on understanding the incentives needed to scale up conservation actions by creating a method to investigate the benefits of coordination and systematic planning. Scenario planning allows different plausible futures to be considered by policy makers, informing decisions about what is needed to steer a country to achieve conservation or any other goal (Peterson et al. 2003). Chapter 3 was the first study in the conservation literature to use information on past community-based conservation actions to simulate their expansion into the future. It contributed to understanding the bounds of the benefits of coordination given different ways in which MPA networks can be expanded. Future studies should not only adapt these methods to understand the benefits of coordination in alternative settings, but also to better understand what incentives could be created to stimulate management that complement the existing network of MPAs.

6.2. Limitations

6.2.1. A focus on conservation goals

A limitation in scope of this thesis is that I considered conservation goals only. I chose this focus because it is the main goal of systematic plans (e.g., Fernandes et al. 2005; Green et al. 2007; TNC 2003). However, numerous goals can be integrated into marine spatial planning initiatives, some of which might be social (e.g., increased personal safety, access to goods or improvement in population health; Foale 2008b) and unrelated to conservation. Increasingly, systematic planning is being integrated with other types of spatial planning, and thus consideration of other goals will play a growing role in allocating management (e.g., Douvere et al. 2007).

Consideration of social goals in parallel with conservation goals would have changed the way I developed the thesis. For example, in Chapter 3, I could have examined the contribution of the different forms of management to fisheries or social capital as well as to conservation goals. In Chapter 4, I could have examined how different approaches to conservation (i.e. systematic planning and opportunistic) contribute to multiple goals and identify trade-offs between them. Furthermore, social goals will influence the preferred form of management. In Chapter 5, therefore, understanding the association between human and social characteristics and other social goals would improve understanding of opportunities for allocating management.

Nevertheless, my focus on conservation goals allowed me to directly address existing gaps in the conservation biology literature to better translate systematic planning into the implementation of conservation actions (Cowling and Wilhelm-Rechmann 2007; Polasky 2008). Additionally, my focus on conservation goals matched the focus of my collaborators, and ensured that results from my work could directly influence conservation decisions by resource managers in Fiji and the Solomon Islands (e.g., Jupiter et al. 2010; Kool et al. 2010).

6.2.2. Working with limited data

A limitation of this thesis is that the research was carried out in countries where only limited ecological and social data were available. Ideally, systematic planning uses extensive datasets on ecological, social, economic and political dimensions of the planning region. These data are always incomplete, but particularly so in the countries where I focused my thesis research. Thus a recommendation for planners to make timely decisions is to undertake rapid spatial prioritisations with key data layers and expert opinion, and review these decisions as better information becomes available (Knight et al. 2006a). This is the method I used. For example, the map of Fiji’s marine ecosystems used in Chapters 2 and 3 was created using all available data. However, ecosystem maps were not ground-truthed or updated to account for changes to ecosystems over time (e.g., from logging runoff or dynamite fishing). So, a substantial amount of work is still required to create better ecosystem maps and identify whether mapped ecosystems are suitable surrogates for Fiji’s marine biodiversity (e.g., Caro and O'Doherty 1999). This work is underway.

As a result of data limitations, this thesis relied heavily on expert knowledge, which has advantages but also limitations. Expert knowledge and information from interviews can be inaccurate (as discussed by Bernard 1994; Einhorn et al. 1977; McClanahan et al. 2005), but in data-limited environments use of expert knowledge is often time- and cost–effective. Thus, it is used extensively in fields such as ecology (e.g., Martin et al. 2005; Schlapfer et al. 1999) and the social sciences (e.g., Sah and Heinen 2001). The methods used in this thesis to collect expert data are well-established and designed to minimize biases (Bernard 1994). Additionally, an advantage of involving experts and stakeholders in research on systematic planning is that their involvement generates buy-in to the outcomes of systematic plans (Knight et al. 2006b).

6.2.3. Partial understanding of conservation opportunity

While I used two novel methods to inform conservation opportunity, the results provide only a partial understanding because conservation opportunity can change with people's attitudes and behaviours. Thus the methods I developed should not replace extensive community consultation and collaboration with stakeholders, which remain essential for systematic planning. More specific limitations of the two methods I used in Chapters 4 and 5 are outlined below.

Modelling conservation opportunity for different forms of management using Maxent provides an overview of areas for potential management but, while Maxent was found to be effective at modelling community-based management in Fiji, its use within other contexts might be limited. Maxent relies on an unbiased sample, or the implementation of methods that deal with sample bias. This means that, to effectively use Maxent to predict the suitability of an area for management, one must know what areas were previously considered by communities for management. In Fiji, to deal with sample bias, I sampled only from traditional fishing grounds involved in the Fiji LMMA network. Understanding sample bias in other contexts could be more difficult, thus making the use of Maxent modelling unsuitable to map conservation opportunity in these cases. Additionally, some of the factors influencing the location of management cannot be predicted from remotely collected data (e.g., individual preference for a form of management), so Maxent models are likely to be less accurate than those based on human and social characteristics that drive different forms of management.

The method used to understand conservation opportunity in Chapter 5, while illuminating human and social characteristics associated with management, is limited in that the results cannot be rapidly transformed into maps depicting conservation opportunity. The social characteristics identified to be associated to conservation opportunity in Chapter 5 are not additive (i.e. villages with three characteristics associated to conservation opportunity are not a third more likely to engage in conservation than villages with two characteristics associated to conservation opportunity), making their mapping relative rather than absolute. Thus, this thesis only provides a first step to effectively incorporate conservation opportunity within spatial prioritisations and more research is needed.

6.3. Management outcomes

Since the start of my PhD, I worked closely with management agencies (e.g., Fiji's National Protected Area Committee) and conservation-focused NGOs (e.g., Wildlife Conservation Society). This ensured that I was addressing questions that were not only contributing to advancing conservation biology, but were also tailored to meet the needs of the agencies implementing conservation actions. Consequently, the research undertaken within this thesis has contributed to marine conservation in Fiji and the Solomon Islands by guiding progress towards international conservation commitments, and influencing conservation approaches and actions.

