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Coral-reef conservation planning in regions with high resource dependence: integrating lessons from socioeconomic and biodiversity approaches

PhD thesis submitted by
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Abstract

Addressing the current biodiversity crisis is challenging. Recent advances in the field of conservation science have emphasized the need to consider whole social-ecological systems, recognising and accounting for the intricate relationships between nature and people. However, even with these well-justified calls for more multidisciplinary and complex approaches to conservation decision-making, the resources dedicated to conservation are still limited. Limited resources make cost-effective conservation essential, and this calls for planning. Conservation planning, the process of deciding where, when, and how to allocate limited resources to reduce biodiversity loss, is a burgeoning field with worldwide applications in a range of realms and contexts. The past decade saw a boom in systematic approaches to protected-area design, with decision-support tools proposing cost-effective sets of candidate reserves achieving explicit, quantitative conservation objectives (to allocate limited resources more wisely) for the least socioeconomic cost to affected stakeholders (to consider human needs).

One of the foundations for the success of protected areas designed using a systematic approach is relevant, high-quality spatial datasets: on biodiversity to maximise its protection, and on the socioeconomic costs of implementing these protected areas to minimise potential negative impacts on resource-dependent human communities. Mimicking approaches used in the terrestrial and temperate marine realms, and to minimise the difficulty and cost of data collection, planners in coral-reef regions often use coral-reef habitats as proxies for marine biodiversity, and lost fishing opportunities as proxies for socioeconomic costs incurred by resource users. These applications are often based on a number of untested assumptions which, if false, could have serious implications for the actual success of protected areas.

In coral-reef regions where most of the marine ecosystem is used by coastal communities, reconciling conflicting conservation and socioeconomic objectives is particularly challenging. Although the need for compromises on both sides is well known, the extent of trade-offs remains unclear. In Chapter 2, I developed a new method to quantify these compromises in Wallis, Alofi, and Futuna, three small Polynesian islands of varying geomorphologic and socioeconomic contexts, where

threatened coral reefs are critical for subsistence fishing. Using Marxan, a reserve selection algorithm, and following current international habitat conservation guidelines, I designed protected-area systems to protect 20% of each coral-reef habitat type, progressively allowing an increasing proportion of fishing grounds to be reserved. Stronger trade-offs between conservation and socioeconomic objectives appeared when reserves were designed to protect habitats mapped with more detail, and using larger management units. The extent of trade-offs also varied according to local socioeconomic and geomorphologic contexts in the three islands. By demonstrating that the ability to achieve conservation and socioeconomic objectives and extent of compromises needed are largely data and context-dependent, I highlighted the need to question the effect on and relevance to objectives of different biological and socioeconomic data used in local-scale coral-reef conservation planning.

In Chapters 4, 5, and 6, I investigated the implications of using different socioeconomic and biological data based on common assumptions about local-scale coral-reef conservation decisions. I used the Madang Lagoon in Papua New Guinea as a case study because 1) this area, covering 40 km^2, is situated in the Coral Triangle (a known hotspot for coral-reef biodiversity), 2) local coastal communities maintain close relationships with their marine environment, and 3) a large international biodiversity expedition in the region facilitated logistics for my data collection, as well as access to a comprehensive set of data rarely available in such contexts. I detailed data collection methods in Chapter 3.

To minimise impacts on fishers, protected areas are often placed away from the most important fishing grounds. A main assumption is that placing important fishing places for fishers inside protected areas will likely incur high socioeconomic costs. These costs are typically quantified through proxies of current fishing opportunities such as proximity to villages, fishing effort, total catch, or average catch per unit effort. In Chapter 4, I investigated the validity of such proxies as indicators of the importance of fishing places for fishers, thus of potential socioeconomic costs if placed inside a protected area. I mapped fishing grounds and their relative importance defined 1) according to the proxies, and 2) as perceived by fishers themselves, and compared them. I found strong spatial differences between reserves designed to minimise costs based on proxies and the perceptions of fishers. There are two possible explanations, both of which have serious implications for planning. First, fishers' perceptions could incorporate spatiotemporal variations in fisheries patterns that are hard to capture with short-term data collection (commonly done for conservation planning and in this study). Second, each dataset tells planners different things, but both could be valid, meaning that both perspectives will be relevant to different objectives. For instance, proxies based on catch data could help planners achieve objectives related to economics and food security by focusing on maintaining fishing effort and the quantity of resource extracted, while fishers' perceptions could help achieve more social objectives by including fishers' values and preferences.

People access and value their marine environment for more than just fishing, especially in coastal coral-reef areas. For example, coral reefs are also accessed for recreation, spirituality, or their aesthetic value, and harvested for cultural purposes. In Chapter 5, I tested the adequacy of using only the importance of places for fishing as a socioeconomic cost of reserve implementation. My hypothesis was that implementing protected areas not only incurs costs to fishers through constraints on fishing, but also to the broader community through revoking harvest and/or access to places that provide other ecosystem services like recreation, traditional medicine, spirituality. I developed a new approach to map and quantify the perceived importance of different places for community members according to all these benefits, including fishing. Similar approaches exist for extensive terrestrial regions in developed countries, promoting the use of such data in conservation planning. However, none of these approaches explicitly demonstrates how this information can be incorporated into planning. I developed a novel method to do so, and demonstrated that locating reserves to minimise costs to fishers is likely to incur significant costs to the wider community by displacing reserves into areas where access to other benefits is important. This result has major implications, since I am demonstrating that this common approach can provide a false sense of achievement of socioeconomic objectives. Without including information on all benefits derived from coral reefs, there is a clear risk of compromising values of local people and undermining the cooperation needed for reserves to be effective in biodiversity conservation.

In conservation planning, it is often assumed that designing reserve systems that encompass a greater diversity of habitats will incidentally protect a higher number of species. It is also assumed that finer-resolution and more complex data are more representative of "the truth", implying that representation of "true" biodiversity and reduction of "true" socioeconomic cost increases with effort in data collection. In Chapter 6, I tested these two assumptions. To do this, I compiled the most comprehensive empirical dataset available for a single small coral-reef region, in the Madang Lagoon, in Papua New Guinea. To use as biodiversity proxies, I created different habitat maps for the region using satellite imagery and ground-truth data. As a reference biodiversity dataset, I compiled georeferenced species lists for macro-algae, corals, fish, and macro-invertebrates collected by taxonomists in the region. As proxies for socioeconomic costs, I compiled the different datasets developed in Chapter 4. As a reference socioeconomic cost, I used the perceived importance of places for derived ecosystem benefits developed in Chapter 5 because these data gave the most comprehensive picture of reef uses and values to local people. Using only areas for which I had all four datasets, I designed reserve systems that aimed at protecting biodiversity proxies while minimising socioeconomic cost proxies, using all possible combinations of data types. I created the Proxy Effectiveness Index (PEI) to measure the effectiveness of such reserves at protecting reference biodiversity and at reducing reference socioeconomic cost. For biodiversity, I found that using more detailed habitat maps does not necessarily lead to representing more species in candidate reserves and that surrogacy effectiveness was highly variable, suggesting the interaction of numerous other factors. Reducing socioeconomic costs based on catch data performed best to reduce the reference cost compared to coarser proxies. Finally, the most expensive combinations of biodiversity and cost proxies do not necessarily provide the most cost-effective reserve systems (i.e. best reference biodiversity representation for the lowest reference socioeconomic cost). I emphasised that obtaining a robust measure of the cost-effectiveness of reserves based on proxies is difficult because of the global lack of reference datasets, and results will vary with numerous factors inherent to the data used for testing and to the test method itself.

My thesis has a number of implications for biodiversity, conservation planners, and human coastal communities in tropical regions, which I discuss in Chapter 7. The implications arise from the risk that identifying candidate protected areas using inadequate data can lead to a false sense of achievement of both conservation and socioeconomic objectives, ineffectively protecting biodiversity while incurring significant impacts on local communities. First, tropical fisheries are complex and highly variable in space and time, and connections between people and coral reefs go well beyond fishing, highlighting the need for socioeconomic assessments that more comprehensively and accurately reflect the perceptions and values of local communities. Second, planners must carefully consider the social, economic, and even cultural goals of their reserves to influence their choice of proxies for socioeconomic costs. Third, protecting habitats may be a good way to achieve biodiversity protection, but the effectiveness of habitats as surrogates of biodiversity cannot be assumed unless it is tested in a similar geomorphologic and socioeconomic context. Finally, return on investment in data collection does not necessarily increase with the cost of data, so planners should focus on relevance, rather than quantity or level of detail, in gathering important information for conservation.

Outputs directly associated with this thesis

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Glossary

Ecosystem services: Ecosystem services are the benefits people obtain from ecosystems. Ecosystem services include provisioning services such as food and water; regulating services such as the control of climate and disease; supporting services, such as nutrient cycles, and cultural services such as spiritual or recreational benefits.

Habitat: A habitat can be described as a combination of physical, chemical and biological properties providing a particular environment for the establishment of biota. In this thesis, a habitat is defined by its geomorphologic and environmental characteristics (e.g. fringing reef), as opposed to the typical place where a particular species occurs (e.g. dugong habitat).

No go area: No go areas are a type of protected area in which access is fully prohibited. In this thesis, no go areas refer to ICUN Ia category (strict nature reserves).

No-take area: No-take areas (or "no-take zones") are a type of protected area, in which extractive activities (e.g. fishing, hunting, collecting) are prohibited.

Planning region: The planning region is the domain across which conservation actions and areas to implement these actions are considered.

Planning units: Planning units are basic spatial units at which a conservation decision is being made. They constitute spatial subdivisions of the planning region, which can be natural, administrative, or arbitrary.

Protected area: According to the IUCN, a protected area (or "reserve") is a clearly defined geographical space, recognised, dedicated and managed, through legal or other effective means, to achieve the long term conservation of nature with associated ecosystem services and cultural values. In this thesis, protected areas refer to areas under protection meeting the IUCN class I-VI categories

Proxy: Proxies (or "surrogates") refer to variables used instead of a variable of interest. The use of proxy data instead of data on the variable of interest is justified by the lower costs associated with the collection of information related to the proxy, compared to the variable of interest. In conservation planning, proxies of biodiversity and of socioeconomic costs are used to guide the design protected areas because comprehensive information on these variables is often lacking.

Socioeconomic cost: The socioeconomic cost of conservation actions is the negative impact on stakeholders resulting from these actions. Conservation actions can affect socioeconomic attributes such as livelihoods, well-being, social values, and economies. Conservation actions can also provide benefits to stakeholders.

Spatial prioritisation: Spatial prioritisation is the process of defining priority areas for the implementation of specific actions to achieve a given objective for a given cost constraint. When applied to conservation planning and the design of protected areas, spatial prioritisation is also termed "reserve design".

Thematic resolution: The thematic resolution of a habitat map is the number of habitat classes.

Spatial resolution: The spatial resolution of a satellite image refers to the size of pixels used in the creation of this image. The spatial resolution of a map refers to the minimum mappable unit (MMU) found in the map, or smallest feature mapped. The smaller the pixels or MMUs, the higher the spatial resolution.

Stakeholders: Stakeholders include all people who will decide, contribute to, or be affected by conservation actions across a planning region. These include communities, resource users, governmental and non-governmental agencies, scientists, and local experts.

Value: In this thesis, the value of a place or an ecosystem service refers to its (subjective) importance to people, which is not necessarily monetary.

Chapter 1.
General introduction

In this chapter, I introduce the rationales and main concepts used throughout my thesis. I review the relevant literature, and identify urgent research gaps to be addressed. Then, I introduce the broad goal of my thesis, with the three main objectives I aimed to achieve throughout my work. Finally, I outline the structure of my thesis and give a brief summary of each chapter.

1. General introduction

Key challenges in systematic conservation planning

Systematic conservation planning to design cost-effective protected areas

Cost-effective conservation planning is a difficult task. With the rapid loss in global biodiversity, the increased demand for resources (Steffen et al., 2015), and limited dedicated funds (Balmford et al., 2003), the main challenge faced by conservation planners is to adequately protect biodiversity from threatening processes, and sustain it into the future (Margules and Sarkar, 2007) for the lowest cost. The problem is exacerbated by the complexity of the potential costs associated with conservation such as planning costs (e.g. data acquisition, analyses, expert advice), management costs (e.g. staff employment, vehicles or boats, fuel), and socioeconomic costs on resource users (e.g. opportunity costs such as direct loss of food or income related to limitations on access to resources, social impact). The full costs of conservation planning are therefore rarely, if ever, estimated with accuracy. As a consequence, conservation planning is not always cost-effective. In this thesis, "cost-effectiveness" refers to the effectiveness of conservation actions at achieving a given set of conservation objectives at a minimum cost to a given group of stakeholders.

In terrestrial, freshwater, and marine realms, protected areas[1] are a key tool used to protect biodiversity from anthropogenic threats (Lubchenco et al., 2003, Fox et al., 2012). However, these are often implemented on an *ad hoc* basis or in remote, unproductive areas to reduce cost to affected stakeholders[2], and consequently fail to adequately represent and protect biodiversity (Margules and Pressey, 2000). On the other hand, it is well recognised that ignoring the potential negative impacts of protected areas on resource users can lead to conflicts between conservation and

[1] In my thesis, I use the terms "protected areas" and "reserves" interchangeably to refer to areas under protection meeting the IUCN class I-VI categories.
[2] Following Margules and Sarkar (2007), "Stakeholders include all those people who have decision-making powers over a region, all those who will be affected by the conservation plans that are formulated, those with scientific or other expertise about the region and those who may commit resources for conservation planning and implementation."

human needs and low compliance, and therefore ineffective conservation (West et al., 2006, Mascia et al., 2010). Finding strategies designed to deliver biodiversity protection without threatening already vulnerable human livelihoods is a well-known problem for conservation planners but concrete solutions remain limited (Sanderson and Redford, 2003).

Systematic conservation planning has been proposed as an attempt to guide decision-makers towards transparent, equitable, and cost-effective solutions to complex conservation problems. Systematic conservation planning is the process of deciding where, when, and how to allocate limited conservation resources in order to meet specific conservation objectives (Margules and Pressey, 2000, Pressey and Bottrill, 2009, Sarkar and Illoldi-Rangel, 2010). The framework shown in **Figure 1.1** outlines the 11 main stages of systematic conservation plans, and is used worldwide in a variety of contexts and spatial scales. One of the key steps in planning is “spatial prioritisation”. Spatial prioritisation[1] is the stage at which conservation planners use prioritisation tools (usually complex computer algorithms) to choose priority areas to allocate conservation efforts that will help them achieve their goals at the least cost. Spatial prioritisation techniques can be used for a variety of purposes, but where the preferred conservation action is the implementation of protected areas, spatial prioritisation is also termed “reserve design”.

Systematic conservation planning approaches, in theory, allow more cost-effective decisions in the sense that compromises between biodiversity protection and socioeconomic impacts are explicitly addressed (Adams et al., 2010). But for these approaches to be successful, the robustness of three key foundations must be verified: 1. the ability to achieve conservation and socioeconomic objectives; 2. the adequacy of data on biodiversity; and 3. the adequacy of data on the socioeconomic context (**Figure 1.1**). In the following, I review how these three key foundations for the success of protected areas designed using systematic conservation planning are currently accounted for in the broader conservation planning theory and

[1] In my thesis, I use the terms “spatial prioritisation” and “reserve design” or “protected area design” interchangeably to refer to the action of choosing the areas to protect in order to achieve conservation (and socioeconomic) objectives.

practice. I also describe the associated challenges and how they are currently addressed.

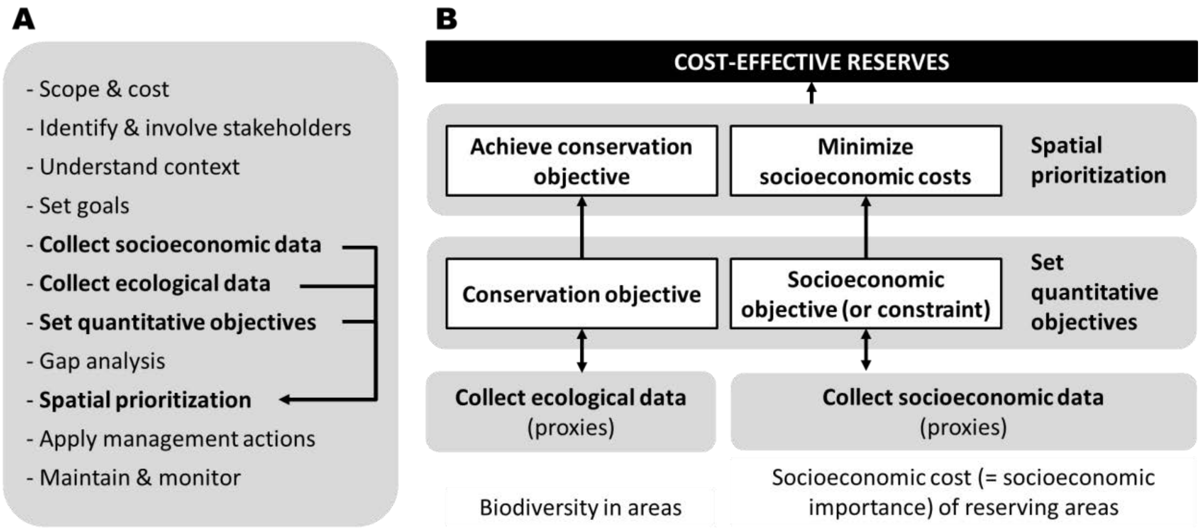


Figure 1.1. Systematic conservation planning framework: A) The 11 stages of systematic conservation planning, not necessarily followed in sequence, simplified and adapted from Pressey and Bottrill (2009). B) Conceptualisation of how data and objectives can shape the cost-effectiveness of reserves.

Achieving conservation and socioeconomic objectives

Quantitative conservation objectives (or targets) are a key tool used in policy at international, regional, national and local levels (e. g. Natural Resource Management Ministerial Council 2010, IUCN World Parks Congress, 2005, UNEP/CBD/COP/10/X/2, 2010). As a result, many conservation plans worldwide attempt to meet recommended objectives in various planning contexts and across different spatial extents, with different interpretations of the same guidelines (e.g. Cowling et al., 2003, Fernandes, 2005, Lombard et al., 2008, Osmond et al., 2010). Additionally, decision-support tools for systematic conservation planning are all based on such objectives. Although it is clear that quantitative objectives can help planners and stakeholders to negotiate conservation decisions and measure their progress, "target-based" conservation approaches are also widely criticised (Agardy et al., 2003a, Balmford et al., 2005, Tear et al., 2005, Wood, 2011). In fact, quantitative objectives are often set arbitrarily, based on the amount of area to be protected (typically, a certain percentage of a habitat type, or of a region or jurisdiction are targeted for protection), rather than objectives reflecting the relative need of habitats and species for protection to achieve the persistence of biodiversity. Most commonly reported limitations of protected areas designed using such objectives include possible perverse outcomes, inadequate, inflexible, and unachievable conservation plans, as well as the inability to consider complex ecological or socioeconomic factors (Carwardine et al., 2009).

Socioeconomic objectives are increasingly being considered in conservation. This trend follows the need to account for human communities and their relationships with the natural environment now recognised in policy (Millennium Ecosystem Assessment (MEA), 2005, UNEP/CBD/COP/10/X/2, 2010). However, protected areas can deliver mixed outcomes for human communities, especially in rural settings: because people's well-being is linked to their natural environment, conservation actions can affect them positively and negatively (Scherl et al., 2004). Consequently, socioeconomic factors tend to be incorporated into conservation plans in different ways. On one hand, terrestrial and marine protected areas are increasingly advocated to increase human well-being for associated communities through maintaining and provisioning ecosystem services and benefits (Lubchenco et al., 2003, Adams et al., 2004, Fox et al., 2012, Sala et al., 2013). As a result, broad conservation goals have shifted from strictly protecting biodiversity towards integrating socioeconomic objectives such as poverty alleviation (Pelser et al., 2013, Gurney, 2014) or maintaining ecosystem services to sustain livelihoods (Fernandes, 2005). However, such goals are rarely incorporated explicitly (e.g. as quantitative objectives) in conservation planning, and when applicable, rarely prioritised to the same extent as conservation objectives. On the other hand, because biodiversity protection remains the core goal of protected areas, and potential negative effects on rural livelihoods and resource-dependent human communities can be expected, socioeconomic factors tend to be accounted for as a constraint (cost) in planning, rather than as quantitative objectives (Gurney et al., 2015). Additionally, most decision-support tools are designed to incorporate socioeconomic information as a constraint to the achievement of conservation objectives (more details on this matter below).

While effective conservation depends on setting adequate and explicit conservation and socioeconomic objectives based on scientific evidence and incorporating them effectively into conservation policy and reserve design, doing so does not guarantee their effective implementation, and the ability of resulting reserves to achieve these objectives. Achieving conservation and socioeconomic objectives in practice can also depend on the following: the data used to design protected areas that are meant to address these objectives (and to measure achievement of objectives) and, indirectly, on whether and how inevitable trade-offs between conservation and people's needs are resolved.

Because reserves designed using systematic conservation planning use specific reserve selection algorithms to answer a set of objectives, the outputs of spatial prioritisation (including the ability to meet objectives, at least on paper) are only as good as the data used as an input. The sensitivity of spatial prioritisation outputs to different types of data on biodiversity (Andréfouët et al., 2012b) or socioeconomic costs (Richardson et al., 2006), as well as the spatial resolution at which planning occurs (size of planning units) or all the above-mentioned (e. g. Leslie et al., 2003, Stewart et al., 2003, Warman et al., 2004) has been studied in a range of contexts. However, this sensitivity is rarely tested or accounted for in practice. This is important because, for a given set of objectives, different pictures of achievement can be obtained if different data are used, and some of these pictures will be misleading in terms of actual achievements on the ground.

In conservation practice, there are almost always trade-offs between conservation and socioeconomic objectives (Halpern et al., 2013), but decisions on acceptable compromises are often subjective and can be driven largely by the political context. This is because measuring and mapping the extent of compromises needed to provide a transparent basis for communication and negotiation is challenging (McShane et al., 2011). Trade-offs between conservation and human needs (e.g. resource extraction, development) in a systematic conservation planning context have been clearly identified (e.g. Stewart and Possingham, 2005, Schneider et al., 2011, Faleiro and Loyola, 2013) but approaches to systematically measure the strength of these trade-offs and the extent of compromises required on both sides are still lacking (but see Polasky et al., 2008, White et al., 2012, Lester et al., 2013). Part of the difficulty in presenting trade-offs objectively is that little is known on their sensitivity to factors such as the data and planning unit sizes used to design reserves, or the social-ecological context of planning regions.

Collecting adequate data on biodiversity for conservation planning

In this section, I introduce the types of biodiversity data typically used in conservation planning, and describe the well-known associated challenges for the discipline.