6.3.1. Guiding progress towards international conservation commitments

Aspects of my thesis assisted Fiji and the Solomon Islands with meeting their commitments as signatories of the Convention on Biological Diversity. Signatory nations, including Fiji and the Solomon Islands, committed to advancing the Program of Work on Protected Areas (PoWPA) to develop ecologically representative networks of protected areas. As part of the PoWPA, signatories had to report on their conservation achievements in terrestrial and marine realms by 2010. The gap analysis assessing Fiji's marine conservation achievements (Chapter 3) provided the marine-focused information required by PoWPA, including national ecosystem-specific conservation objectives, and data on the coverage of each of the ecosystems by the different forms of management (for additional information see Jupiter et al. 2010). Additionally, it was the first national gap analysis to consider the contribution of the different forms of management to national conservation goals.

I contributed to the Solomon Islands' PoWPA commitments by identifying areas that provided relatively more conservation opportunity than others (Chapter 5). This work was incorporated into recommendations in the PoWPA report produced for the Solomon Islands (for additional information see Kool et al. 2010), which influenced revisions of the Solomon Islands' environmental policy.

6.3.2. Influencing conservation approaches (systematic planning vs. opportunistic conservation)

Results from Chapter 4 are being incorporated into a report by Wildlife Conservation Society, providing recommendations on incentives to coordinate conservation action. Understanding the additional benefits of systematic planning over opportunistic conservation (Chapter 4) was identified as a priority activity by the Fijian Department of Environment for 2011 and will contribute to the implementation of the Fiji National Biodiversity Strategy and Action Plan under the Inshore Fisheries thematic section. Furthermore, the results of the simulations in Chapter 4 are currently being used to support grant proposals. Specifically, these are intended to help the Fiji LMMA network design improved sustainable financing initiatives and identify alternative incentives for communities to increase the area of ecosystems under formal or informal protection.

6.3.3. Influencing conservation action

The results of the marine gap analysis (Chapter 3) and the simulation of systematic planning and opportunistic conservation into the future (Chapter 4) are being used by Fiji's National Protected Area Committee and Wildlife Conservation Society. The results are being discussed in workshops with regional and national Fijian government agencies to identify potential areas for complementary conservation actions in inshore marine waters. In 2010, provincial leaders from all Fijian Provinces met to identify new areas for conservation. They used information from the marine gap analysis (Chapter 3) to inform their recommendations about what ecosystems to protect and what forms of management to use (Jupiter et al. 2011).

6.4. Future work and knowledge gaps

The recognition of a knowing-doing gap has resulted in research to address it, but more work is needed. Research to date, including my own, has focused on two themes: (1) scaling down systematic plans, which involves adapting priorities from spatial prioritisations to incorporate local objectives, costs, values and unforeseen constraints on applying conservation actions (e.g., Guerrero et al. 2010; Knight et al. 2006b; Seddon et al. 2010); and, (2) scaling up conservation actions by coordinating and integrating them with a regional perspective to encourage complementary management (e.g., Christie et al. 2009b; Eisma-Osorio et al. 2009). My research highlighted six research gaps, described below, which I suggest are priorities for future work.

6.4.1. Characterizing conservation opportunity in different social-ecological systems

To identify patterns of human and social characteristics that lead to conservation opportunity in different social-ecological systems, many more studies organised around the social-ecological systems diagnostic framework are needed. Different forms of management will be suitable in different social-ecological systems (Johannes 2002; McClanahan et al. 2008; Ostrom et al. 2007). Thus, as multiple forms of management are increasingly incorporated into systematic planning, tailoring their selection to specific social-ecological systems would make systematic plans more relevant for managers. While the exact characteristics that allow for effective management within a particular social-ecological system will be context-specific, some common associations between human and social characteristics and conservation action can be found. For example, Ostrom (1990) identified principles common to successful resource governance in common property systems. These principles were recently reviewed and modified by Cox et al. (2010), and most were found to be robust. While effective resource governance is a precondition of successful management, there may be additional human and social characteristics that influence conservation (e.g., existence value of biodiversity; Foale 2001; Foale and Macintyre 2005: Foale 2008a). Thus, more diagnostic studies that investigate human and social characteristics associated with conservation actions within and outside common property systems would be useful. Some of this research is already being undertaken (Knight et al. 2010), but such studies have yet to be organised around diagnostic frameworks that facilitate comparison and guide generalisations. A better understanding of conservation opportunity would help to direct comprehensive social assessments and minimise the knowing-doing gap.

6.4.2. Integrating changes to policy, markets and government incentives when predicting future conservation

A better understanding is needed of how trends in conservation action will change as a result of policy, markets and government incentives so that we can design or respond to them effectively in the future. Conservation opportunity will change with time because of changes in, for example, public opinion, new rules and regulations on resource extraction, local and regional markets, and factors affecting the availability of alternative livelihoods for stakeholders. For instance, national parks established in Sumatra in the 1980s were rapidly converted to agriculture as a result of rising international coffee prices driven by the failure of the Brazilian coffee plantations (Brechin 2003). Scenario planning can be useful to investigate the consequences of such changes and explore alternative plausible futures (Peterson et al. 2003). In Chapter 4, I used scenario planning to construct two alternative futures for conservation - a systematic and an opportunistic one. This allowed me to understand the plausible bounds of conservation achievements given different degrees of coordination. I suggest scenario planning be further explored to better understand the added benefit of different approaches to conservation and the potential influence of alternative policies.

6.4.3. Understanding the contributions of multiple forms of management to conservation goals

A better understanding of the contributions of different forms of management to conservation goals is critical for future planning. A Before-After Control-Impact-Pairs design to test for the impact of different forms of management over extended periods would be ideal (Russ 2002; Lincoln-Smith et al. 2006. However, numerous challenges are associated with the implementation and enforcement of different forms of management, and these make such experiments difficult to complete.

Reasons include the political or community pressure to change management through time (Russ and Alcala 1999) and lack of compliance to implemented rules (e.g., Robbins et al. 2006). Still, ecological studies on the impacts of different forms of management are critical to understanding their relative contributions to conservation goals.

6.4.4. Reviewing successes and challenges of scaling up and down

Reviews of existing conservation initiatives that are scaling up conservation actions and scaling down systematic plans are needed to identify the types of institutions, resources and processes that are effective. Whilst theoretical frameworks are available, both scaling up and scaling down remains challenging for reasons such as lack of leadership, resources, willingness, technical capacity, and constraints of political boundaries. Yet there are numerous conservation initiatives currently attempting to scale up and down, and their insights can inform other conservation initiatives. Some of these scaling attempts are being documented, for example in the Philippines (e.g., Armada et al. 2009), South Africa (e.g., Knight et al. 2011) and Australia (e.g., Fernandes et al. 2005). However, the ongoing trend (with some exceptions, e.g. Knight et al. 2011) is to highlight achievements only while ignoring failures of conservation initiatives, even though failures can also offer much insight, especially if some challenges have been overcome (Knight 2006). Thus, a review and evaluation of attempts to scale up and down, and related challenges, is critical to inform future conservation action.

6.4.5. Changing the preferred conservation approach based on multiple goals and their trade-offs

Research on how the benefits of systematic planning over opportunistic approaches change with different goals and on the trade-offs between achieving different goals is needed. Ecosystem representation (a surrogate for biodiversity representation) is only one of the many goals relevant to the design of MPAs; other goals include connectivity (e.g., Almany et al. 2009), improved fisheries (e.g., Eisma-Osorio et al.

2009), increased resilience to catastrophic events (e.g., Game et al. 2008), and socioeconomic acceptability (e.g., Adams et al. 2011). MPA networks that address multiple goals will often require trade-offs between them (Aswani and Hamilton 2004; Roberts et al. 2003; Stewart and Possingham 2005; Weeks et al. 2010). In Chapter 3, I examined the marginal benefit of a systematic versus an opportunistic approach to conservation in Fiji, accounting only for goals of ecosystem representation. Future studies should examine this same question but consider multiple goals. Such studies should also consider whether these goals are local or regional, because this will influence whether and by whom they are supported. This information would allow planners and policy makers to develop or improve policy to encourage the implementation of MPA networks.

6.4.6. Adaptive planning

Research is needed on when, how and why plans should be adapted to facilitate integration of local and regional perspectives when scaling up and down. Moving between regional and local perspectives requires adaptive planning processes and products. Undertaking adaptive planning can be difficult for two main reasons. First, the need for adaptation indicates a lack of understanding about the optimal conservation actions to address the resource issue, making it more difficult to generate stakeholder support. Second, stakeholders affected by the systematic plan will be reluctant to support it if, after they invest in adjusting to the systematic plan, it could be reversed (Grantham et al. 2009). However, methods and institutions are being developed to encourage adaptability. For example, social-learning institutions are being established to bring together researchers, landowners and government on a yearly basis to discuss research and management directions (Knight and Cowling 2006). Another example is that of peer review processes being set up to evaluate whether conservation projects are delivering the expected outcomes (CMP 2011). From these beginnings, research on when, how and why plans should be adapted is a priority.

6.5. Conclusion

MPA networks can play an important role in mitigating the decline in biodiversity and ecosystem services. To create effective MPA networks that address regional and local conservation goals, a combination of systematic and opportunistic approaches to conservation is needed. Systematic planning can and must simultaneously consider and be tailored to suite the existing ecological, social, economic and political aspects of planning regions in order to direct conservation action on-ground. Furthermore, opportunistic conservation actions should be coordinated if MPAs are to be more than collections or the sums of their parts. Coordination offers the prospect of MPA networks offering emergent benefits, including resilience across time to threats acting at multiple scales. This thesis is a step towards understanding how systematic planning can be complementary to opportunistic conservation, by providing methods and pragmatic recommendations that can inform future decision-making.

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Appendix

8.1. Systematic planning case studies

To illustrate decisions about scale in systematic planning, I reviewed the literature on marine conservation initiatives in Malaysia, Indonesia, the Philippines and several Pacific island nations. I focused on conservation initiatives that had followed a recognised systematic planning framework. However, I also looked for regional conservation initiatives that did not follow a systematic approach but aimed to create ecologically functional networks of MPAs. I included the latter group because there are important lessons to be learned from these initiatives for effective implementation. Non-systematic initiatives cover similar extents as systematic plans and are more advanced in terms of applying conservation actions. The translation of systematic plans into effective conservation actions, if happening at all, is very preliminary.

To find case studies, I searched multiple electronic sources including the Web of Science, Science Direct, and Scopus using the search terms “marine protected area” AND “conservation” OR “planning” AND “Pacific Island” OR “Malaysia” OR “Indonesia” OR “Philippines”. I searched conservationevidence.com using the terms “marine protected area” OR “conservation plan” for studies within Oceania. I searched reefbase.org in the Protected Area Global Database where I looked for examples of marine protected area networks in the Pacific. Contacting regional experts and browsing the websites of conservation NGOs sought other examples and further information on case studies. Most of these additional documents were in the grey literature. To decide which case studies to include in this review, I used the following criteria: (1) the conservation initiative was intended to guide the design and implementation of an MPA network or an marine park with spatially explicit multi-use management zones (including marine protected areas); or (2) the initiative had the objective of designing systems of complementary MPAs or management zones. I identified 11 case studies as providing sufficient information to be included in this study. Their key characteristics related to scale are summarised in the Appendix Table 8.1.