Overall biodiversity from genes to ecosystems is impossible to quantify comprehensively. The best possible data one can collect to assess overall biodiversity are data on true surrogates[1] (or proxies) of biodiversity. True surrogates of biodiversity are quantifiable variables that are supposed to capture biodiversity as a whole (Sarkar and Margules, 2002). The controversial nature of true surrogates themselves (Sarkar and Margules, 2002) makes choices about them difficult. Indeed, because there is no way to grasp biodiversity as a whole, finding an ideal true surrogate is challenging. Possible true surrogates of biodiversity include data on genetic diversity, functional diversity, or species diversity. Species composition is the most commonly used true surrogate of biodiversity because species are the best-defined category above the genotype (Sarkar and Margules, 2002). However, comprehensive data on species, for instance in the form of species lists or species abundances, are also difficult, time-consuming, and expensive to collect. In the context of conservation planning, biodiversity data are always limited, sometimes strongly: species data are either irregularly distributed over study areas because collection is biased towards the most accessible areas, and/or incomplete because inventories are biased toward selected taxa (e.g. well known taxa, those easiest to record, or particular taxa of interest for the taxonomists involved).

To overcome the limitations associated with the acquisition of comprehensive data on true surrogates of biodiversity such as species diversity, estimator surrogates can be used. Unlike true surrogates, estimator surrogates are in principle easily-collected, quantifiable variables that approximate the true surrogates of biodiversity. This implies that the relationship between the estimator surrogate

[1] In my thesis, I use the terms "surrogates" and "proxies" interchangeably to refer to information on another feature than the primary feature of interest (biodiversity, or socioeconomic costs).

and the true surrogate is well known. In principle, using data on an estimator surrogate of biodiversity instead of the ideal of the true surrogate should allow a considerable reduction in logistics, time, and/or monetary costs of data acquisition (Rodrigues and Brooks, 2007, Grantham et al., 2010). However, by definition, biodiversity is less well represented or predicted by estimator surrogates than by comprehensive data on true surrogates. Examples of common estimator surrogates include selected taxa (e.g. geographic population, genus, family, order), species, habitats, and other environmental variables to represent species diversity.

Historically, habitats[1] have often been used as proxies of biodiversity in land-based conservation. This is mainly backed up by two rarely tested underlying assumptions (but see Lombard et al., 2003). First, habitats are assumed to reflect at least some of the important factors that determine species' distributions, such as temperature, rainfall, aspect, and drainage. Accordingly, the likelihood of occurrence of at least some species is related to specific habitats. Second, higher habitat diversity increases species diversity (Grantham et al., 2010). Although these relationships and theories have been widely tested, there is yet no general consensus (e.g. Rushton et al., 2004, Tews et al., 2004).

Collecting adequate data on socioeconomic costs for conservation planning

In this section, I introduce the types of socioeconomic data typically used in conservation planning in all realms, and describe the associated challenges for the discipline.

The importance of integrating socio-economic information into conservation plans is now widely accepted among conservationists (Klein et al., 2008). Despite claims of numerous long-term benefits of reserves for local communities and other stakeholders (e.g. Gell and Roberts, 2003, Russ et al., 2004b, Mascia et al., 2010, McClanahan, 2010), it is increasingly acknowledged that restrictions on human activities can also compromise access to ecosystem services and compromise livelihoods and well-being (Adams et al., 2004, West et al., 2006, Adams and Hutton, 2007, Cinner et al., 2014). This realisation led to the development of

[1] A habitat can be described as a combination of physical, chemical and biological properties providing a particular environment for the establishment of biota. In this thesis, a habitat is defined by its geomorphologic and environmental characteristics (e.g. fringing reef), as opposed to the typical place where a particular species occurs (e.g. dugong habitat).

approaches to systematic conservation planning that limit the socio-economic costs to stakeholders (Naidoo et al., 2006). However, effective approaches to assess and incorporate relevant social and economic data into conservation planning are still developing (Ban et al., 2013, Kittinger et al., 2014, Le Cornu et al., 2014).

For socioeconomic data to be used in systematic conservation planning, the information must be spatially-explicit and, ideally, quantitative. Typically, to estimate the potential socioeconomic impacts of protected areas on stakeholders, conservation planners collect or model spatial data on the economic value of all areas of the region of interest. For example, costs of protecting land to economically-involved land users can be measured as forgone timber harvest (Schröter et al., 2014), or forgone benefits from alternative land uses (Adams et al., 2010). Recent advances in the field reveal that considering stakeholders as distinct groups, each benefiting differently from the ecosystem, as opposed to a unique homogenous group, leads to more equitable and cost-efficient conservation plans (Adams et al., 2010, Adams et al., 2011). However, the potential costs of protected areas to stakeholders other than extractive resource users are rarely considered explicitly in spatial prioritisation processes. Additionally, recent research shows that distinct stakeholder groups may have similar “worldviews” or “values” associated with coastal environments and the way they use it, emphasizing the use of a value-based approach, rather than a stakeholder-based approach (Voyer et al., 2015).

In marine conservation planning, most published studies focus on the opportunity costs for fisheries (i.e. the forgone value of areas to be protected, for fisheries) (Ban and Klein, 2009). A rapid review of the recent literature reveals that little has changed in the past few years (**Table 1.1**). A few attempts were made to include a wider range of stakeholders and potential negative socioeconomic impacts into marine planning (Klein et al., 2008, Ruiz-Frau et al., 2015), but the debate over which proxies, if any, are best to use is still unresolved (Weeks et al., 2010, Deas et al., 2014).

Table 1.1. Examples of spatially-explicit socioeconomic data used to define cost layers for marine systematic conservation planning.

Areas avoided for conservation	Proxy	Unit(s)	Realm	Study area (country)	Reference
Important areas for fishing	Commercial trawling fishing effort.	total annual hours trawled per 0.01° grid cell, averaged over 9 years, and weighted so that the contribution of more recent trawling was greatest	Tropical coral reefs	Australia	Sutcliffe et al. (2015)
Important areas for fishing	Nineteen opportunity cost layers based on fine-scale fishery catch maps considering: i) total catches, ii) target fish families, iii) local marine tenure, and iv) gear type.	biomass catch loss	Tropical coral reefs	New Caledonia	Deas et al. (2014)
Important areas for fishing	Fishing pressure (model of anthropogenic drivers of marine change based on global fisheries catches, tuna purse seine catch data, and data on coastal fisheries and population density)	metric tons/km^2/yr	Tropical coral reefs	Madagascar	Allnutt et al. (2012)
Important areas for fishing, mariculture, and areas with poor reef condition	Fishing pressure (fixed fishing structures such as FADs/fishing cages/fishing shelters/fixed fish traps, fishing grounds)	occurrences of activity	Tropical coral reefs	Indonesia	Agostini et al. (2012)
	mariculture (seaweed farming, pearl farming)	Occurrence of activity			
	reef condition	% dead coral and rubble			
Important areas for fishing	Fishing pressure (accessibility)	None (lower value for less accessible areas: inshore and closer to shipping lanes)	Temperate	France and UK (English Channel)	Delavenne et al. (2012)
	Fishing pressure	Number of fishing boats weighted by distance to port			
Important areas for	Intensity of use (overlap between uses: high,	Intensity scale	Temperate	Chile	Rojas-Nazar et al.

Areas avoided for conservation	Proxy	Unit(s)	Realm	Study area (country)	Reference
fishing, tourism, poaching	medium, low intensity)				(2012)
Important areas for fishing	Fishing pressure	catch per unit effort in catch/person/h/m^2	Tropical coral reefs	Fiji	Adams et al. (2011)
	Economic value	fishing opportunity in Fiji Dollars per 2500 m^2			
Important areas for fishing and tourism	Fishing pressure	distance from port, wind exposure	Temperate	Greece	Giakoumi et al. (2011)
	Tourism intensity	bed availability on islands			
Important areas for fishing	Fishing pressure	Mixed (population size, population density, number of fishing vessels, number of fishers)	Tropical coral reefs	Philippines	Weeks et al. (2010)
Important areas for fishing	Fishing pressure	Mixed (artisanal catches, population density)	Tropical coral reefs	Philippines	Ban et al. (2009)
Important areas for fishing, industry, ports, shipping	Proximity to towns, large river mouths with industry, ports, shipping channels. Also considered community interest in marine conservation, cultural sites, existing MPAs, dive sites, special and unique areas, areas recommended as good candidates for conservation, villages visited during awareness campaign.	None (relative cost for each proxy)	Tropical coral reefs	Papua New Guinea	Green et al. (2009)
Important areas for fishing	Fishing pressure	Mean annual catch data in kg/km^2	Tropical coral reefs	Australia	Game et al. (2008)
Important areas for recreational and commercial fishing, areas far from	Recreational fishing effort	Number of fishing trips	Temperate	United States of America	Klein et al. (2008)
	Commercial fishing effort	Relative importance of fishing grounds for 19 fisheries			
	Occurrence of scientific	Adjacency of			

Areas avoided for conservation	Proxy	Unit(s)	Realm	Study area (country)	Reference
education and research, institutions, population centres and terrestrial parks.	monitoring sites, research institutions, educational institutions, population centres, and terrestrial parks	planning units			
Important areas for fishing	Economic value	First-sale value of fish and shellfish, fishers interviews with participatory mapping	Temperate	Wales	Richardson et al. (2006)
Important areas for fishing	Fishing pressure	Total rock lobster catch in kg/km^2	Temperate	Australia	Stewart and Possingham (2005)
Important areas for fishing	Fishing pressure	Density of fishing boats	Temperate	Gulf of California	Sala et al. (2002)

The case of local coral-reef planning: research gaps

In this section, I review the above-mentioned key foundations for the success of protected areas designed with systematic conservation planning (i.e. setting objectives, and identifying adequate data on biodiversity and socioeconomic costs) in the particular context of local coral-reef conservation planning.

In coral-reef environments, the difficulty of protecting biodiversity in a cost-effective manner is enhanced by the strong links between biodiversity itself and the people who depend on it, particularly in low-income countries where alternatives to resource-dependent livelihoods are scarce. Coral reefs are one of the most diverse ecosystems in the world (Bellwood and Hughes, 2001) and provide a number of ecosystem goods and services to tens of millions of people in more than 100 countries (Salvat, 1992, Moberg and Folke, 1999). On the one hand, they are under increasing pressure from various global and local threats (Bellwood et al., 2004) and require rapid and efficient protection. On the other hand, they are a critical source of food and income for often poor coastal communities (Whittingham et al., 2003). Therefore, the conservation challenge here lies in a hierarchy of compromises, exacerbated by the specific context of most coral-reef regions:

1. Compromises between the need to protect biodiversity adequately and the insufficient dedicated funds for data acquisition, planning and management.
2. Compromises between the urgent need to protect a coral-reef biodiversity adequately, which underpins sustainable livelihoods in the long-run, and the potential direct negative impacts of implementing marine reserves for local resource-dependent communities;

Critical to addressing both compromises are the needs to:

- formulate conservation and socioeconomic objectives that are context-relevant;
- find adequate proxies of coral-reef biodiversity that are relatively cheap, quick, and easy to collect (i.e. cost-effective proxies); and

- identify and collect adequate socioeconomic data to serve as proxies of socioeconomic costs of protected areas, that are relatively cheap, quick, and easy to collect.

In the next subsections, I follow the same sequence (i.e. setting objectives, and identifying adequate data on biodiversity and socioeconomic costs) to describe how these issues are currently addressed in coral-reef conservation planning when applicable. I identify relevant research gaps at the end of subsections, on which I based the broad objectives of my thesis.

Setting adequate objectives

The particular ecological, social, and political context of most coral reef countries is likely to require different approaches than common top-down conservation planning. In the Coral Triangle for example, there is a recognised mismatch of scales between implementation of conservation actions at the regional and at the local level (Mills et al., 2010). Decisions made for regional planning do not adequately inform local-scale planning, and vice-versa. As a result, regional designs fail to guide conservation at local scales, and local actions rarely constitute ecologically functional systems. Reasons include the limited influence over marine resource management of central governments, the inadequacy of data for regional planning, and the constraints on conservation opportunities by social, economic, and political complexities such as high dependence on marine resources, or unresolved boundaries of customary tenure. This mismatch of scales is likely to translate into a mismatch in objectives as well.

In coral-reef countries, local conservation objectives are often dictated by non-governmental agencies rather than governments, mainly because funding for conservation is lacking at the government level. In some cases, objectives are aligned with international commitments such as the Aichi targets and set by initiatives such as the Coral Triangle Initiative (CTI), which coordinates coral-reef conservation actions at a regional level. Member countries are called to aim for the representation of 20% of each major marine and coastal habitat type within the region (e.g., coral reefs, seagrass beds, mangroves, beach forests, wetland areas and marine/ offshore habitat) in strictly protected “no-take replenishment zones” (to ensure long-term sustainable supplies of fisheries). Objectives for marine protected areas in the CTI clearly incorporate biodiversity and socioeconomic goals (Coral Triangle Initiative, 2008), but socioeconomic goals rarely translate into quantitative objectives. Some countries already translated these objectives into action (e.g. MECM/MFMR, 2010), but like the Aichi Targets, how such targets should be interpreted by each country, or even at local-scale remains unclear. For example, it may not be feasible to reserve 20% of coral reefs in a local system of no-take areas because of the strong dependence by communities on reef resources, yet, other management actions could be undertaken to achieve sustainability of the system. Practical solutions to achieve these ambitious targets, while recognising the trade-offs between the high resource-dependence of coastal communities in coral-reef countries and the urge to implement more protected areas, are lacking.

Additionally, the influence of different types of data on biodiversity and socioeconomic costs, as well as the spatial resolution of management actions (size of planning units) on the ability to achieve objectives have also been investigated in coral reefs contexts (Van Wynsberge et al., 2012, Deas et al., 2014, Van Wynsberge et al., 2015). However, nothing is known on the sensitivity to these factors of the above-mentioned trade-offs and ability to achieve conservation and socioeconomic objectives.

From this subsection on objectives, I identified the following research gap:

RESEARCH GAP 1: Research is needed to investigate the adequacy of international guidelines and policy targets, and the influence of context and data in achieving conservation and socioeconomic objectives.

Finding adequate proxies of coral-reef biodiversity

With the ever-increasing pressures on coral-reef environments, collecting sound, comprehensive data on biodiversity for conservation planning is a daunting, yet essential task. In most countries with coral reefs, constraints on logistics, time, and money for conservation are exacerbated by the remoteness of most areas, the scale and urgency of the task given competing responsibilities, and insufficient funds for biodiversity and ecosystem conservation, respectively. The lack of capacity and technical expertise is also a major constraint. In this context, finding and using efficient estimator surrogates of biodiversity appears as a promising solution to increase our knowledge of coral-reef ecosystems, and thus improve their conservation and the management of associated resources.

The use of proxies of coral reef biodiversity is largely drawn from land-based approaches, as described above, and similar challenges are identified. The effectiveness of several estimator surrogates for aspects of coral-reef biodiversity (true surrogates) has been assessed in the recent coral-reef literature (see **Table 1.2** for studies that explicitly use the terminology “surrogate”). The proxies tested have included environmental variables (e.g. benthic habitats, geomorphic features, depth classes), species assemblages, higher-level taxa (e.g. family or order instead of species), or specific taxa. However, various analyses of a range of variables have given a wide diversity of results, sometimes even contradictory. Failure to control for the following factors, identified for all realms (Margules and Pressey, 2000, Grantham et al., 2010), and more specifically for coral-reef fish (Mellin et al., 2010), preclude any kind of generalisation on surrogates effectiveness:

1. The variety of variables of interest that can be investigated is wide, both for the surrogate and the target. For instance, looking at studies assessing the potential of coral-reef habitats as surrogates of coral-reef fish can first appear confusing, the big picture being basically: habitats are strong, moderate, or poor surrogates for fish species. However, a deeper look reveals that surrogates investigated have included landscapes, seabed, heterogeneity, a mix between geomorphology and benthic cover, while fish (the targets) can be looked at using species richness, abundance, biomass, assemblages (species composition), community structure, or spatial distributions. Bias in data collection for both the surrogate and target also influences effectiveness (e.g. Van Wynsberge et al., 2012).
2. There are as many measures of surrogacy effectiveness as there are surrogate uses. For instance, Beger et al. (2007) found a strong congruence between coral and fish species, but fish are not represented well in a network of protected areas when corals are used to select priority sites, and *vice versa*. Conversely, Dalleau et al. (2010) revealed a weak congruence between different habitat maps and various taxa, but some of the maps performed well when used in a

conservation scenario to represent specific taxa. Similar inconsistencies are found in terrestrial, marine, and freshwater realms (Favreau et al., 2006, Rodrigues and Brooks, 2007, Grantham et al., 2010). A partial solution to confusion and misleading recommendations is to distinguish between pattern-based surrogates, which are used in fundamental ecology mainly to predict the spatial distribution of biodiversity, and selection-based surrogates, which are used in conservation science mainly to protect biodiversity through specific conservation scenarios (Andréfouët et al., 2012a). An efficient pattern-based surrogate will allow accurate prediction of the spatial distribution of the targeted features supposed to be aligned with the surrogate, whereas an efficient selection-based surrogate will allow a good representation of the targeted features in a set of reserves built according to a specific conservation scenario. Effective pattern-based surrogates can be "poor" selection-based surrogates and *vice versa* because they are tested for their ability to provide different kinds of information on biodiversity. Similarly, the effectiveness of surrogates can appear different when different pattern-based or selection-based tests are applied.

3. The variety of spatial extents over which the relationships are studied ranges from very small transects of a few hundreds of metres to whole regions of tens of thousands of square kilometres. The effectiveness of habitats as biological surrogates of a specific target, assessed with a specific method, can show a wide variety of results partly because the combination of functions and processes that link species to their habitats and environment themselves vary between and within spatial scales.
4. Study regions all differ by their biogeographical history and previous conservation and extractive uses. These histories can influence the results of surrogacy tests even when all other factors are held constant. The dynamic nature of marine systems is itself likely to affect the effectiveness of surrogates through space and time. Hence results may be context-specific.

Following the model of conservation planning based on land classes, an increasing number of conservation projects aim at using benthic habitats to help protect marine biodiversity, and more specifically coral-reef biodiversity. Regional action plans such as within the Coral Triangle (Coral Triangle Initiative, 2008) or the

Caribbean (Wilkinson, 2008) for example, based their conservation objectives on coral-reef associated habitat. Launched in 2006, the Micronesia Challenge aimed at conserving a representative fraction of near-shore marine resources (The Micronesia Challenge, 2006). National plans, such as in the USA, aim at protecting a representative fraction of all coral reefs and associated habitat types in each major island group and Florida (United States Coral Reef Task Force, 2000). France's 2011 National Plan of Action aims to protect a certain percentage of marine and coastal habitats by 2020 (Ministère de l'Écologie du Développement durable des Transports et du Logement, 2011). In Choiseul Island in Solomon Islands, the local conservation strategy aims to protect a representative portion of the original extent of each terrestrial and marine ecosystem type (Lipsett-Moore et al., 2010). To build a resilient network of marine protected areas in Kimbe Bay, Papua New Guinea, The Nature Conservancy targeted the representation and replication of major shallow and deep habitats (Green et al., 2009).

While the links between land features and subsets of terrestrial biodiversity have been better studied than associations between coral-reef habitats and their inhabitants, both relationships are still poorly understood for practical conservation design. The complexity of coral-reef ecosystems themselves, the remoteness of these marine environments, the different possible definitions of a coral-reef habitat, and the various methods available for collection of both habitat and biodiversity data are only a few factors making these studies difficult (Knudby et al., 2007). There have been several attempts in the scientific literature to investigate the surrogacy potential of coral-reef habitats for coral-reef biodiversity or subsets of biodiversity (**Table 1.2**), but results differ and are even sometimes contradictory for the same reasons as discussed above. Coral-reef habitats and biodiversity can be described in many different ways and at a variety of spatial and thematic resolutions, and the best methods to measure surrogacy effectiveness are still debated. This highlights the need for more research to first clarify the possibilities in terms of habitat data, and ultimately verify both the ecological and analytical relevance of the use of habitats as surrogates of coral-reef biodiversity in systematic conservation planning.

There is a clear research gap identified above regarding the ability of habitat maps to serve as a cost-effective proxy of biodiversity. In this thesis, this gap is formulated in a broader context of testing the cost-effectiveness of combinations of biodiversity and socioeconomic proxies, and therefore the statement of the research gap appears at the end of the next section. Additionally, since habitat protection is recommended in international policy (UNEP/CBD/COP/10/X/2, 2010), I use habitats as a proxy of biodiversity throughout for the purpose of my analyses, acknowledging (as above) the limitations of this approach.

Table 1.2. Examples of assessments of the effectiveness of proxies (surrogates) of coral-reef biodiversity as addressed in the literature.

estimator surrogate		true surrogate	Test type	Measure of surrogate effectiveness	Conclusions on surrogacy effectiveness	References
Species	fish or coral coral fish	fish + coral fish coral	P[1]	Congruence analysis (Pearson correlation coefficients)	There was not a close correlation between fish and coral assemblages, at least in terms of presence–absence data. Coral-rich sites did not necessarily harbour a maximum of fish species and vice versa.	Beger et al. (2003)
			S[2]	Species-area accumulation curves	Complementarity-based methods always achieved higher accumulative species richness than the other methods, regardless of whether the focus was on fishes, corals, or both taxa combined. However, the same selection procedures differed in their ability to encompass the biotic diversity of fishes and corals. This is mainly due to differences among taxa in distribution patterns and responses to environmental gradients.	Beger et al. (2003)
	coral reef species richness	fish, corals, molluscs	P	Congruence analysis (Pearson correlation coefficients).	Strong cross-taxon congruence between corals and fish. All but one region showed significant but weak cross-taxon congruence between molluscs and corals.	Beger et al. (2007)
			S	Species richness discrepancy	None of the marine faunal taxonomic groups were suitable conservation representation surrogates for the other taxonomic groups in every region.	Beger et al. (2007)
	nematode family copepod family	nematode genera copepod genera	P	Non-metric multidimensional scaling, two-way crossed analysis of similarities, similarity of percentages, rarefaction curves.	This study tested higher-taxon surrogacy for copepods and nematodes. For both groups, family-level identifications seem to be sufficient to analyse the major trends in diversity.	De Troch et al. (2008)

[1] Assessment of pattern-based surrogates (i.e. estimator surrogates that should predict the spatial distribution or patterns of the feature of interest, true surrogate)
[2] Assessment of election-based surrogates (i.e. estimator surrogates that, if used as an input in a reserve selection scenario, should allow a good representation of the feature of interest, true surrogate, within the final reserve network)

estimator	surrogate	true surrogate	Test type	Measure of surrogate effectiveness	Conclusions on surrogacy effectiveness	References
Species	species biomass for 11 phyla	species biomass for 11 phyla	P	Similarity of clusterings between pairs of taxonomic groups.	No taxon was a good surrogate for any other taxon, regardless of how they were grouped taxonomically. Removing rare species did not affect overall surrogate performance. Overall, surrogate performance increased as the number of clusters decreased; however, taxa that performed best as surrogates changed as the number of clusters changed. Importantly, the most readily available data (e.g., fish) were not good surrogates for any other taxon.	(Sutcliffe et al., 2012)
Habitats	Fine-scale seabed habitats along with their broader-scale landscape setting (zones)	fish species	P	Canonical correlation analysis, multiple correlation analysis.	Habitat composition and configuration across scales were strong predictors of assemblage structure and species distributions.	Anderson et al. (2009)
	habitat characteristic s	species distributions	P	Canonical correlation analysis, examination of densities of fish per patch types.	Composition, complexity, and configuration of seafloor at multiple scales predicted assemblage structure and species distributions.	Anderson & Yoklavich (2007)
	coral reef habitats	algae, coral, fish, invertebrates species	P	Congruence analysis (Pearson correlation, Kendall correlation, correlation of dissimilarities, Mantel test).	Richness congruence was low between habitat and species richness. Composition similarities suggested that, in some cases, species and habitat assemblages were linked. Taxon and complexity of habitat map had a significant effect on the relationship between species composition and habitat composition, but the size of neighbourhood in which habitats were considered did not.	Dalleau et al. (2010)

estimator	surrogate	true surrogate	Test type	Measure of surrogate effectiveness	Conclusions on surrogacy effectiveness	References
Habitats	coral reef habitats	algae, coral, fish, invertebrates species	S	Species-area accumulation curves	Use of the most detailed benthic habitat maps always improved species representation over random choice, irrespective of changes in spatial scales. Some habitat maps were efficient surrogates of algae, corals, and commercial fish species at specific sizes of neighbourhood in which habitats were considered. Habitats were poor surrogates for invertebrate species.	Dalleau et al. (2010)
	rugosity (itself a pattern-based surrogate of hard-bottom habitat)	reef fish diversity	S, P	This study focused on the effectiveness of rugosity as a pattern-based surrogate for hard-bottom habitats, themselves known to be a good pattern-based surrogate of biodiversity. The authors proposed to use rugosity as a selection-based or pattern-based surrogate of biodiversity for conservation planning but did not test its effectiveness.	Model based on rugosity correctly predicted areas of both hard-bottom and non-hard-bottom habitat. Offers regional marine resource planners in both developing and developed countries a regional proxy for hard-bottom habitat and an initial indicator of marine biodiversity. This dataset could be used in site prioritisation algorithms.	Dunn & Halpin (2009)
	tropical coastal habitat heterogeneity	fish community structure, grazing, and fisheries value	P	One-way crossed analysis of similarities (ANOSIM), nested ANOSIM and nested ANOVA, Bartlett's test, various metrics.	This study highlighted significant inter-habitat variation in fish communities and intra-habitat variation in community structure, ecological function, and grazing intensity among islands and, to a lesser degree, among reefs around the same island. The correlation between number of species in a habitat and the magnitude of inter-island variation in community structure may be usefully incorporated , along with other economic, social, scientific, and feasibility considerations, within selection algorithms for sitting marine reserves.	Harborne et al. (2008)

estimator	surrogate	true surrogate	Test type	Measure of surrogate effectiveness	Conclusions on surrogacy effectiveness	References
Habitats	habitat	reef fish diversity	P	Partial Mantel test, simple Mantel test, correlogram, single-factor analysis of similarities (ANOSIM).	Overall, the majority of differences in fish assemblage composition (species richness and abundance) could be explained by both habitat type and geographical distance. Resolution of the baseline data did affect the efficacy of the habitat classes in predicting fish assemblage structure.	Lindsay et al. (2008)
	five key factors extracted from the coarse Habitat Classification System (HCS)	coral reef fish ('surrogates of coral reef biodiversity')	P	Multivariate methods: non-metric multi-dimensional-scaling (nMDS) ordinations, multivariate correlation (BIOENV).	The influence of distance from shore was clearly a key driver of patterns of reef fish assemblages. Dominant benthos was also important, but only insofar as inshore habitats were likely to be dominated by macroalgae.	Malcolm et al. (2010)
	coral reef habitats	species, ecological functions, and ecosystem services	S	Species accumulation index of surrogacy value, SAI (Ferrier and Watson, 1997, Ferrier, 2002).	Ecosystem processes and services proved to be ineffective surrogates for species-level information. Habitat selections designed to represent benthic species, fish species, or fish functional classes all proved to be highly effective at representing ecosystem processes. Benthic species were also an effective surrogate of ecosystem services, but fish-based surrogates were only moderately effective.	Mumby et al. (2008)
	substrate rugosity	reef fish assemblages	P	Non parametric Spearman rho correlation coefficient, least-squares simple linear regression.	Rugosity, measured using a well-established field method, had a strong association with all measures of the fish community (abundance, richness, and biomass).	Wedding et al.(2008)

Finding adequate proxies of socioeconomic costs to human coastal communities in resource-dependent regions

In coral-reef conservation planning, socioeconomic costs are incorporated following approaches similar to conservation planning in other marine environments (**Table 1.1**): conservation planners in coral-reef regions have commonly considered fishers as the only or main stakeholder groups potentially affected by protected areas. A major assumption is that maintaining access to areas of economic value or/and of value for food security will ensure minimum socioeconomic impacts on the broader community. The costs of losing access to coral reef areas are also often measured strictly in economic terms, ignoring non-monetary costs such as social impacts. However, in most countries with extensive coral reefs, coastal communities are strongly connected to their local marine environment for their livelihoods and well-being, gaining more from it than just fish for food or income. This means, even where fishing is perceived as the most important activity, that fishers are likely not the only stakeholders who can be negatively affected by protected areas. Another key point is that, like the case of biodiversity proxies, there are many ways to assess or measure costs to a given group of stakeholders such as fishers. Several studies have used socioeconomic cost layers in coral-reef conservation planning exercises, but only a few compared different proxies of socioeconomic costs in this context (e.g. Weeks et al., 2010, Deas et al., 2014).

From this subsection on socioeconomic proxies, I identified the following research gap:

RESEARCH GAP 2: Research is needed to question the adequacy of commonly used proxies of fishing opportunity costs in coral-reef regions where people depend strongly on a variety of ecosystem services and benefits.

From the two above subsections on biodiversity and socioeconomic proxies, and because protected areas designed with systematic conservation planning typically aim for biodiversity representation for a minimum cost, I identified a third research gap:

RESEARCH GAP 3: Research is needed to test the ability of marine protected areas designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.

Thesis goal and objectives

The main goal of my thesis is to improve our understanding of the advantages and limitations of proxies of biodiversity and socioeconomic costs commonly used in conservation planning, with the intention of improving local-scale planning in resource-dependent coral-reef regions. Specifically, I investigate the three research gaps identified above, which make up my three broad research objectives. As objective 1, I aim to investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives. As objective 2, I aim to test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders. As objective 3, I aim to test the ability of marine protected areas designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.

Thesis outline and structure

This thesis includes seven chapters in total, including this general introduction (Chapter 1), four data chapters (Chapters 2, 4, 5, 6) formatted for publication in peer-reviewed journals, a chapter dedicated to collection of data (Chapter 3), to avoid duplication of methods in three of the data chapters, and a general discussion (Chapter 7). **Figure 1.2** shows the overall thesis structure. Each data chapter was modified from the format of the relevant publications to avoid redundancy and ensure a homogeneous terminology throughout the thesis.

To investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives (objective 1), I measure the extent of trade-offs between habitat-conservation objectives based on international policy targets, and local socioeconomic fisheries objectives for the three small Pacific islands of Wallis, Alofi and Futuna (Chapter 2). I assess whether different types of habitat maps, the overlaps between habitats and fishing grounds, and different sizes of planning units influence the achievement of objectives for each island.

In the next section of my thesis, I focus on the Madang Lagoon, Papua New Guinea. This region has a high diversity in species and habitats, with local coastal human communities reliant on ecosystem services and benefits for their livelihoods, making it an ideal case study to achieve objectives 2 and 3. I collected the necessary data (Chapter 3) for the three subsequent chapters with the logistic support of a large international biodiversity expedition occurring in the region, through which I also acquired additional data collected by others involved in the expedition.

I test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders (objective 2). To do this, I compare the spatial distribution of commonly-used socioeconomic costs such as proximity to landing sites, fishing effort, or catch per unit effort, with the importance of fishing grounds as perceived by the local community (Chapter 4). Then I develop, for the first time, an approach for systematic conservation planning to incorporate information on the perceived importance of places for local communities to access a range of ecosystem services and benefits, using participatory mapping techniques (Chapter 5). The rationale is that reserving important areas would limit access to or harvest in these places, and therefore incur a cost to the broader community, not just fishers.

Finally, I test the ability of marine protected areas designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives (objective 3). In Chapter 6, I compile several commonly-used proxies of biodiversity (habitat maps of different thematic resolutions created in Chapter 3) and several commonly-used proxies of socioeconomic costs (created in Chapter 4) and calculate the dollar costs of collecting each of them. Then I design protected areas that aim at protecting biodiversity while minimising socioeconomic costs using all possible combinations of proxies. I develop an index to measure the effectiveness of the resulting reserve systems to represent a reference biodiversity dataset (species) while minimising a reference cost (developed in Chapter 5).

I conclude with a general discussion on my findings, their limitations, management recommendations, and identify opportunities for future work (Chapter 7).

Although I am the lead author of all the chapters constituting this thesis, authorship of chapters for publication is shared with members of my thesis committee: Robert Pressey and Serge Andréfouët (Chapters 2, 4, 5, 6), Louisa Evans (Chapters 4 and 5), as well as several contributing co-authors: Christina Hicks (Chapter 5), Francesca Benzoni (Chapter 6), Ronald Fricke (Chapter 6), Jean-Louis Menou (Chapter 6), and Claude Payri (Chapter 6). I collected the majority of the data presented in this thesis, except for Chapter 2, and one dataset used in Chapter 6. Data used in this thesis that were provided by external sources are identified and cited within the relevant chapters. Tables and figures are shown throughout the text, and additional supporting methods and figures are provided in the appendices.

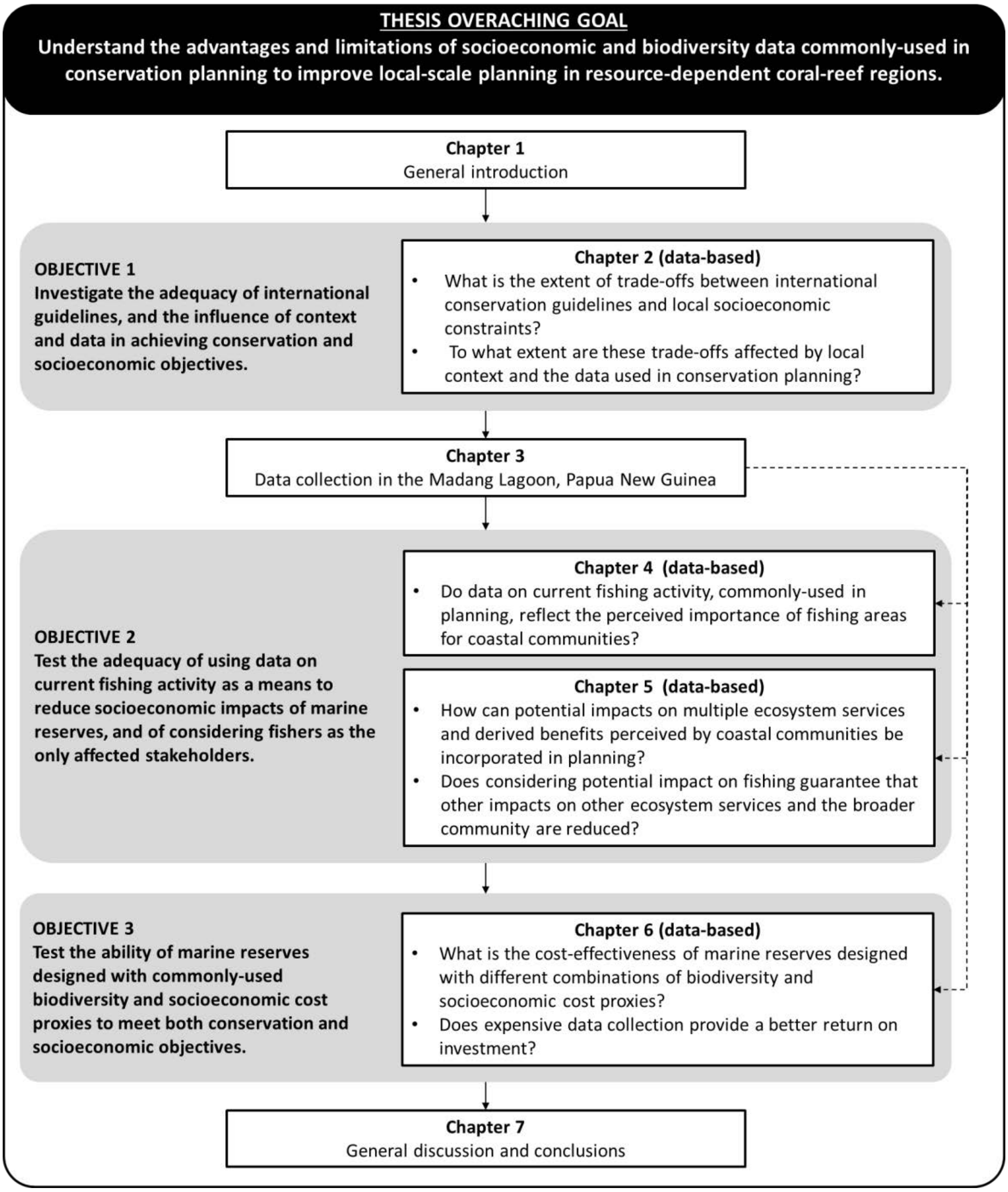


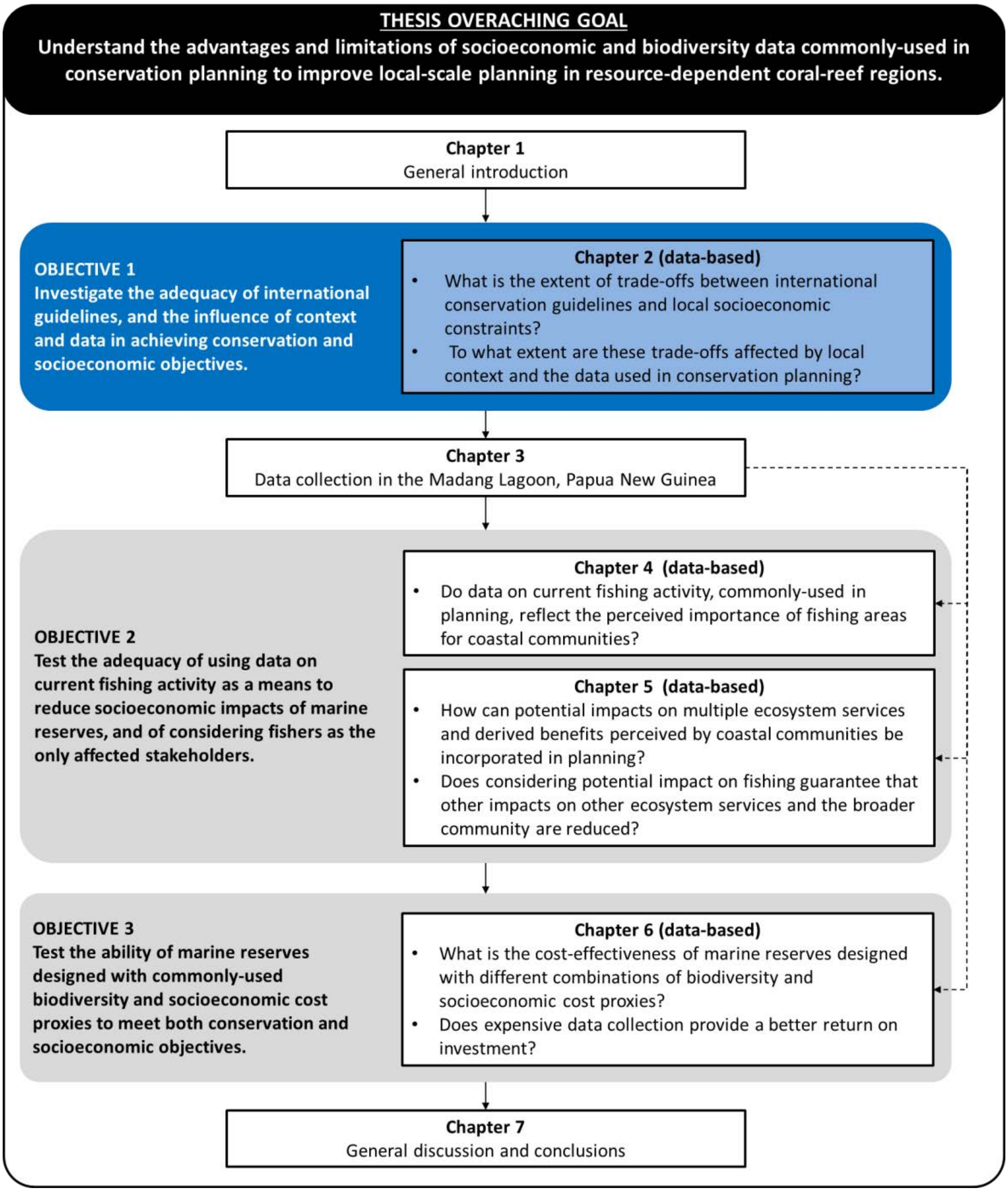
Figure 1.2. Thesis structure.

Chapter 2.

Compromises between international habitat conservation guidelines and local socioeconomic constraints(1)

In this chapter, I measure the extent of trade-offs between habitat conservation objectives based on international policy targets, and local socioeconomic fisheries objectives for the three small Pacific islands of Wallis, Alofi and Futuna. I assess whether different types of habitat maps, the overlaps between habitats and fishing grounds, and different sizes of planning units influence the achievement of objectives for each island.

[1] A version of this chapter has been published: **M. A. Hamel**, S. Andréfouët and R. L. Pressey (2013) Compromises between international habitat conservation guidelines and small-scale fisheries in Pacific island countries. Conservation Letters 6(1): 46-57. This chapter was slightly modified from the published version: some of the supplementary material was directly included in the main text.


THESIS OVERACHING GOAL
Understand the advantages and limitations of socioeconomic and biodiversity data commonly-used in conservation planning to improve local-scale planning in resource-dependent coral-reef regions.
Chapter 1
General introduction
OBJECTIVE 1
Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.
Chapter 2 (data-based)
• What is the extent of trade-offs between international conservation guidelines and local socioeconomic constraints?
• To what extent are these trade-offs affected by local context and the data used in conservation planning?
Chapter 3
Data collection in the Madang Lagoon, Papua New Guinea
OBJECTIVE 2
Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders.
Chapter 4 (data-based)
• Do data on current fishing activity, commonly-used in planning, reflect the perceived importance of fishing areas for coastal communities?
Chapter 5 (data-based)
• How can potential impacts on multiple ecosystem services and derived benefits perceived by coastal communities be incorporated in planning?
• Does considering potential impact on fishing guarantee that other impacts on other ecosystem services and the broader community are reduced?
OBJECTIVE 3
Test the ability of marine reserves designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.
Chapter 6 (data-based)
• What is the cost-effectiveness of marine reserves designed with different combinations of biodiversity and socioeconomic cost proxies?
• Does expensive data collection provide a better return on investment?
Chapter 7
General discussion and conclusions

2. Compromises between international habitat conservation guidelines local socioeconomic constraints

Abstract

Wallis, Alofi and Futuna are three small islands in the central Pacific Ocean, characterised by different reef geomorphologies. Following a request from the local Environment Service, I developed an indicative conservation plan for each island with two objectives: 1) representing 20% of the extent of each coral reef habitat within no-take areas while 2) keeping all subsistence fishing grounds open for extraction. The first objective, which was based on the 2003 Convention on Biological Diversity target, now appears more ambitious than the current Convention on Biological Diversity (Aichi) targets which recommends the protection of "10% of coastal and marine areas". I found that both objectives could not be achieved simultaneously and that large compromises are needed. Due to the small size of these islands, and the dependence of local communities on coral reef resources, the fishery objective significantly limited the extent of most habitats available for conservation. The problem is exacerbated if the conservation plan uses larger conservation units and more complex habitat typologies. My results indicate that international conservation guidelines should be carefully adapted to small Pacific islands, and that incentives to make the necessary reductions in available fishing grounds feasible will probably be needed.

Introduction

In response to increasing global and local threats to marine and coastal ecosystems, the international community has established a worldwide system of plans of action with ambitious conservation guidelines (Butchart et al., 2010, Wabnitz et al., 2010). These guidelines typically target percentage representation of marine and coastal habitats within marine protected areas (MPAs), involving various levels of restrictions on extractive uses within the seven IUCN protected area management categories (IUCN, 2008, Day J. et al., 2012). If reached successfully, these representation objectives are expected to help protect habitats, promote the viability of species, and ensure long-term and sustainable benefits to fisheries, thus sustaining economies and livelihoods as well as biodiversity.

In 2003, participants in the Cross-cutting Theme on Marine Issues at the Vth IUCN World Parks Congress in Durban, South Africa, called on the international community to include in networks of marine protected areas (MPAs) at least 20-30% of each marine habitat by 2012 (IUCN World Parks Congress, 2005). Since then, targets such as these have been identified for countries or whole regions. Regional action plans target protection in no-take areas of at least 20% of habitats associated with coral reefs (Coral Triangle Initiative, 2008), or effective conservation of at least 30% of near-shore marine resources by 2020 (The Micronesia Challenge, 2006). The USA's national conservation strategy aims at protecting at least 20% of all coral reefs and associated habitats in each major island group and Florida (United States Coral Reef Task Force, 2000). In Choiseul, Solomon Islands, the local conservation strategy aims to protect 10% of the original extent of each terrestrial and marine ecosystem (Lipsett-Moore et al., 2010).

In 2010, based on the last meeting of the Conference of the Parties to the Convention on Biological Diversity in Nagoya, Japan, the previous 20-30% global targets were revised and the new objective set to protect "10% of coastal and marine areas [...] through [...] systems of protected areas and other effective area-based conservation measures" by 2020 (UNEP/CBD/COP/10/X/2, 2010). Yet, there is increasing recognition that 1) percentages larger than 10% are likely needed in the longer term for effective conservation (Rodrigues and Gaston, 2001, Svancara et al., 2005, Gaines et al., 2010), and 2) failing to frame targets in terms of individual habitats (e.g. rezoning of the Great Barrier Reef Marine Park, Fernandes et al., 2009) is likely to strongly bias conservation to habitats easiest to protect and perhaps least in need of protection, as demonstrated widely on land (Scott et al., 2001).