Table 8.1. Systematic planning case studies

	MPA network	*Objectives*	*Time frame and budgets of the conservation initiative*	*Status*	*Planning region extent (sea)*	*Variables determining choice of planning region*	*Data used for planning*	*Reasons for data choice*	*Planning software*	*Applying conservation actions*	*References*
A	South Cebu MPA Network, Philippines	Implementation of MPA network	Application of conservation actions and collection of data has extended for over 10 years. Budget unknown	Conservation actions proceeding, but MPA network design is still under development. 22 MPAs have been established through community-based initiatives.	1250 km^2	Based on social network and biophysical information	Data on conditions of natural resources, resources use and social information on villages.	Used data available and undertook limited additional data collection	Not used	MPAs currently cover 300 ha, 0.24% of municipal waters in the planning region. MPAs initially set up to address local issues. Currently these are also directed toward representation and complementarity goals.	Laffoley (2008) Eisma-Osorio et al. (2009)
B	Kimbe Bay, Papua New Guinea	Assessment of what is currently protected and implementation of MPA network	4 years for data collection and identification of potential MPAs. Application of conservation actions underway. US$400 000 for data collection and planning	Broad areas of interest established. Detailed delineation of MPAs and application of conservation actions in progress. 21 managed areas have been established since the initial plan was developed.	13000 km^2	Primarily based on biophysical criteria (previously defined as a seascape) then modified to suit social boundaries. Provincial boundaries delimit the eastern part of the planning region. The western boundary was adjusted to include groups of selected villages.	30-60 m resolution maps derived from satellite imagery. Data from field surveys of coral reef communities (particularly reef fishes), mangrove forests, seagrass beds, fish spawning aggregation sites, and turtle nesting areas. Complemented by maps derived from local and indigenous knowledge.	Used data available and undertook limited additional data collection	MARXAN (with 10 ha planning units)	Areas where management has been implemented so far are those where there was the strongest interest for conservation. These were prioritised in the design stage. Average size of the multiple-use managed areas is 957 ha. The median is 52 ha. Currently there are 2 designated MPAs, one is 23 ha and the other is 46 ha.	Green et al. (2007) Green et al. (2009) Nate Peterson pers. Comm.
C	Danajon Bank MPA network, Northern Bohol Island, Philippines	Implementation of the MPA network	Application of conservation actions and data collection has been undertaken for over 10 years. MPA network design still	Conservation actions proceeding, but no MPA network design has been developed.	Danajon Banks cover 272 km^2. Management is envisioned across the whole double	Ecological criteria informed the vision for the Danajon Bank MPA network. However, conservation efforts initially concentrated on 4 municipalities within Danajon Bank and are now expanding to include other municipalities in the region.	Maps and site-specific information derived from local biological data (transects), fisheries data (~2 km^2 resolution) and local knowledge	Used data available and undertook limited additional data collection	Not used	MPAs and managed areas are small and in different stages of implementation. These are within areas where there is strong community support. Information was only available for 26 of the 31 managed and MPAs. These varied from 7.1 ha to 128 ha. The average size is 42 ha and	Armada et al. (2009) Christie et al. (2006)

			underway Budget unknown		barrier reef.					the median size is 35 ha.	
D	Roviana and Vonovana Lagoon, Solomon Islands	Implementation of MPA network	Implementation of conservation actions taking place for over 10 years Budget unknown	Conservation actions proceeding, but no MPA network design has been developed. 31 MPAs have been established.	Initiative began across the Roviana and Vonovana lagoons. Boundaries have not been fixed	This region is traditionally owned by clans friendly to the Christian Fellowship Church. The Church plays a large role in this region's governance and provides leadership for the MPA initiatives.	Indigenous ecological knowledge and satellite imagery (1:24 000) were used to produce habitat maps. Sea tenure governance boundaries were collected to inform management.	Used data available and undertook limited additional data collection	Not used	MPAs are placed in ecologically significant sites (e.g. spawning aggregations) that are also socially suitable (e.g. lack of territorial conflict, community support, and proximity to villages to facilitate monitoring and enforcement). These vary in size from 34 to over 500 ha.	Aswani and Lauer (2006) Aswani and Hamilton (2004) Aswani et al. (2007) Aswani and Furusawa (2007)
E	Pere, Papua New Guinea	Local-scale MPA network	MPA design started in 2008 and is ongoing. Budget unknown	Draft MPA network design created. No quantitative targets for protection have been established.	75 km^2	Alignment of boundaries with the Pere traditional reef tenure. Pere village includes 5 wards managed together by 5 local councillors.	Maps derived from local knowledge	Used data available	Not used	*Information not available*	Pere Community (2009)
F	Palau	Assessment of what is currently protected and guidance for future conservation actions	5 years for MPA design. Application of conservation actions still underway Budget unknown	Prioritisation at a national level completed in 2007. Since then, application of conservation actions has begun. Further prioritisation and delineation of MPAs to take place at state level	3114 km^2	National boundaries used because this was an exercise to identify national priorities	1:20 000 habitat maps derived from a combination of satellite imagery and indigenous knowledge	Used data available	SPOT and MARXAN (with 15 ha planning units)	Two terrestrial protected areas have been established (870 and 30 ha). Both are within 5 to 8 km of the communities. They area easily accessed by tourists and dependent on revenue generated by the tourism industry.	Hinchley et al. (2007) U. Sengebau, pers comm..
G	Reimanlook, Marshall Islands	Identify conservation features of interest, and develop step-by-step guidelines for establishing a conservation actions on individual atolls.	6 months for MPA network design AU$ 130 000 for data collection and MPA design	National gap analysis completed and a national MPA design strategy developed. Future work requires application of the national strategy, identification of priority atolls for national investment, and designing	14,249 km^2	Used national boundaries covering marine resources to a depth of 100m and including the entire lagoon and all terrestrial areas	Habitat maps based on satellite imagery, nautical charts, and coral reef maps created by IMARS using Landsat 30m high-resolution imagery. These were complemented with more detailed data from local knowledge and	Used data available	Not used	Individual atoll local governments have established new MPAs (additional to traditional ones) – e.g. in Jaluit, Rongelap and Ailuk, but none of these is yet an explicit outcome from the Reimanlok plan.	Reimaan National Planning Team (2008)

				MPAs and applying conservation actions within individual atolls.			literature review				
H	Wakatobi Marine National Park, Indonesia	Regional-scale multi-use zoning of a previously declared marine park, including MPAs	Marine park rezoned between 2006 and 2007. Budget unknown	MPA network design developed and signed by the district government. Socialisation of the plan (where plan is shown to and discussed with communities) in progress.	13900 km^2	The planning region adhered to the boundaries of the marine park established in 1997. These became the new boundaries of the Wakatobi district.	Maps based on rapid ecological assessment, 60 m resolution satellite imagery, nautical charts, and local and traditional knowledge.	Used data available and undertook limited additional data collection	MARXAN	The zoning system produced MPAs varying from 13 to 365 km^2. Size and location of MPAs based on biological data and social considerations (resource use assessments). Zoning system is still in a period of socialisation. Enforcement is largely directed at 'outsiders' rather than local communities. Effectiveness of conservation actions constrained by insufficient funding and lack of equipment required to delineate zones and empower communities to protect their own resources.	Renosari and Elverawati (2007) Barmawi et al.(2005) TNC (2008) Wilson pers comm.
I	Karimunjawa National Park, Indonesia	Regional-scale multi-use zoning of a previously declared marine park	Zoning of marine park revisited between 2003-2005. Budget unknown	Design of the MPA network completed. Baseline data have been collected and procedures for monitoring put in place. The application of conservation actions has started.	1,101 km^2	The planning region adhered to the boundaries of the previously established marine park. The whole Karimujawa archipelago is found within the park	Ecological, socioeconomic and fishing surveys	Used data available and undertook limited additional data collection	Not used	Zoning incorporated both biological and socio-political information. 1116 km^2 were zoned for different management arrangements, of which 4.45 km^2 were designated as MPAs. Application of designated management arrangements difficult although awareness of regulations are increasing.	TNC et al. (2008) Campbell (2006)
J	Berau Marine Conservation Area, Indonesia	Regional-scale multi-use zoning of a previously declared marine park	Started in 2005. Still underway Budget unknown	MPA network design in progress	12000 km^2	Follows the boundaries of the Berau District and covers 95% of the district waters.	A combination of scientific and traditional knowledge.	Used data available and undertook limited additional data collection	MARXAN	Parameters used in deciding the location and size of MPAs were biological (e.g. targeted spawning aggregations of groupers and turtle nesting sites). MPAs vary from 1.5-20 km^2. No information on the application of actions is	TNC et al. (2008)

										available.	
K	Federated States of Micronesia	Assessment of what is currently protected and guidance for future application of conservation actions	Data collection and potential areas for management took 1 year (2001-2002) Budget unknown	Prioritisation completed. Potential MPAs are given two levels of importance: those for which immediate conservation is needed; and those where conservation action is less urgent.	*Information not available*	Boundaries of the Federated States of Micronesia	Maps derived from local knowledge and inventories	Used data available	Not used	*Information not available*	TNC (2003)

8.2. Percentage contributions to quantitative objectives based on expert assessment of the differential effectiveness of various forms management

	Permanent closures	Temporary closures (controlled)	Temporary closures (uncontrolled)	Other management	Total (considering differential effectiveness)
Fringing Reef					
Corals	0.56	2.01	1.28	15.86	19.72
Targeted invertebrates	0.56	1.76	0.26	7.93	10.51
Non-targeted	0.56	2.27	1.53	17.84	22.20
Targeted fish	0.56	2.01	0.38	7.93	10.89
Non-targeted fish	0.56	2.27	1.28	17.84	21.95
Coralline algae	0.56	2.01	1.28	15.86	19.72
Range	***0.56***	***1.76-2.27***	***0.26-1.53***	***7.93-17.84***	***10.51-22.20***
Non fringing reef					
Corals	1.17	1.89	0.17	14.10	17.33
Targeted invertebrates	1.17	1.65	0.03	7.05	9.90
Non-targeted	1.17	2.13	0.25	15.86	19.41
Targeted fish	1.17	1.89	0.05	7.05	10.16
Non-targeted fish	1.17	2.13	0.19	15.86	19.35
Coralline algae	1.17	1.89	0.17	14.10	17.33
Range	***1.17***	***1.65-2.13***	***0.03-0.25***	***7.05-15.86***	***9.90-19.41***
Mangrove					
Targeted invertebrates	1.61	2.89	0.00	8.96	13.46
Non-targeted	1.61	3.43	0.01	22.39	27.45
Targeted fish	1.61	1.80	0.00	6.72	10.14
Non-targeted fish	1.61	2.16	0.00	13.44	17.22
Mangrove	1.61	3.43	0.01	11.20	16.25
Seabirds	1.61	3.43	0.01	8.96	14.01
Bats	1.61	3.43	0.01	11.20	16.25
Range	***1.61***	***1.80-3.43***	***0.00-0.01***	***6.72-22.39***	***10.14-27.45***
Intertidal					
Targeted invertebrates	0.13	2.11	0.00	7.22	9.46
Non-targeted	0.13	2.71	0.00	16.25	19.09
Targeted fish	0.13	2.41	0.00	7.22	9.76
Non-targeted fish	0.13	2.71	0.00	16.25	19.09
Seabirds	0.13	2.86	0.00	9.03	12.02
Range	***0.13***	***2.11-2.86***	***0.00***	***7.22-16.25***	***9.46-19.24***
Other benthic substrate (0-5m)					
Targeted invertebrates	0.28	0.46	0.38	11.37	12.49
Non-targeted	0.28	0.59	1.01	25.59	27.47
Targeted fish	0.28	0.52	0.63	11.37	12.81
Non-targeted fish	0.28	0.59	1.01	25.59	27.47
Range	***0.28***	***0.46-0.59***	***0.38-1.01***	***11.37-25.59***	***12.49--27.47***
Other benthic substrate (5-10m)					
Targeted invertebrates	0.04	1.28	0.85	19.06	21.23
Non-targeted	0.04	1.65	2.27	42.88	46.84
Targeted fish	0.04	1.46	1.42	19.06	21.98
Non-targeted fish	0.04	1.65	2.27	42.88	46.84
Range	***0.04***	***1.28-1.65***	***0.85-2.27***	***19.06-42.88***	***21.23-46.84***
Other benthic substrate (10-20m)					
Targeted invertebrates	0.01	0.38	0.26	10.98	11.63
Non-targeted	0.01	0.49	0.69	24.71	25.89
Targeted fish	0.01	0.43	0.43	10.98	11.85
Non-targeted fish	0.01	0.49	0.69	24.71	25.89

Range	***0.01***	***0.38-0.49***	***0.26-0.69***	***10.98-24.71***	***11.63-25.89***
Other benthic substrate (20-30m)					
Targeted invertebrates	0.10	0.20	0.11	12.85	13.26
Non-targeted	0.10	0.26	0.30	28.92	29.58
Targeted fish	0.10	0.23	0.19	12.85	13.37
Non-targeted fish	0.10	0.34	0.30	28.92	29.66
Range	***0.10***	***0.20-0.34***	***0.11-0.30***	***12.85-28.92***	***13.26-29.66***

8.3. Survey: LMMA key informants

1) Name: ....