Although quantitative conservation objectives are a foundation of systematic conservation planning (Margules and Pressey, 2000) and a common tool in policy, many conservation plans based on such targets are either unfeasible or ineffective in the short term (Agardy et al., 2003b, Mace et al., 2010, Wood, 2011). Important reasons include limited funds, inadequate biological or socioeconomic data, and insufficient areas available for conservation. The question of the ecological relevance of targets and general conservation guidelines is being debated within the scientific community (Carwardine et al., 2009) but the difficulty of complying with these recommendations is not yet well addressed.

How achievable are these global conservation objectives in regions such as in developing Pacific Islands, where people depend heavily on marine habitats and associated resources for day-to-day survival (Dalzell et al., 1996, Bell et al., 2009, Govan et al., 2009)? Because of their small sizes and the dependence of local communities on coastal resources, these countries might be unable to balance food security with the closure of near-shore fisheries for protection of habitats. Finding areas available for conservation that are not used for resource extraction can be difficult in these countries and, if fishing grounds are to be closed, there is often limited scope for compensating resource users or finding them alternative livelihoods (Govan et al., 2009). Consequently, tensions are likely between resource users and proponents of conservation (Hviding, 2006).

In this chapter, I demonstrate the conflict between conservation objectives based on the previous mid to upper range of international conservation guidelines and small-scale fisheries objectives for the three Pacific islands of Wallis, Alofi and Futuna, a French overseas territory where coral reef habitats are mainly exploited for subsistence. I focus on a conservation objective of 20% of each marine habitat for two reasons. First, I believe that the present 11th Aichi target (10%) will be seen as an underestimate of required protection of near-shore marine environments (Allison et al., 2003, Botsford et al., 2003). Second, I note that some jurisdictions have specified larger percentages (20-30%, or even greater) of near-shore marine habitats to be protected (Airame et al., 2003, Fernandes et al., 2009, Mills et al., 2011). Specifically, I show the spatial extent of the trade-off between sets of objectives for conservation and fisheries. In this case, I considered the best protection option for marine habitats and associated resources (i.e. the implementation of no-take zones) and the best-case short-term scenario for fisheries (i.e. no restriction on take in current fishing grounds). Different sizes of potential no-take zones and different habitat data were tested to understand their influence on the trade-offs between conservation and fisheries objectives.

Materials and methods

Study sites and context

Wallis, Alofi and Futuna are three Polynesian high islands belonging to the French Territory of Wallis and Futuna Islands, in the central South Pacific (**Figure 2.1**). Both Wallis and Futuna, the largest islands, are inhabited. Alofi is uninhabited but used daily by Futuna habitants for agriculture, fishing and leisure. The three islands vary in size and in habitat complexity.

Because tourism is not well developed, demand for reef fish is confined to local communities, who exploit the reefs and lagoons for their own subsistence. Almost a third of the population practises artisanal small-scale fishing in Wallis, Alofi and Futuna, mostly with nets and spearguns (Egretaud et al., 2007b, Egretaud et al., 2007a). Most catches are either eaten by the fishers themselves or exchanged or given away, with the remainder sold to buy fuel for fishing boats. The total lagoon fishery production, estimated at 200-300 tonnes per year (Kronen et al., 2008, Ministère de l'Outre-Mer, 2011), is lower than the local demand, estimated at 900 tonnes per year (Egretaud et al., 2007b). This indicates that reduction in production resulting from marine conservation actions will be problematic.

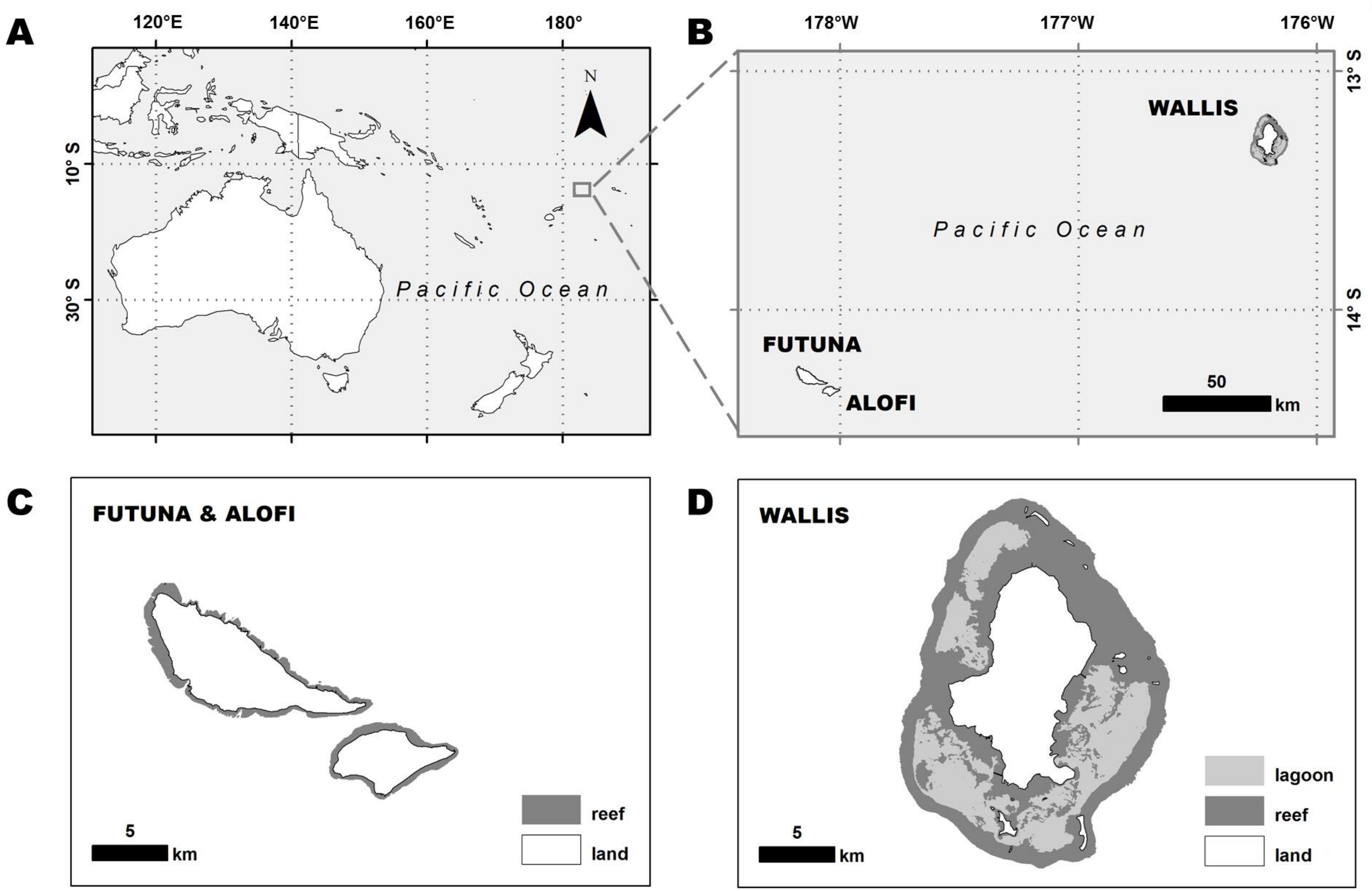


Figure 2.1. Location of Wallis, Alofi and Futuna islands (a, b) and main reef features of Futuna and Alofi (c) and Wallis (d).

Thus far, there are no MPAs in Wallis, Alofi and Futuna that have been integrated into a territorial management plan (Verducci and Juncker, 2007). However, three small informal customary MPAs have been established in Wallis, based on a *ad hoc* decisions between local fishers, customary authorities, and the Territorial Environment Service (Egretaud et al., 2007b). Wide agreement among both the local communities and authorities to increase the protection of these islands' reefs and manage their resources led the local authorities to launch a territorial Management Plan for Marine Areas (in French, *Plan de Gestion des Espaces Maritimes*). Based on France's national biodiversity strategy (Ministère de l'Écologie du Développement durable des Transports et du Logement, 2011) which parallels the Aichi biodiversity targets (UNEP/CBD/COP/10/X/2, 2010), the future territorial management plan for Wallis aims to move from the existing informal reserves to a new MPA network at the territory level.

Objectives for the indicative conservation plan for this study

An indicative conservation plan was set up for the three islands to reflect objectives of the Territorial Environment Service, with two sets of objectives based respectively on international habitat conservation guidelines and local fishery requirements. Wallis managers initially considered the guidelines of the previous 2002-2010 CBD Strategic Plan, which targeted a mid- to long-term protection of 20-30% of each habitat (IUCN World Parks Congress, 2005). They also considered a network of no-take areas as the most efficient tool to provide rapid ecological benefits. Thus, for this study, the conservation objective was to include 20% of the total extent of each habitat within no-take areas: a more ambitious amount than the new CBD marine target (11th target in UNEP/CBD/COP/10/X/2, 2010) and, unlike the CBD guidelines, not considering zonings that permit extractive uses.

Here, the fishery objective was intended to avoid conflicts with local communities highly dependent on reef habitats both culturally and economically. It prevented the main current fishing grounds for nets and spearguns from being identified as no-take areas. The data and analyses below allowed an assessment of the extent to which these potentially conflicting objectives can be reconciled.

Coral-reef habitat maps

Two types of habitat maps with two levels of detail were used (see detailed methods in Andréfouët and Dirberg (2006)). First, a high-resolution geomorphic habitat map (hereafter the "geomorphic" map) of the three islands was derived from the Millennium Coral Reef Mapping Project (Andréfouët et al., 2006a). Millennium Coral Reef Mapping Project maps were all created from Landsat 7 ETM+ satellite imagery at 30 m spatial resolution. Second, a very high-resolution detailed habitat map (hereafter the "geomorphic + benthic" map), combining geomorphic attributes and benthic data, was derived from digital aerial photographs at 2 m spatial resolution (see Andréfouët and Dirberg (2006), Andréfouët et al. (2007), and Dalleau et al. (2010) for more information on these maps). In Wallis, Alofi and Futuna, geomorphic habitats were described in 16, 4 and 3 thematic classes, respectively. The combination of data in the geomorphic + benthic maps allowed coral-reef and other associated habitats (e.g. terraces dominated by soft substrate, seagrass beds) to be described in 55, 6 and 3 thematic classes, respectively, excluding land features. To understand how map type would affect the feasibility of conservation and fishery objectives, all analyses were conducted for both habitat maps.

Fishing grounds data

In 2006, an environmental study was conducted in Wallis, Alofi and Futuna to initiate the *Plan de Gestion des Espaces Maritimes* (Egretaud et al., 2007b, Egretaud et al., 2007a). Socio-economic and environmental data were collected to map the different uses of the marine environment and to understand the views and expectations of stakeholders in terms of management. The fishery geographic information system includes the boundaries of coral reef fishing grounds, itemized for different fishing gears and techniques (gleaning, line, speargun, net, and informal offshore fishing). For my analysis, I focused on net and speargun fisheries because they were the dominant gears (Egretaud et al., 2007c, Kronen et al., 2008). For Alofi and Futuna, only speargun data were available.

Conservation units

Each island of interest was partitioned into manageable conservation units by superimposing a grid of square cells on all areas containing coral reef habitats on the maps. To understand whether the size of conservation units would affect the feasibility of reconciling conservation and fishery objectives, all analyses were conducted for two different sizes of conservation units (500 x 500 m and 200 x 200 m, hereafter "large" and "small", respectively). The number of conservation units used in the following analyses is indicated in **Table 2.1** for each island.

Table 2.1. Total number of conservation units used in the analyses for each island.

Island	Habitat map	Size of conservation units 500 x 500 m	200 x 200 m
Wallis	Geomorphic	1,033	5,902
	Geomorphic + Benthic	1,012	5,765
Alofi	Geomorphic	105	467
	Geomorphic + Benthic	83	317
Futuna	Geomorphic	225	1,079
	Geomorphic + Benthic	150	623

Reef area left available for conservation

The set of possible no-take areas or areas left for conservation considered in these analyses initially excluded all fished units. For each combination of island / habitat map / size of conservation unit, I first measured the proportion of conservation units available for conservation when all fishing areas were left open. Within all conservation units (fished, and unfished), I computed the number of habitats (excluding land features) and their extent, and the occurrence of net or speargun fishing. Then I measured the proportion of habitats that could potentially meet the 20% objective within unfished conservation units. All data were analysed using ESRI® ArcMap™ 10.0 and R (R Development Core Team, 2008).

Trade-off between objectives for habitat conservation and small-scale fisheries

Conservation objectives were fully achieved when 100% of habitats could meet the 20% objective. Full achievement of the fishery objective meant that 100% of initial fished units were available for harvest. I measured the actual extent to which conservation objectives must be compromised to fully achieve the fishery objective, and *vice versa*. For this, I created trade-off curves with Marxan (Possingham et al., 2000, Ball et al., 2009), a software system for systematic conservation planning. Fished conservation units were attributed a cost of 1. Non-fished conservation units had zero cost.

I applied Marxan iteratively, starting with the set of unfished conservation units (full achievement of the fishing objective) and progressively increasing the percentage of fished conservation units to be considered as potential reserves (allowing increasing achievement of the habitat conservation objectives). To do this, I defined a series of increasing cost thresholds by the percentages of the total fished conservation units that could be moved to no-take zones: from 0 to 100% in 10% increments. For each of these 11 cost thresholds, 1000 different sets of conservation units were selected to meet the objectives. Across these 1000 repeat runs, I selected the best solution (with the lowest total cost) for each percentage threshold. For each threshold, I recorded the proportion of habitats that would meet their objectives in the best solution.

Results

Avoiding *a priori* any fishing ground in no-take areas had a large impact on the number of conservation units available for conservation, and on the extent of habitats that could be protected. At best, for both sizes of conservation units, about 60% of the potential conservation units would be left available in both Wallis and Futuna, and about 20% in Alofi (**Figure 2.2**). In these sets of available conservation units, a substantial proportion of the total number of habitats could not meet the 20% objective (unrepresented and under-represented habitats in **Figure 2.3**).

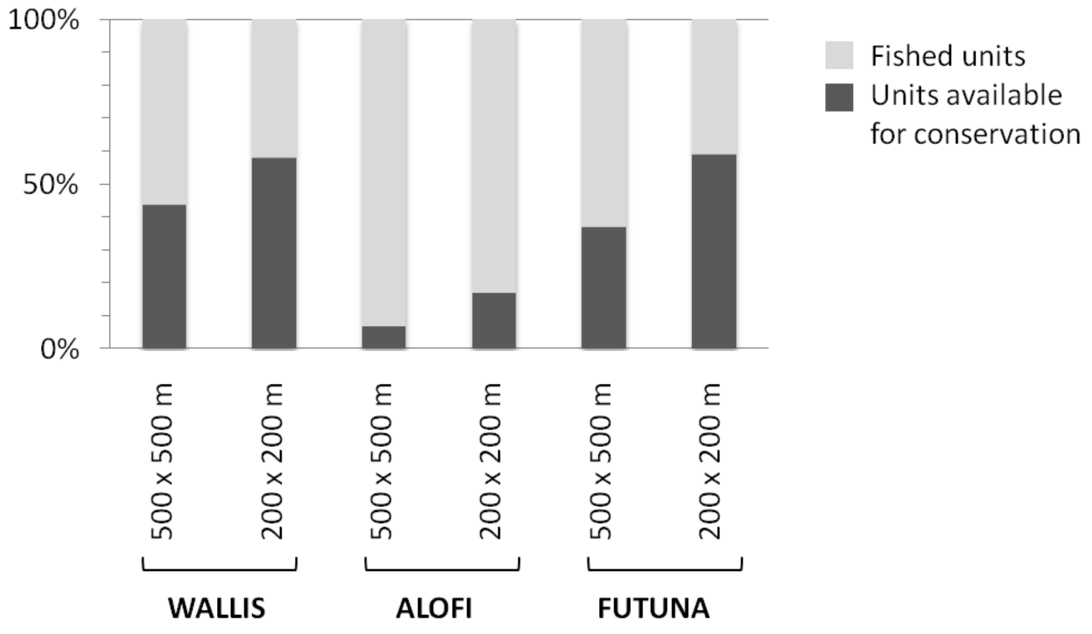


Figure 2.2. Percentages of large and small conservation units fished (with speargun and net) or available for habitat conservation in Wallis, Alofi and Futuna.

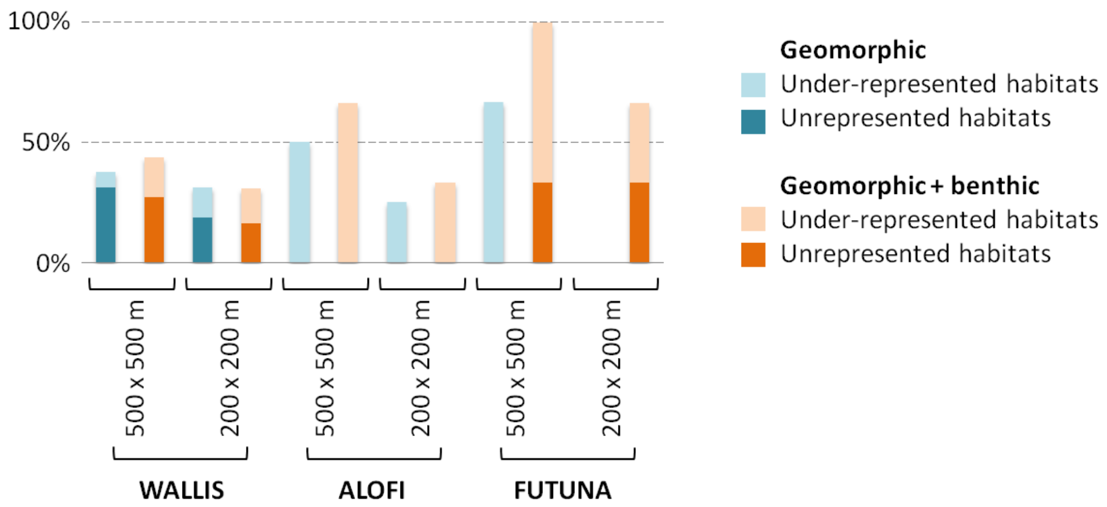


Figure 2.3. Percentages of habitats that were under-represented or unrepresented in units available for conservation after exclusion of fished units. Under-represented habitats are those for which conservation objectives were only partially achievable (less than 20% of total extent available for conservation). Unrepresented habitats were those that could not be protected at all because their entire extents were fished. Figures are shown for the three islands (Wallis, Alofi and Futuna), two sizes of conservation units (500 x 500 m and 200 x 200 m), and two types of maps (geomorphic in blue, and geomorphic + benthic in red). I considered 16 geomorphic habitats and 55 geomorphic + benthic habitats for Wallis, 4 geomorphic habitats and 6 geomorphic + benthic habitats for Alofi, and 3 geomorphic habitats and 3 geomorphic + benthic habitats for Futuna.

Figure 2.4 shows the spatial distribution of unrepresented and under-represented habitats. **Figure 2.5** shows the extent of each habitat left available for conservation (see **Table 2.2** and **Table 2.3** for lists of specific under-represented and unrepresented habitats types and percentage representation in unfished areas). Fringing reefs are of concern over the three islands. All geomorphic fringing reef classes (3/3 for either size of conservation unit) (**Table 2.2**) and most geomorphic + benthic fringing reef classes (11/13 for smaller conservation units, 12/13 for larger ones) (**Table 2.3**) could not meet their 20% objectives. In Wallis, barrier reef and channel habitats would also be under-protected, especially when considering geomorphic + benthic habitats and larger conservation units (10/26 classes, against 5/26 for smaller units). In addition, none of the seagrass beds (4/4 geomorphic + benthic classes) would be represented in Wallis. Using smaller conservation units in Futuna would allow all three geomorphic + benthic habitats to meet their conservation objectives. Overall, conservation objectives cannot be fully achieved if all fishing grounds are avoided, except for Futuna when geomorphic habitats and smaller conservation units are considered.

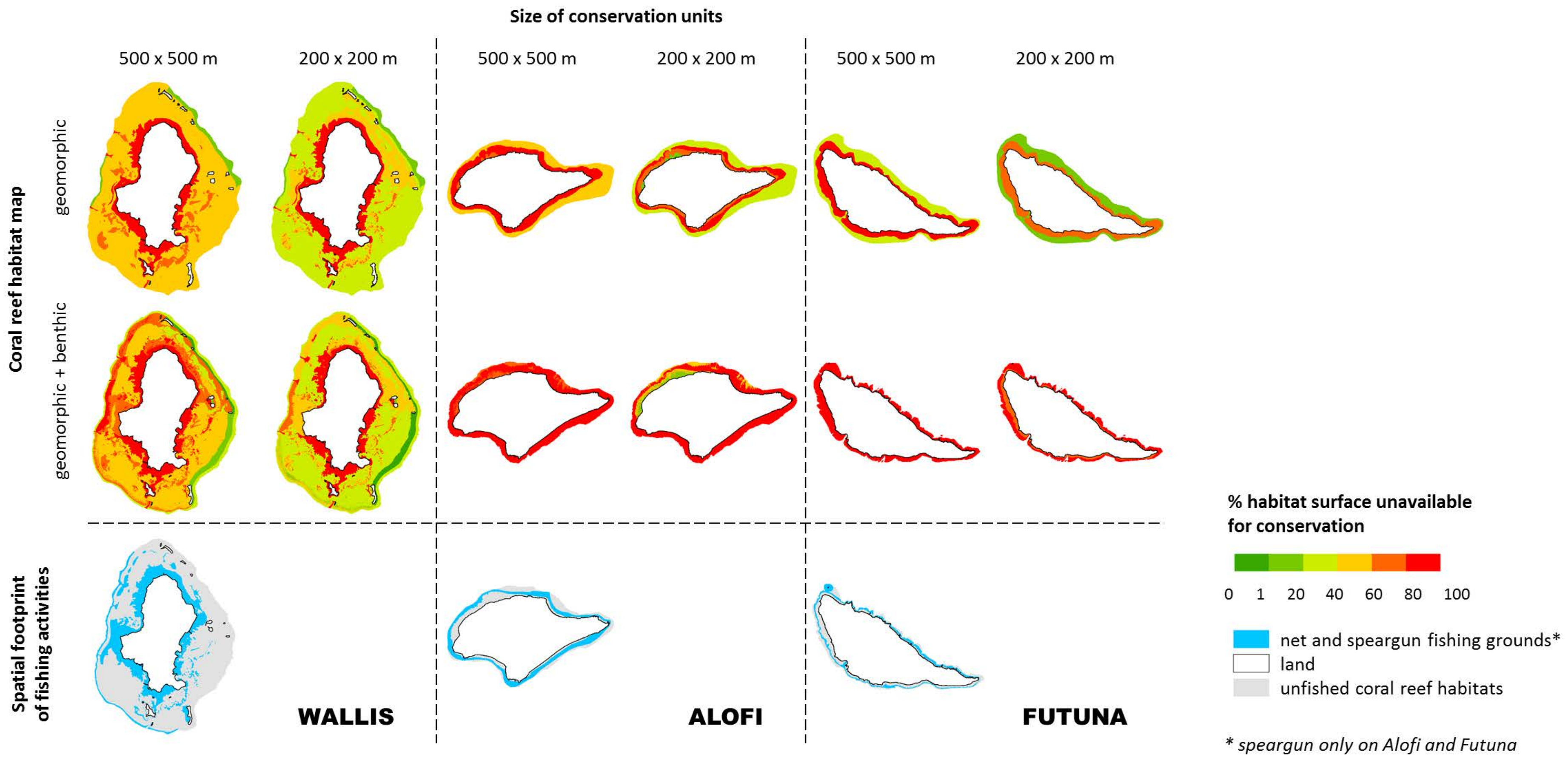


Figure 2.4. Spatial patterns of under-represented and unrepresented habitat types in units available for conservation after exclusion of fished units, both shown in red. Under-represented and unrepresented habitats were those for which conservation objectives were only partially achievable (more than 80% of total extent unavailable for conservation, so less than 20% available) or not achievable, respectively. Effects of exclusion of fished conservation units are shown for each island, two sizes of conservation units, and two types of maps. The lower panels show the spatial footprint of net and speargun fishing in blue. For Alofi and Futuna Islands, only data on speargun fishing could be obtained.