2) Sex: ☐ Male ☐ Female

3) Organisation: ....

4) Position: ....

5) Are you from Fiji?

☐ Yes ☐ No

6) How long have you been in Fiji?

☐ < 1 year ☐ 1-2 years ☐ 2-5 years ☐ 5-10 years ☐ >10 year

First, I would like you to outline your experience in establishing LMMAs in Fiji

7) How many years have you been working with implementing LMMAs in Fiji?

☐ < 2 years ☐ 2-5 years ☐ 5-10 years ☐ >10 years

8) How many LMMAs have you helped to establish?

☐ <5 ☐ 5-10 ☐ 10-25 ☐ >25

9) How many LMMAs are you currently working with?

☐ 1 ☐ 2 ☐ 3 ☐ 4 ☐ 5 ☐ >5

10) Which Qoliqolis do you work with: ....

....

....

....

....

....

....

11) To what extent are the fisheries and environment government departments involved in the Qoliqoli in which you work?

- ☐ Very highly involved (work closely with the community, e.g. influence education and awareness, and enforce resource use regulations
- ☐ Highly involved (enforce resource use regulations on a daily or weekly basis)
- ☐ Occasional involved (enforce resource regulations on a monthly basis)
- ☐ Generally not involved (enforce resource regulations on a yearly basis or less)

DETERMINING THE EXTENT AND LOCATION OF LMMAS

I am trying to understand the factors that will influence the size and location of LMMAs and no take reserves. To do this I would like information on your perception of:

- **Small and large LMMAs, nearshore and offshore**
- **Small and large no take reserves in Fiji, nearshore or offshore**

1) Do all LMMAs you work with follow Qoliqoli boundaries?

   ☐ Yes *(please skip to the next section)*
   ☐ No

2) Does the LMMA you work in have temporary or permanent closures?

3) What do you consider to be a small closure in Fiji?

   ☐ < 0.5km^2
   ☐ < 1km^2
   ☐ < 3km^2
   ☐ < 5km^2
   ☐ < 50% of Qoliqoli
   ☐ Other

4) What do you consider to be nearshore closure in Fiji?

   ☐ Adjacent to the beach
   ☐ < 1km from the beach
   ☐ < 5km for the beach
   ☐ <10km from the beach
   ☐ < 20km from the beach
   ☐ Up to Qoliqoli boundaries
   ☐ Up to the reef crest
   ☐ Other

5) In your opinion, what influences the **location** of **closures** within the LMMAs you work with? ..

....

....

....

....

....

....

....

6) In your opinion, what influences the **size** of the **closures** within the LMMAs you work with? ...

....

....

....

....

....

7) Based on your experience, please rank the importance of the factors that influence the villagers in determining the **location** of **closures**. Please tick 0 if they played no influence. 1 indicates least important and 5 indicates greatest importance. For the factors you have considered important, could you please provide a brief explanation as to why? Also, if you have identified other factors, please list, rank and explain them.

	Not Present	Present					Explanation of why this variable is important
	0	1	2	3	4	5	
Distance from village							
Location of enforcement office							
Size of reserve							
Number of people in village							
Number of fishers that don't have motor boats							
Location of adjacent reserves							
Distance from adjacent communities							
Habitat types surrounding community							
Habitat health							
Fishing pressure							
Preference of chief							
Biodiversity value							
Important Ecological Processes (e.g. spawning aggregations)							

8) Based on your experience, please rank the importance of the factors that influence the **size** of **closures**. Please tick 0 if they played no influence. 1 indicates least important and 5 indicates greatest importance. For the factors you have considered important, could you please provide a brief explanation as to why? Also, if you have identified other factors, please list, rank and explain them.

	Not Present	Present					Explanation of why this variable is important
	0	1	2	3	4	5	
Distance from village							
Size of Qoliqoli							
Location of reserve							
Number of people in village							
Number of fishers that don't have motor boats							
Size of adjacent reserves							
Distance from adjacent communities							
Habitat types surrounding community							
Naturally/locally perceived demarcations of environment							
Preference of chief							
Ability to conserve biodiversity							
Important ecological processes (e.g. spawning agrregation)							

The next questions are about the whole LMMA you work with (including the no take zones)

9) What was the awareness/compliance of the national fisheries regulations before you started working with the communities implementing LMMAs?

- ☐ High
- ☐ Medium
- ☐ Low

10) What is the current awareness/compliance of the national fisheries regulations?

- ☐ High
- ☐ Medium
- ☐ Low

11) What were the focus resource management measures for the setup of the LMMAs you work with?

	Number
☐ Size limits	_____
☐ Gear restrictions (types and mesh sizes)	_____
☐ Species ban	_____
☐ Control of harvesting of specific life stages	_____
☐ Licensing	_____
☐ No take zones	_____

Other (please specify) ....

12) Are particular **Qoliqolis** targeted for management?

- ☐ Yes *(please skip to the next section)*
- ☐ No

13) Please list all the factors that influence why a particular Qoliqoli would be targeted for management? ....

....

....

....

....

....

....

CONSTRAINTS AND OPPORTUNITIES FOR CONSERVATION

Now I am going to ask you about opportunities and constraints faced when setting up LMMAs

Opportunities

1) In your experience, what have been the main **reasons** for **communities** establishing **LMMAs**

2) Are these the same **reasons** for communities establishing **closures**?

   ☐ *Yes (please skip the next question)* ☐ *No*

3) Could you please list additional variables or explain **how** the **reasons** for **establishing LMMAs and closures** are **different**?

4) Do **closures target** specific habitats or features?

   ☐ *Yes*
   ☐ *No (Please go to question 8)*

5) What do they **target**?

   ☐ Coral reefs ☐ Lagoons

- ☐ Seagrass
- ☐ Mangroves
- ☐ Spawning aggregations
- ☐ Areas that contain many habitats
- ☐ Beaches
- ☐ Areas with low fishing pressure
- ☐ Areas with medium fishing pressure
- ☐ Areas with high fishing pressure
- ☐ Areas thought to have healthier habitats
- ☐ Areas where resource degradation is perceived
- ☐ Areas with easy access for tourists

Other: ....

Constraints

6) What factors **constrain** conservation initiatives in Fiji?

- ☐ Dependence on resources for subsistence or cash
- ☐ Ability to monitor/enforce regulations
- ☐ No perceived need for resource regulations
- ☐ Lack of integration between land and sea management

Other: ....

7) Assuming communities are interested in expanding closures, **how much** do you believe they could effectively protect?

- ☐ <25%
- ☐ 25-50%
- ☐ 50-75%
- ☐ >75%

8) What percentage of the villagers (male and female) in the Qoliqoli depend on marine resources for their day to day livelihood, whether as an primary means of obtaining cash or for subsistence?

- ☐ <25%
- ☐ 25-50%
- ☐ 50-75%
- ☐ >75%

9) What percentage of the villagers (male and female) fish using motorboats?

- ☐ <25%
- ☐ 25-50%
- ☐ 50-75%
- ☐ 75-100%

10) How far from the communities can fishermen fish without motorboats?

- ☐ <1 km
- ☐ <3 km
- ☐ <5 km
- ☐ <7 km
- ☐ < 10 km
- ☐ <15 km

11) What percentage of fishers do you believe have the opportunity to leave the fishing industry and earn an equal or better livelihood locally?

- ☐ none
- ☐ <25%
- ☐ 25-50%
- ☐ 50-75%
- ☐ 75-100%

12) If fishers do not have the opportunity for alternative livelihoods what are the factors constraining them? If none of the specified factors, please indicate 0 for “Not Present”. If present please indicate the level of importance of that factor from 1 (minor influence) to 5 (major influence):

	Not Present	Minor				Major
	0	1	2	3	4	5
Prefer fishing over other available jobs	☐	☐	☐	☐	☐	☐
Do not have access to required financing	☐	☐	☐	☐	☐	☐
Do not have required skills	☐	☐	☐	☐	☐	☐
No other jobs available locally	☐	☐	☐	☐	☐	☐

☐ Other (please specify)..

13) What other types of jobs are available within your village?......................................

....

....

....

....

....

....

....

SCALING UP CONSERVATION INITIATIVES

Now, I would like to ask about scaling up conservation initiatives to achieve regional scale marine protected area networks.

1) If you work with more than one LMMA, did you consider the LMMAs already set up when working with communities on what and how to manage and protect?

☐ Yes ☐ No (please go to question 3)

2) What aspects of established LMMAs informed your new LMMAs, please provide examples

☐ General lessons learned Example: ....

☐ Socioeconomic constraints Example: ....

☐ Biological successes/failures Example: ....

☐ Complementary habitats Example: ....

☐ Complementary management Example: ....

☐ Connectivity between MPAs Example: ....

☐ Other (please specify) ....

3) What are the **constraints** to **resource management collaboration** between **communities** within the same Qoliqoli? If none of the specified factors apply, please indicate 0 for "Not Present". If present, please indicate the level of importance of that factor, 1 indicates minor importance and 5 indicated major importance.

	Not Present	Minor				Major
	0	1	2	3	4	5
Differences in culture	☐	☐	☐	☐	☐	☐
Different perceptions of resource decline	☐	☐	☐	☐	☐	☐
Different interests in conservation/resource management	☐	☐	☐	☐	☐	☐
Lack of capacity to coordinate	☐	☐	☐	☐	☐	☐
Different dependences on marine resources	☐	☐	☐	☐	☐	☐
Lack of leadership in the community	☐	☐	☐	☐	☐	☐
Previous conflict	☐	☐	☐	☐	☐	☐
Competitiveness for resources	☐	☐	☐	☐	☐	☐
Church affiliations	☐	☐	☐	☐	☐	☐

☐ Other (please specify) .... and rate ....

4) What are the **constraints** to **resource management collaboration** between different **Qoliqolis**? If none of the specified factors apply, please indicate 0 for "Not Present". If present, please indicate the level of importance of that factor, 1 indicates minor importance and 5 indicated major importance.

	Not Present	Minor				Major
	0	1	2	3	4	5
Differences in culture	☐	☐	☐	☐	☐	☐
Different perceptions of resource decline	☐	☐	☐	☐	☐	☐
Different interests in conservation/resource management	☐	☐	☐	☐	☐	☐
Lack of capacity to coordinate	☐	☐	☐	☐	☐	☐
Different dependences on marine resources	☐	☐	☐	☐	☐	☐
Lack of leadership to represent Qoliqoli	☐	☐	☐	☐	☐	☐
Geographic separation between Qoliqolis willing to work together	☐	☐	☐	☐	☐	☐

☐ Other (please specify).. and rate ....

5) Based on your experience, what do you think is the **maximum number** of **communities** that could collaborate in resource management (adjust management strategies considering the management strategies of others, for mutual benefit)?

☐ < 5 communities (same Qoliqoli)
☐ 5 -10 communities (same Qoliqoli)
☐ < 5 communities (different Qoliqoli)
☐ 5 - 10 communities (different Qoliqoli)
☐ All communities within the Qoliqoli
☐ No restrictions on the number of communities as long as they all belonged to the same Qoliqoli
☐ No restrictions on the number of communities or which Qoliqoli's they belonged to

6) Based on your experience, what do you think is the **maximum number** of **Qoliqolis** that could collaborate in resource management (adjust management strategies considering the management strategies of others, for mutual benefit)?

☐ 2 Qoliqoli
☐ 2 – 5 Qoliqoli
☐ 5 -10 Qoliqoli
☐ All Qoliqoli's within an island
☐ No maximum number

8.4. Survey: Opportunities and constraints for marine resource management in the Solomon Islands (household survey)

***Project overview*:** This study is about opportunities and constraints for establishing rules for fishing in villages in the Solomon Islands. Our specific research questions are: (1) Are there village characteristics that facilitate the establishment of fishing rules in the Solomon Islands? (2) Are different fishing rules associated with different village characteristics?