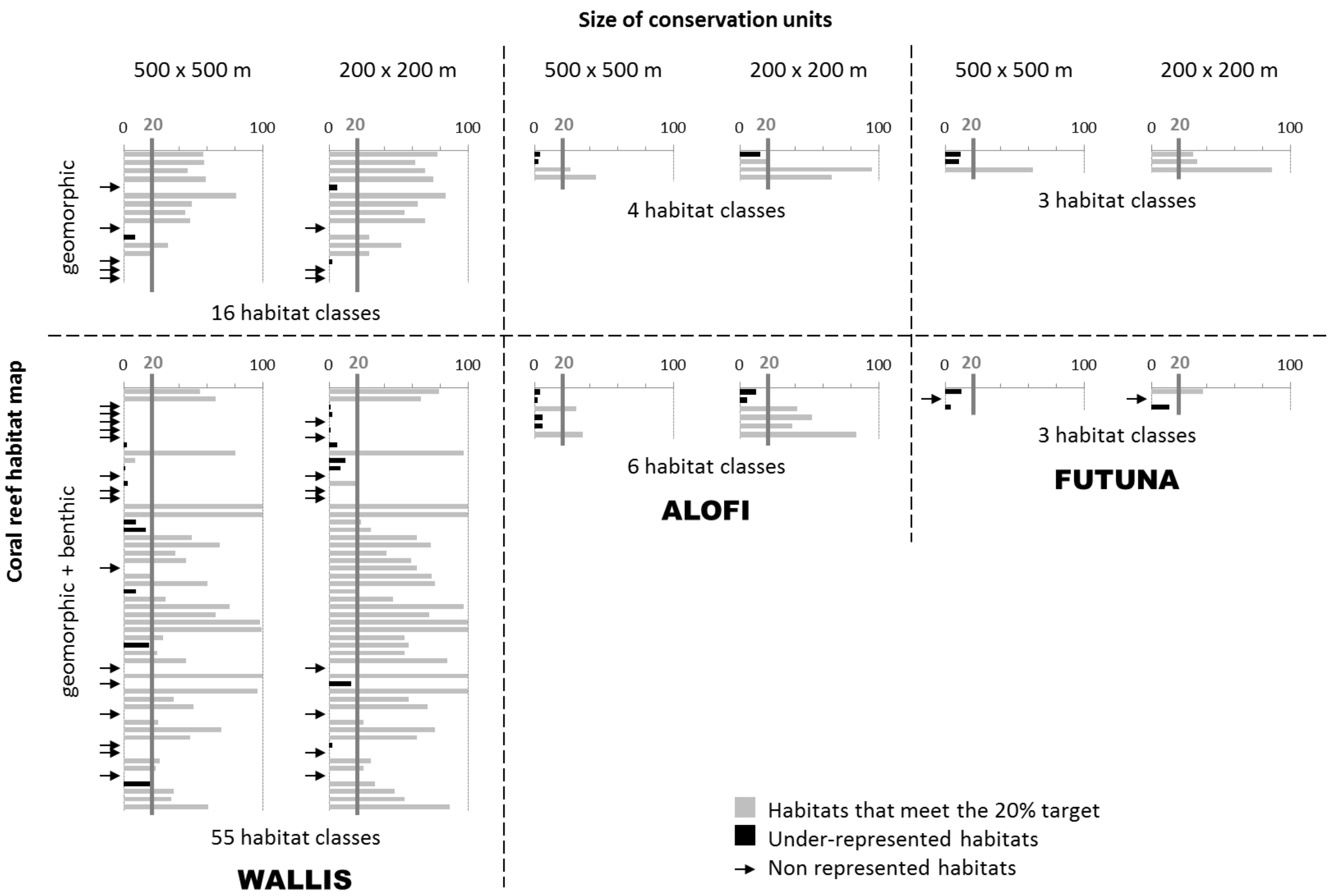


Figure 2.5. Effects of exclusion of fished conservation units on the extent of individual coral reef habitats available for conservation, for the three islands, two sizes of conservation units, and two types of maps. Each bar corresponds to one habitat class. Habitat class labels have been omitted for clarity. Habitats for which conservation objectives were achievable are shown in grey. The vertical grey line indicates the 20% objective for each habitat. Geomorphic and geomorphic + benthic habitats for which conservation objectives were not achievable (less than 20% of total extent available for conservation) are shown in black. Habitats that could not be protected at all are shown with arrows.

Table 2.2. Percentages of total extent of geomorphic habitats (level 5 classification) available for conservation after exclusion of all fished conservation units (500 x 500 m and 200 x 200 m) for Wallis, Alofi and Futuna. Only under-represented and unrepresented habitats are listed. Points indicate habitat types that would not be represented at all in the set of potential reserves. Asterisks indicate habitat types for which the 20% objective could be achieved with the smaller conservation units (200 m x 200 m). Under-represented and unrepresented habitats are mapped in red in **Figure 2.4**.

Island	Geomorphology (level 1)	Geomorphology (level 2)	Geomorphology (level 3)	Geomorphology (level 4) [1]	% surface (500 x 500 m)	% surface (200 x 200 m)
Wallis	Oceanic	Oceanic island	Outer Barrier Reef Complex	Pass	.	6
			Coastal Barrier Reef Complex	Channel	.	.
				Barrier reef pinnacle/patch	8	20*
			Lagoon exposed fringing	Reef flat	.	2
				Enclosed lagoon or basin	.	.
			Fringing of coastal barrier complex	Reef flat	.	.
Alofi	Oceanic	Oceanic island	Ocean exposed fringing	Forereef	4	15
				Reef flat	3	20*
Futuna	Oceanic	Oceanic island	Ocean exposed fringing	Forereef	11	20*
				Reef flat	10	20*

[1] Our analyses in this study were for the level 5 of the geomorphic classification, which is a combination of levels 1, 2, 3, and 4. Only unrepresented or under-represented habitats are listed here.

Table 2.3. Percentages of total extent of geomorphic + benthic habitats available for conservation after exclusion of all fished conservation units (500 x 500 m and 200 x 200 m) for Wallis, Alofi and Futuna. Only under-represented and unrepresented habitats are listed. Points indicate habitat types that would not be represented at all in the set of potential reserves. Asterisks indicate habitat types for which the 20% objective could be achieved with the smaller conservation units (200 x 200m) only. Under-represented and unrepresented habitats are mapped in red in **Figure 2.4**.

Island	Geomorphology (level 1)	Geomorphology (level 2)	Benthos (level 1)[1]	% surface (500 x 500 m)	% surface (200 x 200 m)
Wallis	Fringing reef	Reef flat	Hard substrate with dispersed coral	.	1
			Soft substrate with dispersed coral	.	2
		Terrace	Seagrass/algae bed	.	.
			Seagrass/algae bed	.	1
			Seagrass/algae bed	.	.
			Seagrass/algae bed	2	6
		Reef slope	Coral	8	12
			Soft substrate with dispersed coral	1	8
			Soft substrate	.	.
			Soft substrate	3	20*
			Coral	.	.
			Soft substrate	.	.
	Coastal barrier reef	Reef flat	Hard substrate with dispersed coral	9	20*
			Mixed substrate with dispersed coral	16	20*
		Terrace	Algae bed	.	20*
			Soft substrate with dispersed coral	9	20*
	Barrier reef	Reef flat	Mixed substrate with dispersed coral	18	20*

[1] Our analyses in this study were for a combination of both geomorphology levels and benthos level 1 of the geomorphic + benthic classification. Only unrepresented or under-represented habitats are listed here.

Island	Geomorphology (level 1)	Geomorphology (level 2)	Benthos (level 1)[1]	% surface (500 x 500 m)	% surface (200 x 200 m)
			Coral	.	.
		Terrace	Algae bed	.	16
			Soft substrate	.	.
		Pass	Hard substrate	.	2
			Hard substrate with dispersed coral	.	.
	Lagoon patch reef	Reef flat	Soft substrate with dispersed coral	.	.
		Terrace	Soft substrate	19	20*
Alofi	Fringing reef	Reef flat	Hard substrate	4	12
		Reef slope	Hard substrate with dispersed coral	2	5
		Reef flat	Hard substrate with dispersed coral	6	20*
			Coral	6	20*
Futuna	Fringing reef	Reef flat	Hard substrate	12	20*
			Coral	.	.
		Reef slope	Hard substrate with dispersed coral	4	13

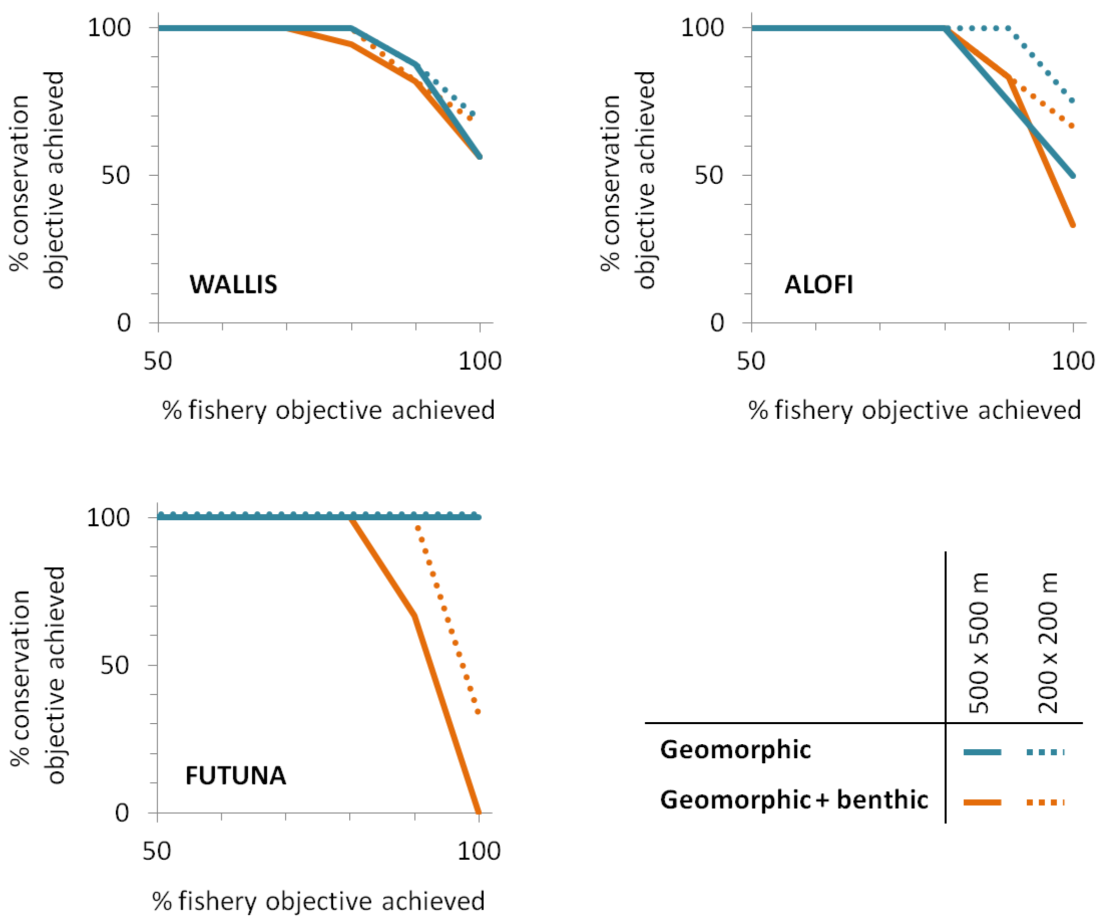


Figure 2.6. Trade-offs between conservation and fishery objectives in Wallis, Alofi and Futuna, for two types of maps (geomorphic and geomorphic + benthic) and two sizes of conservation units (500 x 500 m and 200 x 200 m). % conservation objective achieved corresponds to the percentage of habitats that met their 20% objectives. % fishery objective achieved corresponds to the percentage of fished conservation units excluded from the sets of conservation units selected as potential no-take areas.

The trade-off analyses showed that, if local communities harvested 100% of their fishing grounds (full achievement of the fishery objective), only 69% of all Wallis habitats and 75% of Alofi habitats, at best, can meet their conservation objectives (**Figure 2.6**). For all habitats to reach their objectives, at least 20% of fished conservation units must be made available for conservation in Wallis and Alofi. In general, geomorphic habitats could meet their conservation objectives with less impact on fisheries than geomorphic + benthic habitats. The habitat conservation objectives were also achieved with less impact on fisheries when smaller conservation units were used. A combination of geomorphic habitats and smaller conservation units reduced (Wallis, Alofi) or eliminated (Futuna) the trade-off between conservation and fishery objectives. Modifying sizes of conservation units and thematic resolution of habitat maps had the greatest impact on the trade-off in Futuna, the simpler island in terms of habitats.

Discussion

Achieving 20% habitat conservation targets within no-take areas appears incompatible with the socio-economic context of small Pacific island countries such as Wallis, Futuna and Alofi. Managers and conservation planners worldwide are already well aware of the trade-offs between conservation and fisheries objectives (see Proceedings of the Fourth World Fisheries Congress: Reconciling Fisheries with Conservation, , 2004), and MPA designs now account for local socio-economic constraints (Klein et al., 2008). Here, I quantified the extent of trade-offs needed to reconcile two conflicting types of objectives, in the particular context of small tropical coral reef islands, rich in habitats, limited in extent, and with high levels of use for daily subsistence, and thus with limited scope to favour habitat protection over food security.

To my knowledge, no study has previously assessed the extent of trade-offs between fisheries and strict habitat conservation in tropical island contexts. My analyses, based on immediately available datasets, provide quantitative and spatially explicit answers to questions about potential compromises to be made between core objectives for managing marine regions. I found that achievement of both strict conservation and fishery objectives is generally not possible in Wallis, Alofi and Futuna, regardless of the habitat maps and sizes of conservation units considered. However, although the 20% no-take objectives were not all achievable without reducing fishing areas, the achievement of objectives for specific habitats was greatly influenced by the type of habitat maps and the size of the conservation units, so choosing these variables carefully is fundamental to understanding such trade-offs.

My results show that using more detailed habitat maps makes the achievement of habitat objectives more difficult. Very detailed habitat maps provide a finer description of physical and biological variation across reef systems, and Dalleau et al. (2010) suggested that finer-resolution coral reef habitats might also be better surrogates of species diversity. However, finer-resolution maps also contain more habitat classes that need to be represented in conservation designs, increasing the total area required to achieve objectives. The reason for this increase is the greater mismatch between the boundaries of conservation units and those of more detailed habitat classes, as demonstrated for terrestrial planning by Pressey and Logan (1995).

Decisions about the sizes of conservation units also influence the potential to achieve both habitat conservation and fishery objectives. In general, smaller conservation units allow a more exact habitat representation, for any given spatial and thematic resolution of map, in the sense that fewer objectives are over-achieved (see Pressey and Logan, 1995, Pressey and Logan, 1998 for land systems, Mills et al., 2010 for coral reefs). My findings agree with these studies, showing that smaller conservation units lead to easier compromises. However, the appropriate size of conservation units must take into account other factors such as data resolution, manageability, and effectiveness of conservation actions in the long-run (Gaines et al., 2010, Mills et al., 2010). For example, single large marine reserves may provide more ecological benefits, while several small ones may allow for more flexibility in location or accommodate human needs. In different contexts large or small reserves will be preferred for their ease of enforcement.

Prioritising the fishery objective in Wallis, Alofi and Futuna, as requested by the Territorial Environment Service, resulted in a deficit of reef areas available for conservation. The consequences were that several habitat types could not be protected at all, or could only partially achieve their objectives. These results were expected because several types of reef habitats are preferentially used by net and speargun fishers for their accessibility, wind and current exposure, and resource abundance.

An extreme response to avoiding conflicts between conservation and fishery objectives is to concentrate future fisheries in already heavily fished or overfished areas. However, failing to protect such areas from further fishing is likely to preclude their ability to restore or enhance stocks, and ignores the potential benefits of spillover of larvae and adults to supplement other fished areas. Therefore, avoiding conflicts at all costs might not be the best strategy in the long run, even to maximise benefits to fishers themselves. Although subject to controversy, there is a now a wide body of work demonstrating the benefits of MPAs to adjacent fisheries (Roberts et al., 2001, Hilborn et al., 2004, Russ et al., 2004a, Kaiser, 2005, Harrison et al., 2012). To incorporate these perspectives, my indicative conservation designs could be refined by considering past fishing activities, fishing pressure and yields, and the locations of potential MPAs relative to fished areas.

The achievement of objectives for both fisheries and conservation as I defined them here for small Pacific islands is clearly not feasible without compromises on both sides. Methods to identify achievable, realistic objectives early in the process of conservation planning are needed urgently for these countries. By achievable, I mean objectives that can actually be reached through effective conservation and management actions, not simply on paper (Riegl et al., 2009). By realistic, I mean conservation objectives that converge towards the strict application of international objectives while allowing some flexibility to minimise socio-economic impacts on local communities heavily dependent on fishing. On the fisheries side, objectives must allow flexibility to minimise impacts on targeted species, other species, and physical habitats.

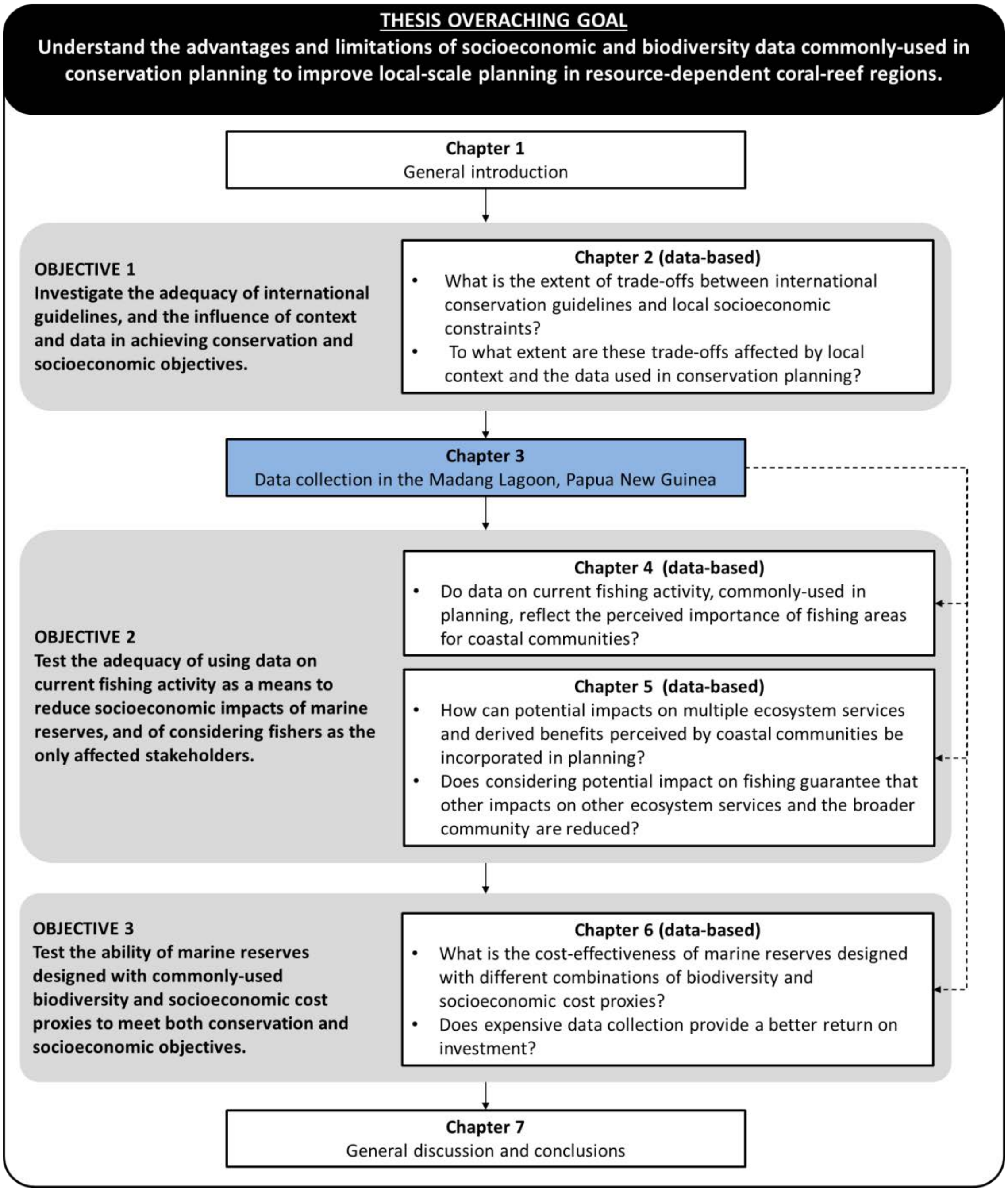
Current international conservation guidelines such as the CBD targets have moderated their options through time to allow some flexibility for complying countries. If no-take areas are seen by some as the best protection option for marine habitats and associated resources (see Graham et al., 2011a for a review on benefits for coral reefs), they are also difficult to implement in many contexts. Low compliance can be expected, especially in small Pacific countries that depend heavily on coastal habitats and associated resources. There is now a multitude of spatial arrangements for marine management that do not require total and permanent restrictions on harvest (see for example Cinner and Aswani, 2007, Gaines et al., 2010, Agardy et al., 2011, Mills et al., 2011). The effect of different management strategies on trade-offs between fisheries and the conservation of ecosystem functions has also been investigated (Brown and Mumby, 2014), and similar analyses could be used in the context of habitat protection. The IUCN protected area management categories themselves offer a range of options to avoid the hard trade-offs demonstrated in this chapter.

Incentives to accept marine conservation measures are also being investigated and include compensatory services such as schools or medical facilities (e.g. Aswani and Weiant, 2004) and buyouts or alternative livelihoods (e.g. Niesten and Gjertsen, 2010, Jones and Qiu, 2011). For islands where most fishing activities provide food for subsistence, incentives might also need to include additional imports of food. My analyses show that the extent of trade-offs between objectives and the need for such incentives can be demonstrated readily. Appropriate responses to trade-offs must then be formulated amongst the affected communities, decision makers concerned with fisheries and conservation, and conservation scientists.

Chapter 3.

Data collection in the Madang Lagoon, Papua New Guinea

Chapters 4, 5 and 6 are based on data collected in the Madang Lagoon, Papua New Guinea. To avoid redundancy in the thesis, in this chapter I describe general data collection and processing methods. I first introduce the Madang Lagoon, then I describe the data I collected and the data that was provided by collaborators. The data collected in the Madang Lagoon consisted of: marine habitat mapping, biodiversity surveys, fisher surveys, and household surveys. Finally, I also explain basic data requirements for conservation planning, briefly introduce Marxan, the spatial prioritisation algorithm used throughout the thesis, and describe how I defined planning region and planning units for all the following planning exercises.


THESIS OVERACHING GOAL
Understand the advantages and limitations of socioeconomic and biodiversity data commonly-used in conservation planning to improve local-scale planning in resource-dependent coral-reef regions.
Chapter 1
General introduction
OBJECTIVE 1
Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.
Chapter 2 (data-based)
• What is the extent of trade-offs between international conservation guidelines and local socioeconomic constraints?
• To what extent are these trade-offs affected by local context and the data used in conservation planning?
Chapter 3
Data collection in the Madang Lagoon, Papua New Guinea
OBJECTIVE 2
Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders.
Chapter 4 (data-based)
• Do data on current fishing activity, commonly-used in planning, reflect the perceived importance of fishing areas for coastal communities?
Chapter 5 (data-based)
• How can potential impacts on multiple ecosystem services and derived benefits perceived by coastal communities be incorporated in planning?
• Does considering potential impact on fishing guarantee that other impacts on other ecosystem services and the broader community are reduced?
OBJECTIVE 3
Test the ability of marine reserves designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.
Chapter 6 (data-based)
• What is the cost-effectiveness of marine reserves designed with different combinations of biodiversity and socioeconomic cost proxies?
• Does expensive data collection provide a better return on investment?
Chapter 7
General discussion and conclusions

3. Data collection in the Madang Lagoon, Papua New Guinea

Study area

Madang Lagoon, the largest and most ecologically diverse lagoon on the north coast of Papua New Guinea (Jebb and Lowry, 1995, Jenkins, 2002a, Miller and Sweatman, 2004), is located in the Coral Triangle region, extending 16 km north to south and 4 km west to east, with a surface of 40 km^2 and a maximum depth of 54 m (**Figure 3.1**). My choice of study area had two motivations. First, my socioeconomic work was combined with an extensive biological survey (the Papua-Niugini 2012-2014 expedition, Muséum National d'Histoire Naturelle), not only facilitating logistics but also providing data on habitats and biodiversity for future planning (e.g. Fricke et al., 2014). Due to the size and scope of the expedition, the comprehensiveness of the overall dataset for one region (e.g. socioeconomic assessment, marine habitats and marine biodiversity) was fairly unique. Second, similar to most coral-reef countries (e.g. Bell et al., 2009, Burke et al., 2011), coastal communities of the Madang Lagoon rely on coral-reef resources for their day-to-day life, subsistence, and income (Marnane et al., 2002, Kinch et al., 2005, Jenkins, 2011). However, unrestricted and sometimes destructive fishing practices, coupled with rapid population growth, make small-scale fisheries unsustainable and threaten both ecosystems and human communities (Cinner and McClanahan, 2006). Building approaches that achieve objectives for conservation, fisheries, and livelihoods is therefore critical.

For my socioeconomic surveys, I focused on Riwo (Ziwo), the largest coastal community, because of its central location (**Figure 3.1**), large population (National Statistical Office of Papua New Guinea, 2002), and history of involvement in conservation and resource management. Three of the four Wildlife Management Areas (WMAs) in the lagoon were established by clans of Riwo, with the support of local and international NGOs (Jenkins, 2002a, Jenkins, 2002b).

The Madang Lagoon, and the Riwo community in particular, was an ideal setting for a local-scale planning exercise. In my thesis, I chose local-scale planning because it is more relevant to developing coral-reef countries where decentralised governance is common (Berkes et al., 2006), government institutions do not have the adequate resources for large-scale management (Mills et al., 2010), and because ultimately, broad-scale plans must be interpreted locally, at which point fine-scale biodiversity and cost information are critical (Pressey et al., 2013).

Figure 3.1. (next page). Map of the Madang Lagoon, Papua New Guinea, showing data collection sites. Inset (A) shows the location of the Madang Lagoon in Papua New Guinea, and the geomorphological context. Map in (B) shows the portion of the Madang lagoon that was covered by the satellite image used for habitat mapping, and the locations of survey sites for fish, macro-algae, corals, macro-invertebrates, and habitats. Map in (C) shows detail in the Riwo region, including names of main islands, and reefs (italic), and locations of the socioeconomic surveys (main landing sites for the fisher surveys, and households for the household surveys). Of the four Wildlife Management Areas in the study area, three (Sinub, Tabad and Laugum in the northern section) are managed by the Riwo community. In "fully protected areas", all forms of resource extraction as well as anchoring and rubbish disposal are prohibited, and access to visitors from commercial tours is limited. In "high level managed fishing areas", only line fishing is allowed. In "low level managed fishing areas", subsistence fishing without destructive methods is allowed.

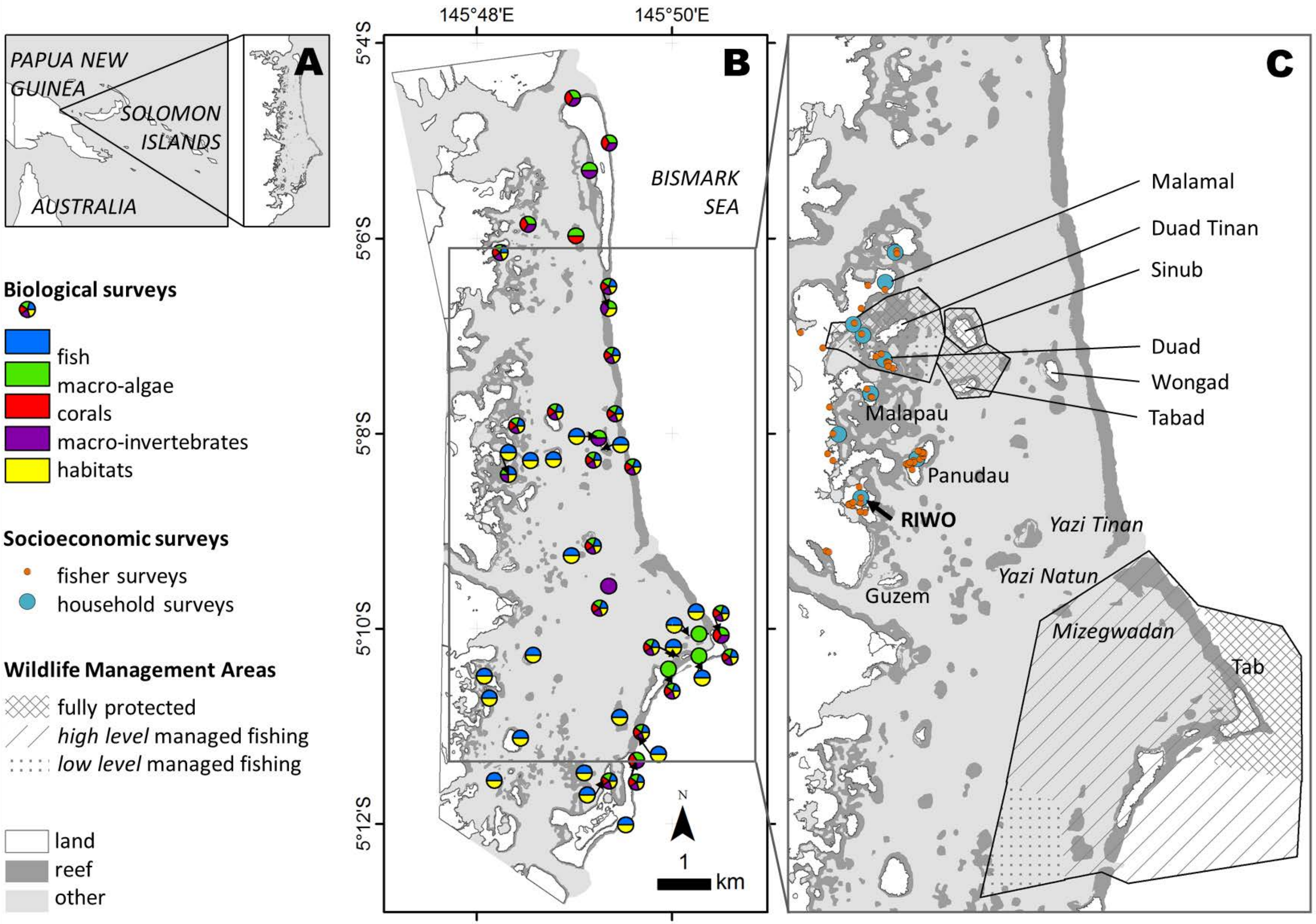


Figure 3.1. Map of the Madang Lagoon, Papua New Guinea, showing data collection sites.

Marine habitat mapping

For Chapters 4, 5 and 6, I produced and used a high spatial resolution map of the coral reefs and associated habitats occurring in the Madang Lagoon, with a hierarchical classification of habitats consisting of seven thematic levels. The map was created following the steps in the "user approach" described in Andréfouët (2008): *a priori* manual delineation of habitats, ground truthing, contextual editing, classification and merging of habitat segments. The map was based on a high spatial (1.85 m multispectral) and spectral (8 bands) resolution Worldview-2 satellite image of the study area acquired June 12th, 2010. With optimal conditions, satellite imagery such as this one can provide information on marine habitats down to 20 meters depth but more generally around 10-15 meters for coral reefs (see for example Andréfouët et al., 2003, Andréfouët et al., 2012b). In the following, a "habitat" is defined at the landscape scale, roughly occurring within a square of 10 by 10 meters.

First, habitat segments (or "polygons") were manually delineated using ArcGIS (ESRI, 2010), according to differences in colour and texture on a true colour image.

Second, a representative sample of habitats was visited in the field for ground truthing in October 2012 as part of the Papua-Niugini 2012-2014 expedition (Muséum National d'Histoire Naturelle). At each of the 39 sites visited (**Figure 3.1**), underwater photographs were taken at regular time intervals to document the different habitats encountered, during 45-80 minutes SCUBA dives, from the surface down to 30 m depth. Semi-quantitative data on variables such as geomorphology, depth, benthic cover, abiotic substrates, rugosity and main reef building communities were also recorded following the medium scale approach (Clua et al., 2004). Photographs were subsequently sorted by site and habitat type.

Third, combining information from the satellite image, photographs, and field data, I classified the habitats segmented *a priori* according to seven themes: 1) the five levels of coral reef geomorphology classification used in the Millennium Coral Reef Mapping Project (Andréfouët et al., 2006b), hereafter "G1", "G2", "G3", "G4" and "G5" from the coarsest to the finest description; 2) a coarse classification of substrate type, hereafter "S"; and 3) a coarse classification of benthic cover, hereafter “B”. Habitat segments that had the same classification at all levels were merged.

Finally, I created an illustrated habitat typology following my classification, with photographs, to document the diversity of coral reef habitats in the Madang Lagoon for future reference.

Maps for each of the seven classification levels, the full classification itself, and the typology are shown in **Appendix 2**.

Biodiversity surveys

Biodiversity surveys for macro-algae, fish, corals and macro-invertebrates were conducted during the Papua Niugini Expedition from October to December 2012 by expert taxonomists for each taxa (respectively Prof Claude Payri, Dr Ronald Fricke, Dr Francesca Benzoni and Dr Jean-Louis Menou). For macro-algae, 29 sites were visited in the Madang Lagoon, with experts recording a total of 202 different species records. For corals, 19 sites were visited, with 280 species recorded. For fish, 35 sites were visited, with 588 species recorded. For macro-invertebrates, 20 sites were visited, with 295 species recorded (**Figure 3.1**). Fifteen sites were visited, at which all four taxa were recorded. Sites were chosen to cover a diversity of marine habitats. These surveys were fairly unique in the sense that taxonomic inventories generally include fewer biological groups (often fish and corals only), or groups that have been sampled at different stations throughout the targeted area in the course of non-related research programs (but see Cleary et al., 2008, Dalleau et al., 2010, Jimenez et al., 2012). I used the species lists produced for each taxa at each sampled site in Chapter 6.

Fish data collection

At 19 sites (**Figure 3.1**), a visual fish census was conducted *in situ* by snorkeling at a depth between 0-4 m, during 45-80 minutes. At 14 of the 19 sites, rotenone and hand nets were used to collect small and cryptic species. At 16 of the 19 sites, visual census data was complemented with an analysis of all high definition photographs taken opportunistically *in situ* during the habitat census (see “Marine habitat mapping” section above). Eight sites in total comprised a full census with *in* *situ* observations, observations from photographs, and specimen collection. At the remaining 19 sites, fish species were identified based on photographs only (18 sites), or specimen collection (1 site). Identification of specimens and photographs to species level was done, based on the gross morphology and using reference literature. The final fish species list was compiled from underwater visual census, photographic records and laboratory identification of specimens (Fricke et al., 2014).

Macro-algae, corals and macro-invertebrates data collection

At each site (**Figure 3.1**), taxonomists sampled their taxa of interest (macro-algae, corals or macro-invertebrates) during 60 to 80 minute SCUBA dives, from the surface down to 50 m. The presence of each species was recorded, and photographs were taken *in situ* for all three taxa. Macro-invertebrate specimens and fragments of coral colonies were collected when species identification underwater was difficult. Macro-algae specimens were systematically collected for the voucher and tissue collection. Digital photographs were later analysed to confirm preliminary *in situ* records. For macro-algae, identification to the species level was done on the basis of gross morphology and histology, and using reference literature. DNA analyses were also conducted for various groups of macro-algae to assist further with species identification. For corals, identification of collected specimens to species level was done on the basis of skeleton morphology, using reference literature and, whenever possible, original species descriptions and type material illustrations. Species records *in situ* were combined with species identified with photographs and laboratory analyses to produce final species lists at each site.

Fisher surveys[1] (fishing activity)

For Chapters 4 and 6, I produced and used maps of current fishing grounds for Riwo fishers at the time of the survey, with information related to the fishing trips each fishing ground represented.

[1] The study involving fisher surveys was granted James Cook University Human Research Ethics Approval H4766.

Fisher surveys were conducted in all hamlets and villages constituting the Riwo community (**Figure 3.1**), mostly in Tok Pisin language. Local Bel language or English were also used occasionally). One fisher per boat returning to the main landing sites around Riwo was surveyed for five minutes on average (**Appendix 3**). I recorded gender, transport, gear type, trip duration (in hours), catch weight (in kilograms), and number of fishers contributing to the catch. Fishers were also asked to draw the area where their fishing took place, using high-resolution satellite imagery (1:15,000). Over 20 days, information was recorded from 68 fishing crews, or an estimated 14% of Riwo fishers who sell their catch.

All maps from the fishing surveys were digitised manually in ArcGIS (ESRI, 2010).

Household surveys[1] (perceived ecosystem services and benefits)

To map how and why Riwo people value different areas they visit in the Madang Lagoon, I adapted the participatory GIS methods described in Raymond et al. (2009) to the context of the Madang Lagoon. I surveyed heads of households through opportunistic sampling in all main villages, hamlets, and islands making up the Riwo community (**Figure 3.1**). I targeted heads of households whose main occupation was fishing (males and females) for their good spatial knowledge of the marine environment (see **Appendix 5** for more details on the representativeness of my sample). Ninety percent of respondents themselves fished as a main cash activity.

In November 2012, I held open meetings with the Riwo community to explain the project, engage with community members, and answer questions. Once the project was approved by chiefs and residents of the targeted villages, I held four preliminary focus groups (fishers and non-fishers, females and males) to identify the perceived benefits provided by the Madang Lagoon to the Riwo community. I separated these groups to assess similarities and differences in the ways they benefit from their environment. Picture cards illustrating different benefits (**Appendix 4**) adapted from the list of “landscape values” in Brown (2004) were prepared *a priori* to help with the discussions. Images were obtained from the

[1] The study involving household surveys was granted James Cook University Human Research Ethics Approval H4766.

Internet and chosen to be as culturally relevant as possible. I asked participants to identify and discuss the different reasons why people access their marine environment in the community. Cards were used only at the end of the discussion to refine the list and ensure that no benefits were missing. The compiled list of benefits (**Table 3.1**) was used to assist in the survey described below.

The household surveys had two parts: a structured socioeconomic questionnaire, and participatory mapping of important places. The socioeconomic questionnaire (**Appendix 4**) consisted of general questions about the household, fishing habits, and consumption of fish and other seafood. The participatory mapping of important places is described in more details below. Full surveys lasted one hour on average and one map was produced for each household. Over 20 days, 52 households were surveyed, corresponding to 17% of Riwo households (National Statistical Office of Papua New Guinea, 2002).

For Chapters 4, 5 and 6, I produced and used maps of areas accessed and valued by Riwo community members for various perceived ecosystem services and derived benefits, from the participatory mapping exercise. To do so, I adapted the participatory GIS methods described in Raymond et al. (2009) to the context of the Madang Lagoon.

The participatory mapping exercise (**Appendix 4**) consisted of three parts or "games" designed to: 1) identify relevant benefits and rank them in order of importance; 2) map where members of households would regularly go in the Lagoon for specific benefits, and; 3) quantify how important it was to access each place for these benefits (**Plate 1**). The first game was a "card game", involving a non-spatial activity in which respondents were asked why they visited the lagoon. All the picture cards representing the benefits identified in focus groups were presented and explained. Respondents were asked to select cards that represented benefits that were important to their households, then to rank cards in order of importance for the households' day-to-day life. The purpose of this ranking was to determine the order that the benefits would be introduced for discussion and mapping in the subsequent games (i.e. the most important benefits would be discussed first).

Table 3.1. Main reasons identified by local coastal communities to visit certain places in the Madang Lagoon, with associated types of ecosystem service and use categories.

Code	Services and benefits (reason for accessing the marine environment)	Ecosystem service type	Use category
FI	Fishing (harvest of marine resources for income and food)	life sustaining, economic	extractive
RI	Perceived biological richness	biological richness	non-extractive
AE	Aesthetic enjoyment	aesthetic	non-extractive
RE	Recreation (picnic and fire on the beach, swimming, and playing)	recreation	close to non-extractive
TM[1]	Traditional medicine (collecting shellfish, seaweed, or swimming near healing stones)	therapeutic and cultural	extractive and non-extractive
ED	Education and knowledge sharing (educating the youth about fishing, marine hazards)	learning	virtually non-extractive
LI	Lime material to chew with betel nuts (collecting dead or live shells and coral, later crushed for chewing)	economic and cultural	extractive
SP	Spiritual value (church or cemetery, spirits, and history of ancestors)	spiritual	non-extractive
WR[2]	Wreck (sunken ships or planes used for fishing, tourism, and non-tourism diving)	life sustaining, economic, recreation	extractive and non-extractive
TO[2]	Tourism (guiding tourists, collecting diving fees)	economic	non-extractive

The second game was a “drawing game”, in which respondents were asked to draw on a high-resolution satellite image (1:15,000) the main regions (polygons) in the lagoon that they visited for each benefit selected in the card game. The highest ranked benefit was mapped first and the lowest last. Once all regions visited for a particular benefit were drawn, the third game, a “token game”, began. In this game, respondents were asked to distribute tokens (shells or stones) among the regions drawn for each benefit, to reflect the relative importance of accessing each region for the benefit of interest. More tokens on a region indicated higher importance. Tokens were distributed according to the highly variable number of regions drawn, using the following general rule of thumb: for each household and for a

[1] The traditional medicine benefit (TM) corresponds to both non-extractive and extractive activities. Therefore, it was separated into two distinct benefits for analyses: TM (non-extractive) and TM (extractive).
[2] These benefits were not used for analyses because they were identified by fewer than 10% of people and overlapped with other benefits (wrecks overlapped with tourism and some fishing)

given benefit, I multiplied the number of regions drawn by two, then rounded the resulting number up to the next multiple of five (e.g. if three polygons were drawn, ten tokens were given). To give the same importance to all households regardless of the number of regions identified, all scores for each household were then standardised to a scale of 0-100 for a given benefit, where 100 was the total number of tokens distributed to the household for a given benefit. Surveys lasted one hour on average and one map was produced for each household, with a different colour for each benefit.

All maps from the household surveys were digitised manually in ArcGIS (ESRI, 2010).

Plate 1. Household surveys. Upper left photograph: the “card game”. Upper right: the “drawing game”. Bottom left: the “token game”. Bottom right: recording the position of the household. Photos by Mélanie A. Hamel.

Prerequisites for systematic conservation planning

Reserve design

To select candidate reserves based on biodiversity and socioeconomic cost proxies, I used the Marxan software (Ball et al., 2009) a decision support tool commonly used for systematic conservation planning. Although Marxan was briefly described, and already used in Chapter 2, this section provides further details necessary to fully comprehend the analyses conducted in the following chapters. Marxan feeds a simulated annealing algorithm with spatial data on the features to protect (here, biodiversity proxies), and on the cost of protecting each area (here, socioeconomic cost proxies). With this information, the algorithm finds optimal solutions: the smallest possible reserve systems that meet all conservation objectives under cost constraints. There are two typical reserve selection problems: the "minimum set" reserve design problem, which aims to capture a set amount of biodiversity for the least cost; and the "maximum coverage" problem, which aims to capture as much biodiversity as possible below a fixed budget. Marxan is generally used to solve the "minimum set" reserve design problem but can be configured to solve the "maximum coverage" problem.

Planning region and planning units

For conservation planning purposes and spatial prioritisation performed in Chapters 4, 5 and 6, the boundaries of the planning region and the shape and size of planning units (basic spatial units at which a conservation decision is being made) must be clearly defined.