Taking part in this study is voluntary and you can stop undertaking the survey at any time without prejudice or explanation, and withdraw any unprocessed information provided. All information provided during this survey will be kept strictly confidential.

Background

1) Name: ____

2) Clan/tribe name:____

3) Name of the language you speak at home: ____

4) Religious affiliation:

☐ Traditionalist
☐ SDA
☐ AOG
☐ SSEC
☐ Jehovah's Witness
☐ COM- Anglican
☐ COC
☐ CotWL
☐ BC
☐ RHEMA
☐ UC
☐ CFC
Other_______________

First, I would like to ask you about your fishing rights

5) Presently, who else is allowed to fish in the same place as you for finfish fisheries

☐ My wife/husband
☐ My children
☐ People in my clan
☐ People in my wife/husbands clan
☐ Anyone in my village
☐ Anyone - including people outside my village

6) Presently, who else is allowed to fish in the same place as you for trochus or other cash fisheries

☐ My wife/husband
☐ My children
☐ People in my clan
☐ People in my wife/husbands clan
☐ Anyone in my village
☐ Anyone - including people outside my village

7) Do you have sea/river ownership disputes with others clans/villages? Could you provide a brief explanation of what they are?

____
____
____
____
____
____
____
____

8) Where is your favourite fishing ground to get fish for food? (locate and name all the places)

____
____
____
____
____
____
____

9) Where is your favourite fishing ground to get fish to sell? (locate and name all the places)

____
____
____
____
____

Now I will ask you about the fishing rules in your fishing grounds

10) Are there fishing rules in your fishing grounds? (if possible, mark on map)

☐ No

☐ Yes

Custom	New
☐	☐ Temporary closures
☐	☐ Permanent closures
☐	☐ Species restrictions
☐	☐ Size limits
☐	☐ Fishing gear restrictions
☐	☐ Temporal restrictions
☐	☐ Effort restrictions

Details (incl. whether restrictions are clan or family based)

11) Do the fishing rules apply for both fish you eat and fish you sell? (note if it is different for different rules)

Can you tell me about why these fishing rules were established? (note if it is different for different rules)

12) What benefits do you get from the establishment of these fishing rules? (note if it is different for different rules)

13) Are there any disadvantages for you associated to the establishment and maintenance of the fishing rules? (e.g. area used for fishing had to change)

____

____

14) What happens if someone breaks the rules?

____

____

____

____

____

____

15) During the past week, how many of the people with fishing rights do you believe respect the fishing rules?

☐ Everyone ☐ Half of the people

☐ Most people ☐ Only some people ☐ None

16) Were you involved in making the fishing rules? ☐ Yes ☐ No

17) Who is most responsible for making the fishing rules?

________ Chief _________Church ________ Organisation

________ Community _________ Business

18) Who is most responsible for enforcing the fishing rules?

________ Chief _________Church ________ Organisation

________ Community _________ Buiseness

19) The last time a fishing rule was established, which groups in the village supported or did not support the fishing rules (groups may be based on jobs, sex, clan/tribe, religion, age, etc)

____

____

____

____

20) Do outsiders fish in your fishing grounds? ☐ Yes ☐ No

Now, I would like to ask you questions about your life in your village.

21) In the past year, how often were organized meetings of village residents held to discuss village issues and events?

☐ Daily ☐ Quarterly ☐ Yearly
☐ Weekly ☐ Half yearly ☐ Never
☐ Monthly

22) What is generally discussed in village meetings? (if they previously answered never ask what they would like to see discussed)

☐ Health ☐ Marine resources ☐ ___________________
☐ School ☐ Development ☐ ___________________
☐ Terrestrial resources ☐ ___________________ ☐ ___________________

23) In the last village meeting that you went to, how many people in your people attended?

☐ Everyone ☐ Half of the people
☐ Most people ☐ Only some people ☐ None

24) Is this the typical amount of attendees in a village meeting?

☐ Yes ☐ No, typically more ☐ No, typically less

25) I'd like to know how many people in your clan you generally would trust to respect the clan rules?

☐ Everyone ☐ Half of the people
☐Most people ☐ Only some people ☐ None

26) I'd like to know how many people in your village you generally would trust to respect the rules of the village?

☐ Everyone ☐ Half of the people
☐ Most people ☐ Only some people ☐ None

27) Who is the most important person in the village regarding resource management decisions?

Name: _____________________________ Position___________________________________
Name: _____________________________ Position__________________________________
Name: ______________________________ Position____________________________________

28) Who has the most influence in convincing the villagers about resource management problems?

Name: _______________________________ Position__________________________________
Name: _______________________________ Position__________________________________
Name: ________________________________ Position___________________________________

29) In the last 5 years have you had group efforts to improve to repair or build village infrastructure or clean the environment? ☐ Yes ☐ No ☐ Both

30) How many types of associations/ village groups are members of your household involved with?

For example fishing and women's group (ask them to name the types)

____

____

I would like to ask you questions about your jobs (including fishing).

31) Can you tell me which of your jobs you spend most time on?

☐ Farming ☐ ____________________ ☐ ____________________

☐ Fishing ☐ ____________________ ☐ ____________________

32) Which of the jobs above are most important for cash income?

1st ____________________ 2nd ____________________ 3rd ____________________

33) How many times per week does someone in this household go fishing?

____

34) How many times did you eat what you fished in the last 3 days?

____

35) How many times did you eat other meat in the last 3 days?

____

36) Last time you went fishing, how much of your catch did you eat and sell?

☐ Eat all, sell none

☐ Eat most, sell a little bit

☐ Half-half

☐ Sell most, eat a little bit

☐ Sell all, eat none

37) During the past year, how does the **number** and **size** of fish you catch from your (sea/river/lake) compare to 10 years ago?

No.
☐ More fish
☐ Same
☐ Less fish

Size
☐ Bigger fish
☐ Same
☐ Smaller fish

38) If you think there has been a change, what do you believe has caused it?

Notes

39) Village name:____

40) Sex: ☐ Male ☐Female

41) Age: ☐ <20 ☐ 20-40 ☐ 40-60 ☐ 60+