Because the Riwo waters were surveyed for all aspects examined in the thesis (i.e. coral reef habitats, biodiversity, fishing, and perceived ecosystem services and benefits), the planning region was first broadly defined as the extent of marine habitats and islands within the section of the Madang Lagoon used by the Riwo community (roughly corresponding to their customary tenure). I later refined the boundaries of my planning region by overlaying a grid of planning units on the map of marine habitats and islands for the Madang Lagoon, and the spatial footprint of my socioeconomic dataset of interest. For Chapter 4, the 312 planning units intersecting both marine habitats and all fishing grounds mapped in the fisher surveys constituted the planning region. For Chapters 5 and 6, the 319 planning units intersecting both marine habitats and areas accessed for the whole range of ecosystem services and benefits mapped in the household surveys constituted the planning region. **Figure 3.2** shows all the planning units used in Chapters 4, 5 and 6.

Two criteria determined the size of planning units. First, single or grouped planning units needed to be small enough to facilitate local management while allowing use of alternative areas. I used the smallest zone of the Wildlife Management Areas (17 ha) as a reference. Second, planning units needed to be large enough to account for possible errors in mapping. Fishers were able to map specific features such as wrecked ships and planes with an accuracy of between 5 m and 200 m. Therefore to minimise mapping errors, I chose to define planning units of 300 m by 300 m or 9 ha. I used square planning units but marginal ones were trimmed to the Madang Lagoon boundaries.

Spatial layers of conservation features and conservation costs

For the conservation planning exercises and analyses performed in Chapters 4, 5 and 6, I incorporated all data (i.e. marine habitat maps, species lists from the biodiversity surveys, fishing grounds from the fisher surveys, and areas accessed for ecosystem services and derived benefits from the household surveys) into a geographic information system. Then I interpolated relevant information into all planning units constituting the planning region (**Figure 3.2**) to create spatial data layers that can be used in conjunction with reserve selection algorithms. I described the methods used to interpolate each relevant spatial layer in each chapter. All spatial layers were created using a combination of R (R Development Core Team, 2008) and ArcGIS (ESRI, 2010).

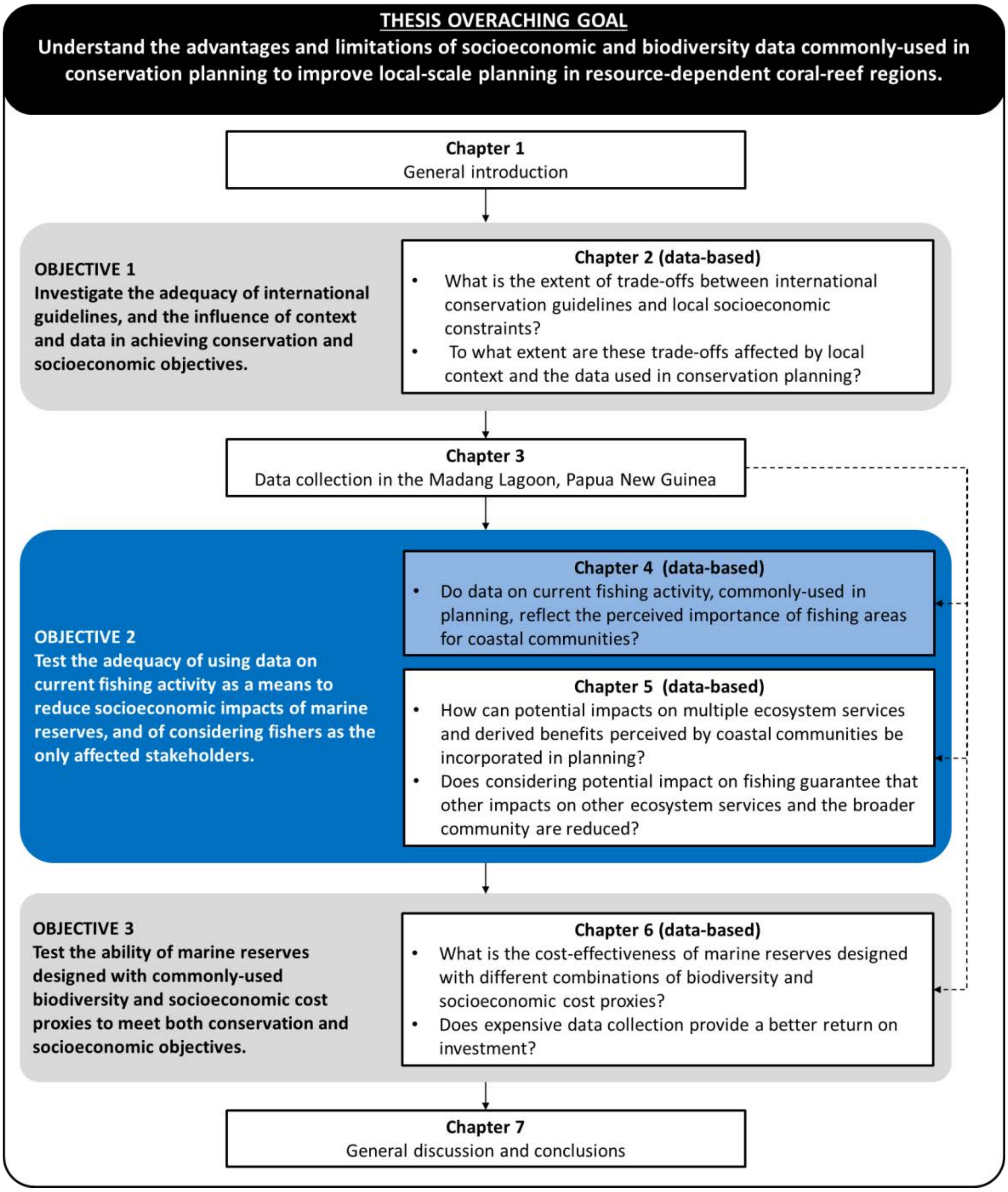
Figure 3.2. Location of the planning region (area of the Madang Lagoon used by the Riwo community) and planning units used in Chapters 4, 5, and 6.

Chapter 4. [1]

The importance of fishing grounds as perceived by local communities can be undervalued by common measures of importance used in conservation planning

In this chapter, I investigate the adequacy of using information on current fishing activity in coral-reef conservation planning to minimise negative socioeconomic impacts of marine protected areas. I compare the spatial distribution of commonly-used socioeconomic costs such as proximity to landing sites, fishing effort, or catch per unit effort, with the importance of fishing grounds as perceived by the local community.

[1] This chapter was modified to address comments from reviewers for Conservation Letters on the submitted version, and slightly modified from the journal article version to avoid redundancies. The submitted version is: **M. A. Hamel**, R. L. Pressey, S. Andréfouët and L. S. Evans (in revision) Commonly-used surrogates of opportunity costs to fishers can undervalue the importance of fishing areas for local communities. Conservation Letters. Details on data collection are described in Chapter 3, and some of the supplementary material was directly included in the main text.


THESIS OVERACHING GOAL
Understand the advantages and limitations of socioeconomic and biodiversity data commonly-used in conservation planning to improve local-scale planning in resource-dependent coral-reef regions.
Chapter 1
General introduction
OBJECTIVE 1
Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.
Chapter 2 (data-based)
• What is the extent of trade-offs between international conservation guidelines and local socioeconomic constraints?
• To what extent are these trade-offs affected by local context and the data used in conservation planning?
Chapter 3
Data collection in the Madang Lagoon, Papua New Guinea
OBJECTIVE 2
Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders.
Chapter 4 (data-based)
• Do data on current fishing activity, commonly-used in planning, reflect the perceived importance of fishing areas for coastal communities?
Chapter 5 (data-based)
• How can potential impacts on multiple ecosystem services and derived benefits perceived by coastal communities be incorporated in planning?
• Does considering potential impact on fishing guarantee that other impacts on other ecosystem services and the broader community are reduced?
OBJECTIVE 3
Test the ability of marine reserves designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.
Chapter 6 (data-based)
• What is the cost-effectiveness of marine reserves designed with different combinations of biodiversity and socioeconomic cost proxies?
• Does expensive data collection provide a better return on investment?
Chapter 7
General discussion and conclusions

4. The importance of fishing grounds as perceived by local communities can be undervalued by common measures of importance used in conservation planning

Abstract

Approaches to design marine protected areas must account for the dependence of people on marine resources. Otherwise, the implementation of such protected areas will likely incur negative socioeconomic impacts and compliance will be low. Conservation planners measure the negative impact on (or "cost" to) stakeholders of conserving an area, proportionally to the importance of this area to these stakeholders. In marine planning, the importance of fishing areas to fishing communities is often measured with proxies of fishing opportunities, such as proximity to shore or empirical data on current fishing activities. It is assumed that avoiding fishing opportunity costs will reduce negative impacts of protected areas on coastal communities. But does measuring the importance of fishing areas through proxies of fishing opportunities always reflect their importance for fishing as perceived by the community? In the Riwo community of the Madang Lagoon (Papua New Guinea), I surveyed fishers (n=68) at landing sites to map current fishing activities over 20 days. I also surveyed households (n=52), using participatory mapping techniques, to identify areas of perceived importance for fishing. The importance of fishing areas according to proxies of fishing opportunities were not correlated with their importance as perceived by the community. To illustrate the implications for planning, I compared marine reserves designed to minimise fishing opportunity cost proxies, and costs related to the perceived importance of fishing areas. I suggest that both types of costs should be used as complementary measures: current fishing activity to maintain food and income security, and perceived importance of fishing areas to maintain social values.

Introduction

Applications of systematic conservation planning to marine ecosystems are now widespread (e.g. Leslie, 2005, Álvarez-Romero et al., 2013). Increasingly, these approaches recognise that successful conservation and management of natural resources rely on a good understanding of the social-ecological system of interest, and the active involvement of local stakeholders in decision-making (e.g. Lundquist and Granek, 2005, Klein et al., 2008, Ban et al., 2013, Kittinger et al., 2014, Le Cornu et al., 2014). Socioeconomic assessments at the beginning of a project can help to engage stakeholders, understand opportunities for and constraints on implementation, and reduce the negative impacts on people of restrictions on resource use.

Systematic conservation planning attempts to minimise negative impacts of conservation actions on stakeholders by considering social and economic "costs" in the design of protected areas (Naidoo et al., 2006, Ban and Klein, 2009). In marine social-ecological systems, the most common method to estimate costs has been through opportunity costs to fishers, often estimated with empirical data or models of catch (e.g. amount of resource caught, monetary value of catch), effort (e.g. number of boats or fishers, distance to ports, population density), or catch per unit of effort (e.g. Ban and Klein, 2009, Weeks et al., 2010, Adams et al., 2011, Deas et al., 2014).

More broadly, ecosystems provide a diversity of tangible and intangible services (and disservices) to people (Millennium Ecosystem Assessment, 2003). These services benefit different people in different ways (Daw et al., 2011a, Fisher et al., 2014), leading to different perceived importance (or subjective "value") of services and associated places (Lele et al., 2013). This chapter focuses on the provisioning services of fisheries. The importance of or attachment to a fishing place for a fisher may well be influenced by thoughts, feelings, and beliefs over and above, or even instead of, catch itself (e.g., how reliable, safe, or beautiful the fishing ground is). Using catch data to measure the utilitarian value of fishing places for coastal communities such as food or income is critical to minimise conflicts with conservation and maintain livelihoods. However, catch data could ignore other factors defining the importance of fishing places, such as place attachment, that could also conflict with conservation actions (see Brown and Raymond, 2007 on analogous landscape values). Here I quantify and compare, at the scale of one coastal community, the importance of fishing areas as perceived subjectively by households (hereafter “perceived fishing value”), and compare them with the importance of fishing areas based on empirical data on fishing opportunities (hereafter “current fishing activity”).

Specifically, I aim to answer the following questions:

1) Does data on current fishing activity always reflect the perceived value of fishing grounds for local coastal communities?
2) Do reserves designed to minimise impacts on current fishing activity also reduce costs based on the perceived fishing value of areas, and *vice versa*?
3) If data on current fishing activity does not reflect the perceived value of fishing grounds, should we attempt to reconcile the two or choose only one?

To address these questions, I focused on the Riwo community of the Madang Lagoon in Papua New Guinea.

Materials and methods

General methods for this chapter are summarised in **Figure 4.1**.

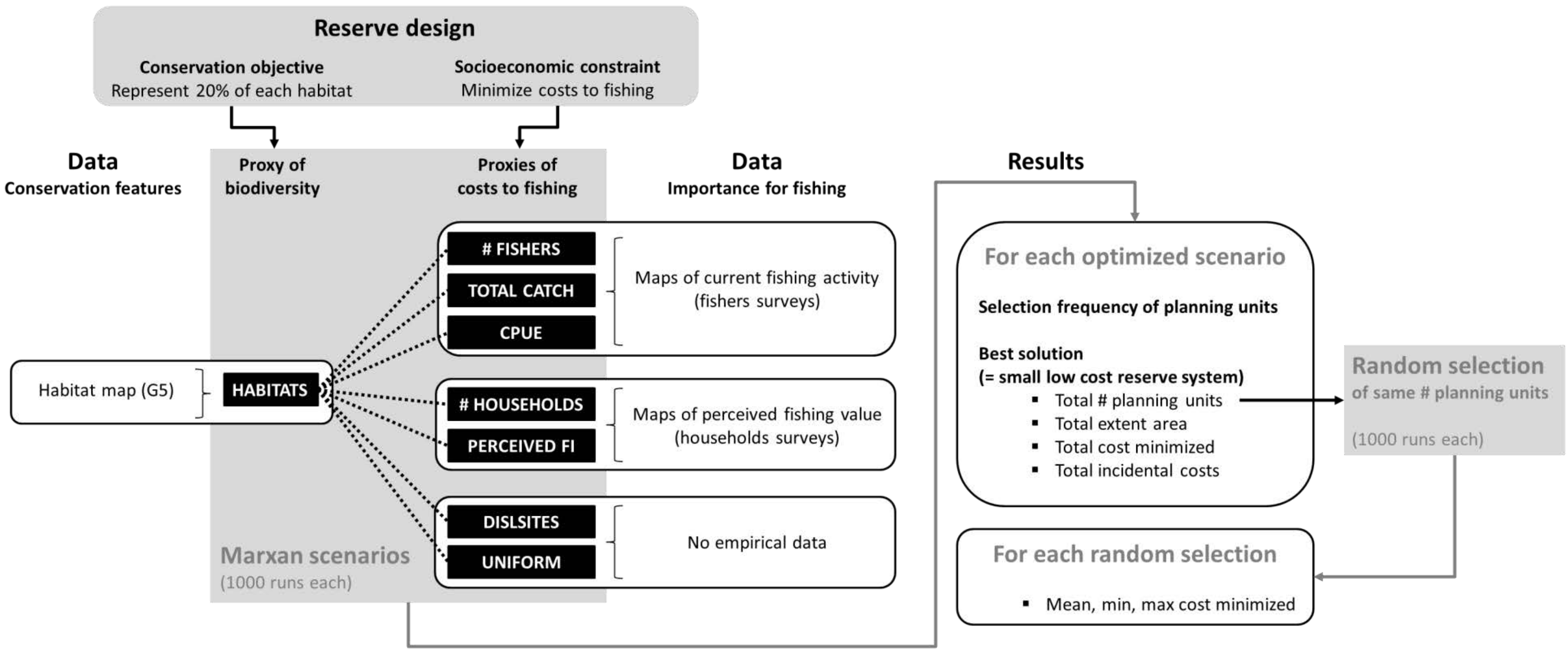


Figure 4.1. Summary of the conservation planning approach and of the scenarios contrasted in Chapter 4. “# FISHERS” refers to the number of fishers visiting each planning unit. “TOTAL CATCH is the total catch (kg). “CPUE” is the average catch per unit effort (kg/person/h) across fishers. “# HOUSEHOLDS” is the number of households valuing the planning unit for fishing. “PERCEIVED FI” is the sum of all recorded perceived fishing values based on the number of tokens allocated. “DISLSITES” is proximity to landing sites, to reflect findings that opportunity costs are higher when reserves are closer to villages. “UNIFORM”, a reference cost, gives a uniform importance to all planning units.

Habitats as proxies of biodiversity

In this chapter, I used fine geomorphologic habitats, as described in the G5 level of classification (details in Chapter 3), as a conservation feature to protect. Using marine habitats as conservation features in data-poor regions is relevant because, unlike species data, complete maps of consistently-defined habitats are often available. I created a spatial layer for spatial prioritisation, containing the necessary information on habitats. For each of the 312 planning units within the planning region (details in Chapter 3), I computed the type and extent of all habitats contained in the planning unit.

Importance of areas for fishing as a proxy for conservation cost

In this chapter, I used the maps and associated data on current fishing activity (fisher surveys, details in Chapter 3), and those on perceived fishing value (household surveys, details in Chapter 3) to define the potential costs of protected areas to fishing. I assumed that protecting areas deemed important for fishing will incur a high cost to fishing, and *vice versa*. Therefore, I defined costs to fishing based on the importance of areas for fishing. I created spatial layers for spatial prioritisation, containing information on the costs to fishing, of reserving each planning unit. To do this, I derived various proxy measures of the importance of all areas drawn in the surveys for fishing, for both datasets. Then I interpolated these measures to the grid of planning units constituting the planning region.

Three proxies were based on current fishing activity (**Figure 4.2**, **Table 4.1** and **Table 4.2**). For each planning unit:

- # FISHERS: the number of fishers visiting;
- TOTAL CATCH: the total catch (kg);
- CPUE: the average catch per unit effort (kg/person/h) across fishers.

Two proxies were derived from the perceived fishing value data (**Figure 4.3**, **Table 4.1** and **Table 4.2**). For each planning unit:

- # HOUSEHOLDS: the number of households valuing the planning unit for fishing;

- PERCEIVED FI: the sum of all recorded perceived fishing values based on the number of tokens allocated.

I also derived two other proxies that were not based on empirical data (**Table 4.1** and **Table 4.2**). For each planning unit:

- DISLSITES: proximity to landing sites, to reflect findings that opportunity costs are higher when reserves are closer to villages (e.g. Green et al., 2009, Weeks et al., 2010, Giakoumi et al., 2013, Mazor et al., 2014). To measure proximity, I calculated the sum of distances between each planning unit and each of the main Riwo landing sites, then calculated cost with a negative linear function of the summed distances. I used the mean of all distances to reflect the importance of each planning unit for fishing to the broader community of fishers. Indeed, in Riwo, landing sites correspond to specific households that specific fishers access, rather than, for example, a beach that all fishers access.
- UNIFORM: uniform importance of all planning units, as a reference.

All proxies combined data on all gears, transport methods, and targeted resources. All spatial layers for the seven proxies were normalised as percentage of maximum to allow for comparisons between proxies. As a result, the importance of each planning unit according to each proxy of interest varied between 0 and 100 (**Figure 4.4**).

Relationships between pairs of proxies derived from data on current fishing activity (fisher surveys), perceived fishing value (household surveys), and not derived from empirical data, were measured with Pearson's correlation coefficient. I did not investigate non-linear relationships, which would have little value for systematic conservation planning applications (**Appendix 6: Figure 1**).

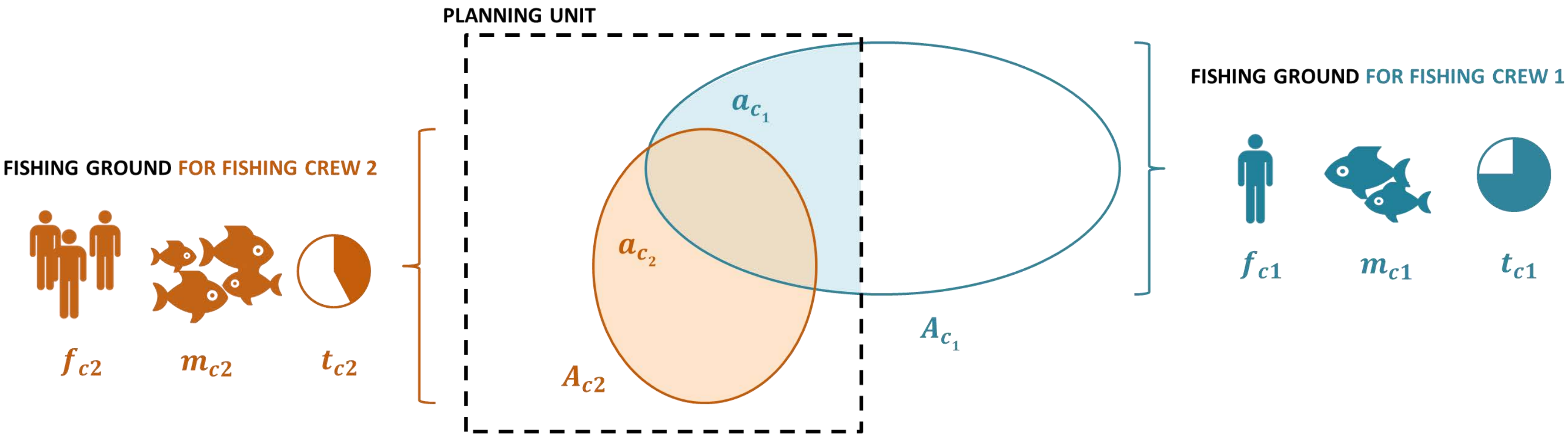


Figure 4.2. Schematic representation of all variables used to derive the cost layers based on data on current fishing activity (fisher surveys). Two fishing grounds (1, in blue, and 2, in orange), each delineated by a different fisher representing his/her crew, are portrayed. Each fishing ground is associated with the number of fishers (respectively $f_{c1}$and f_{c2}) in the fishing crew, the total weight of fish caught by the crew (respectively $m_{c1}$and m_{c2}), the time spent fishing by the crew (respectively $t_{c1}$and t_{c2}), and the total area covered by the crew (respectively $A_{c1}$and A_{c2}). The importance of a planning unit is calculated as described in **Table 4.1** and **Table 4.2**. One planning unit is represented, covering partially the area a_{c1} of fishing ground 1 and entirely fishing ground 2 ($a_{c2} = A_{c2}$).

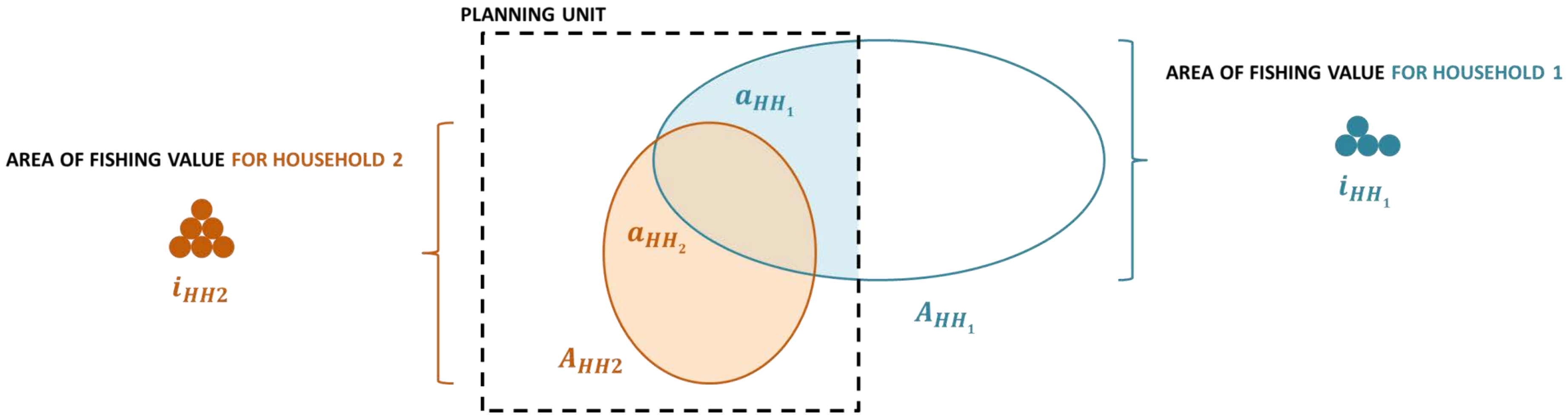


Figure 4.3. Schematic representation of the variables used to derive spatial datasets for perceived fishing value, showing the importance of planning units. Two areas of fishing value are presented, each delineated by a different head of household representing his/her household (1, in blue, and 2, in orange). Each area of fishing value is associated with its perceived importance for fishing (respectively i_{HH1} and i_{HH2} based on the number of tokens distributed by the respondent), and its total area (respectively A_{HH1} and A_{HH2}). The importance of a planning unit is calculated as described in **Table 4.1** and **Table 4.2**. One planning unit is represented, covering partially area a_{HH1} of the polygon delineated by household 1 and entirely the polygon delineated by household 2 ($a_{HH2} = A_{HH2}$).

Table 4.1. Proxies of the importance of planning units for fishing, derived from data on current fishing activity (fisher surveys), perceived fishing value (household surveys), and not derived from empirical data. Variables are detailed in **Table 4.2**.

Dataset	Proxy code	Formula[1] (for each planning unit)	Description
Current fishing activity (fisher surveys)	# FISHERS	$x = \sum_{c=1}^{C} f_c$	Number of fishers visiting the planning unit
	TOTAL CATCH	$x = \sum_{c=1}^{C} w_c . m_c$ with $w_c = \frac{a_c}{A_c}$	Total catch (kg). Assumes productivity is constant in all parts of the fishing ground.
	CPUE	$x = \frac{1}{C} . \sum_{=1}^{C} CPUE_c$ with $CPUE_c = m_c . \frac{1}{f_c} . \frac{1}{t_c}$	Average of all CPUEs (kg/person/h) recorded in the planning unit.
Perceived fishing value (household surveys)	# HOUSEHOLDS	$x = H$	Number of households
	PERCEIVED FI	$x = \sum_{h=1}^{H} WFIV_h$ with $WFIV_h = v_h . w_h$ and $w_h = \frac{a_h}{A_h}$	Sum of all fishing values recorded in the planning unit. Fishing value of one polygon was based on the number of tokens associated with the polygon. Since the total allocated set of tokens varied between households, each set was scaled to 100 for analyses.
No empirical data	DISLSITES	$x = SUMDIST_{l_{max}} - SUMDIST_l$ with $SUMDIST_l = \sum_{l=1}^{L} d_l$ and $SUMDIST_{l_{max}}$ =maximum value for $SUMDIST_l$	Negative linear function of the sum of all distances from the planning unit to the main landing sites (km)
	UNIFORM	$x = 100$	

[1] Variables are described in **Table 4.2**

Table 4.2. List of variables used to calculate the importance of each planning unit for fishing, in relation to the different datasets they were derived from.

Dataset	Variable name	Description
Current fishing activity (fisher surveys)	C	total number of fishing crews visiting the PU (0 ≤ N≤ 68)
	f_c	number of fishers per crew c visiting the PU
	w_c	proportion of fishing ground for crew c covered by the PU
	a_c	area (m2) of fishing ground for crew c covered by the PU
	A_c	total area (m2) of fishing ground for crew c
	m_c	total weight (kg) caught by crew c in visited fishing grounds
	t_c	time (hours) spent fishing by crew c
Perceived fishing value (household surveys)	H	number of households “using” the PU (0 ≤ H≤ 52)
	v_h	fishing value (number of tokens scaled to 100 arbitrary units) attributed to the zone used by household h within the PU
	w_h	proportion of the zone used by household h covered by the PU
	a_h	area (m^2) of zone used by household h covered by the PU
	A_h	total area (m^2) of zone used by household h
No empirical data	L	total number of main landing sites (L=9)
	d_l	Euclidian distance (m) between landing site l and PU, estimated geographically without consideration of possible obstacles

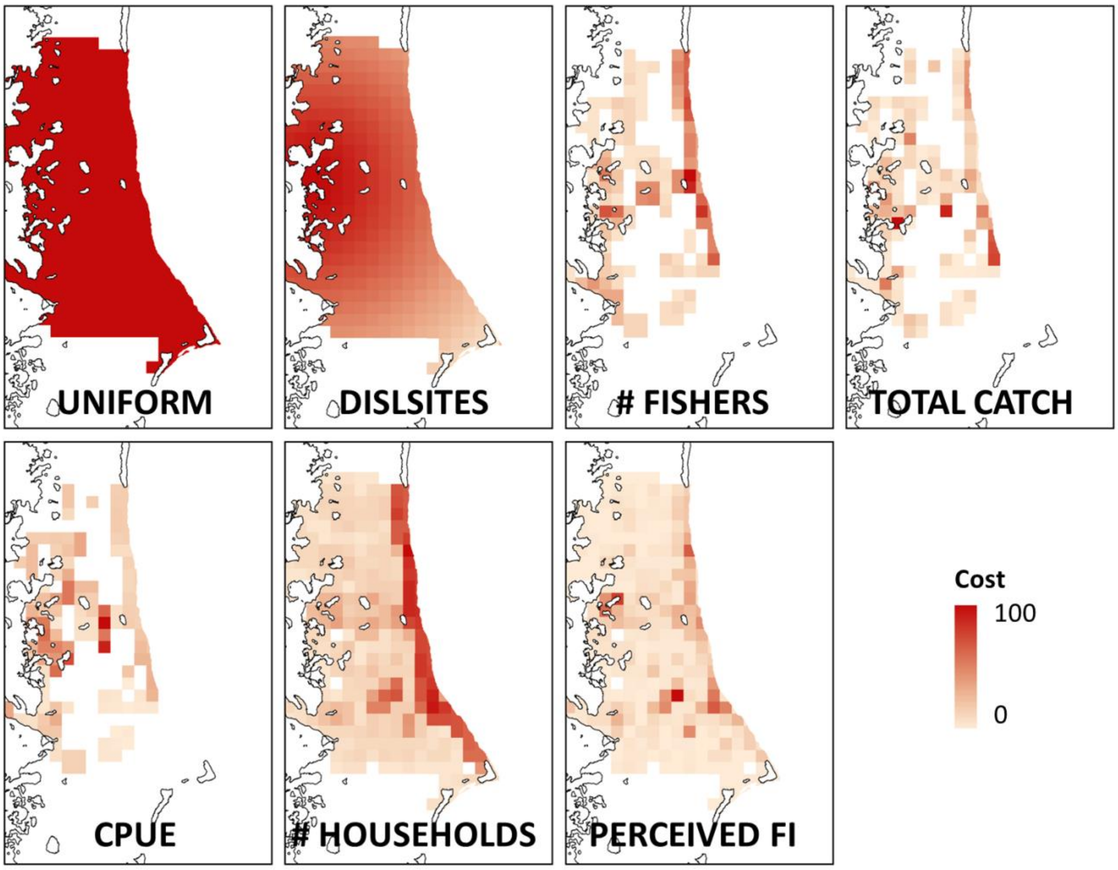


Figure 4.4. Proxies of the socioeconomic cost of reserving each planning unit, derived from measures of the importance of each planning unit for fishing (normalised as percentages of maximum). The top row shows uniform importance (UNIFORM), proximity to landing sites as a proxy (DISLSITES), and two proxies derived from data on current fishing activity: the number of fishers visiting each place (# FISHERS) and the total weight of catch (TOTAL CATCH). The bottom row shows the averaged catch per unit effort (CPUE, derived from data on current fishing activity) and two cost proxies derived from value data: the number of households valuing each place (# HOUSEHOLDS) and the value of each place for fishing measured with the summed number of tokens (PERCEIVED FI).

Spatial prioritisation

I used Marxan, the most commonly used systematic conservation planning optimisation algorithm (see Chapter 3 for details), to select cost-effective systems of marine reserves that meet conservation objectives while minimising conservation costs. I designed seven scenarios (one for each cost layer in **Figure 4.4**) to compare the configuration of candidate reserves obtained by minimising each of the cost proxies separately. All scenarios aimed to meet the same conservation objective: 20% of the total extent of each habitat in the study area, following a precautionary interpretation (as in Hamel et al., 2013) of Aichi target #11 (UNEP/CBD/COP/10/X/2, 2010). Intentionally ignoring existing Wildlife Management Areas, each prioritisation proposed candidate reserves that would restrict fishing activities, while representing each habitat to a specified level and minimising costs related to fishing to the Riwo community.

I ran the seven scenarios 1000 times each. For each scenario, I recorded the selection frequencies of each planning unit across the 1000 runs. I also identified the best (lowest-cost) solution across the 1000 runs. For each of the seven best solutions (one for each scenario), I recorded: the number of planning units selected as part of the reserve system, the total extent of the reserve system, the total cost of the proxy minimised, the total costs incurred incidentally for the other six proxies. Then I randomly selected planning units in the planning region 1000 times, using the seven values (i.e. numbers of planning units) recorded for the best solutions for comparison. Marxan's species penalty factor[1], was adjusted to ensure habitat objectives were always met. Default values were used for all other Marxan parameters.

Results

Fishing activities of the Riwo community are fully described in **Appendix 5**.

[1] The species penalty factor, or "spf", is a penalty added to the total score of the reserve system when the objective (here 20%) for a conservation feature (here, habitat) is not met. The highest the score of the reserve system, the less cost-effective the system is. A high spf forces the algorithm to find solutions that achieve objectives at all costs.

Proxies comparisons before reserve design

Proxies derived from data on current fishing activity (from the fisher surveys), perceived fishing value (from the household surveys), and not derived from empirical data, had distinct spatial patterns. Fishing grounds recorded in the fisher surveys were smaller and patchier than areas of fishing value delineated in the household surveys (**Appendix 6: Figure 2**), reflected in strongly right-skewed distributions and high proportions of zeroes in the costs of planning units (**Figure 4.5**).

Fisher surveys showed that 43% of planning units were fished. From my sample of 68 fishers, a maximum of 24 and average of 6 visited the same planning units. The maximum total catch in a planning units was 16.3 kg (2.1 kg on average) with most planning units (85%) having a total catch at less than 5 kg. The highest average CPUE among fished planning units was 2.9 kg/pers/h with most planning units (88%) having CPUE values less than 1.0 kg/pers/h.

From the household surveys, 94% of planning units were valued for fishing. The maximum number of households from my sample of 52 valuing a planning unit for fishing was 33. Most valued planning units (87%) had a fishery value (sum of standardised tokens) smaller than 40 units, with a maximum of 210 (17 on average).

Costs based on proximity to landing sites differed spatially from those based on current fishing activity and perceived fishing value (**Figure 4.4** and **Figure 4.5**). Although stronger correlations were observed between pairs of proxies derived from a same dataset, no correlations were significant across all combinations (p-value >0.05). More people valued and visited for fishing planning units covering the barrier reef, fringing reefs, and the main islands than other parts of the study area. Noteworthy was the high importance of two large patch reefs, *Yazi Tinan* and *Yazi Natun* (**Figure 3.1**), evident only from the data on perceived fishing value.

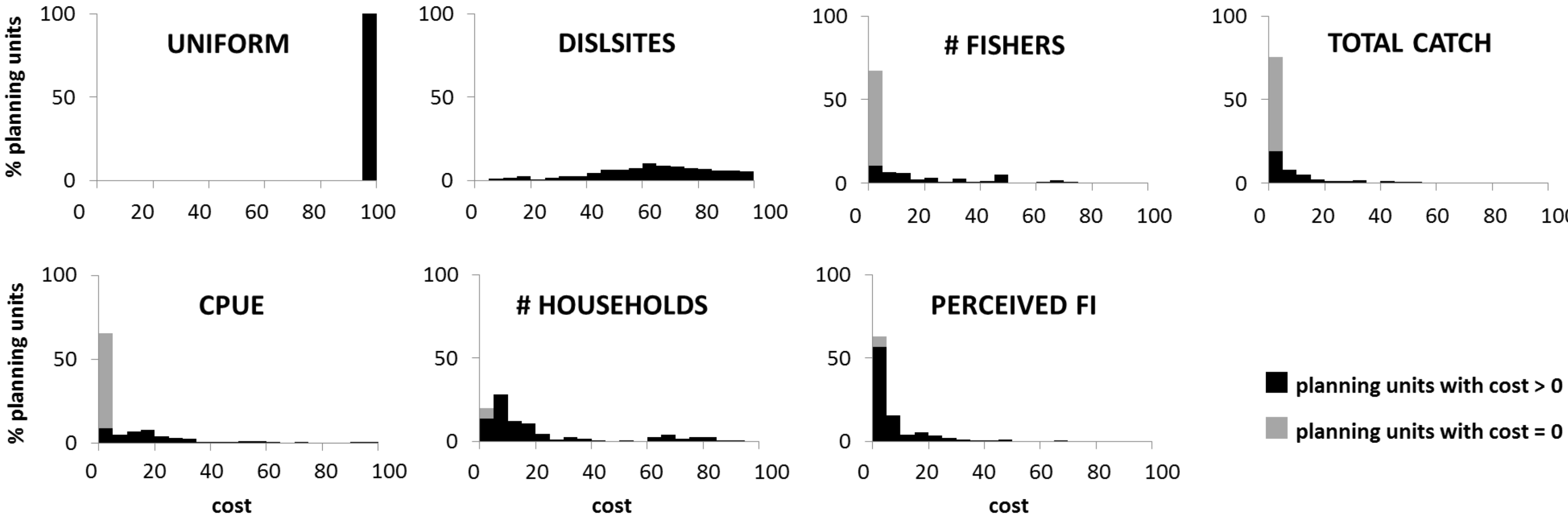


Figure 4.5. Distribution of cost values among planning units for each proxy. The grey parts of bars for catch and value indicate the percentages of planning units unfished or unvalued and having zero cost.

Spatial prioritisation

Selection frequencies (**Figure 4.6**) for the UNIFORM scenario varied only slightly between planning units, except for one place in the south-east that was always selected because it contained most of one habitat type. Smaller marginal planning units were selected less frequently because they contained smaller habitat extents at the same cost. Predictably, selection frequency in the DISLSITES scenario was higher in planning units more remote from landing sites.

Selection frequencies of scenarios using proxies based on current fishing activity were similar, but distinct from those based on perceived fishing value (**Figure 4.6**). Priorities based on current fishing activity reflected the preference of fishers for some fringing reefs and the barrier reef (see habitats map for classification G2 in **Appendix 2**), with planning units containing these formations never selected. Costs derived from current fishing activity were small for the widespread "deep lagoon" habitat, leading to selection of associated planning units with equal frequencies. Overall, selection frequencies of scenarios using proxies based on current fishing activity were dominantly intermediate or zero. This reflects the scope for higher-cost planning units to be left unselected, with representation objectives achievable through selection of lower- or zero-cost planning units at moderate frequencies.

Selection frequencies of scenarios using proxies based on perceived fishing value were low for planning units associated with the highly-valued barrier reef and *Yazi Tinan* and *Yazi Natun* reefs (**Figure 4.6**). Planning units at the northern and southern margins of Riwo's waters had low value-related costs and were selected frequently. Selection frequencies for PERCEIVED FI resembled those based on similarly skewed # FISHERS, TOTAL CATCH, and CPUE (**Figure 4.5**), except for occasional selections of costly planning units. This difference reflected fewer planning units with zero PERCEIVED FI and high number and spatial arrangement of planning units with very low costs. This difference between selections frequencies based on current fishing activity and those based on perceived fishing value was more pronounced for # HOUSEHOLDS because of the greater lack of very low-cost options to achieve objectives (**Figure 4.5**).

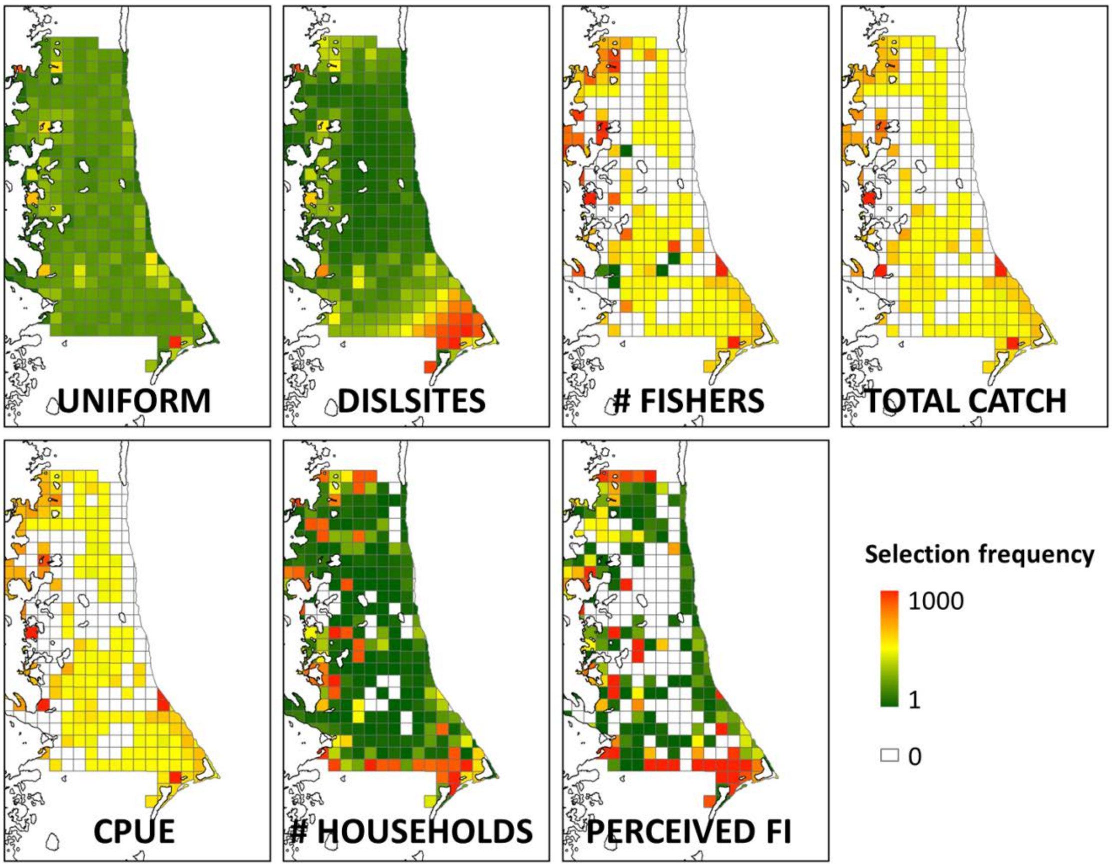


Figure 4.6. Selection frequencies, across 1000 runs, of planning units for all scenarios. Scenario names are indicated in capital letters, corresponding to respective cost layers in **Figure 4.4**. Planning units left white were never selected.

There were strong significant correlations (0.99 ≤ r ≤ 1.0, p-values <0.0001) between selection frequencies for scenarios using proxies based on current fishing activity. However, these selection frequencies were not significantly correlated with those based on perceived fishing value.

Best solutions for scenarios that minimised costs based on current fishing activity achieved conservation objectives with more planning units and larger total extents, but with very low costs compared to scenarios based on perceived fishing value (**Figure 4.7** an**d Appendix 6: Figure 3**). Scenarios based on catch data also gave the largest percentage reductions in total costs compared to random selections (**Figure 4.7**). As expected, the best solution for UNIFORM was not better than random. DISLSITES was moderately better. Across all scenarios, percentage reductions from random were larger for cost variables with more strongly right-skewed distributions (**Figure 4.6**), indicating greater potential for achievement of objectives at low cost.

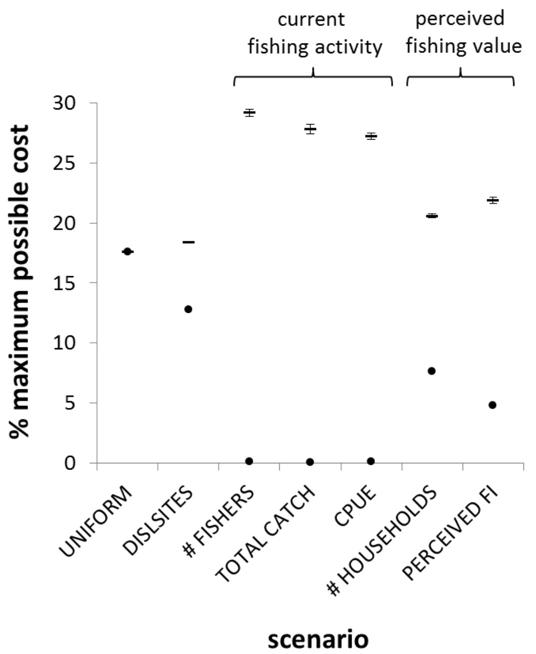


Figure 4.7. Best (lowest-cost) solutions for each scenario compared to random selections of the same number of places. Black dots indicate best solutions across 1000 runs. Black lines and error bars indicate mean cost from 1000 random selections and 95% confidence intervals, respectively. Maximum possible cost, for a given cost layer, is the sum of costs for all places.

The UNIFORM and DISLSITES scenarios had incidental costs related to current fishing activity and perceived fishing value much larger than minimum (**Figure 4.8**). The three scenarios based on current fishing activity had incidental costs related to perceived fishing value substantially larger than minimum (from 10.5% to 21.7% of the maximum possible cost, against 4.8 to 7.7% in scenarios that specifically aimed at minimizing costs related to perceived fishing value). The two scenarios based on perceived fishing value had incidental costs related to current fishing activity much larger than minimum (from 4.5% to 14.4% of the maximum possible cost, against 0.1 to 0.2% in scenarios that specifically aimed at minimizing costs related to current fishing activity).

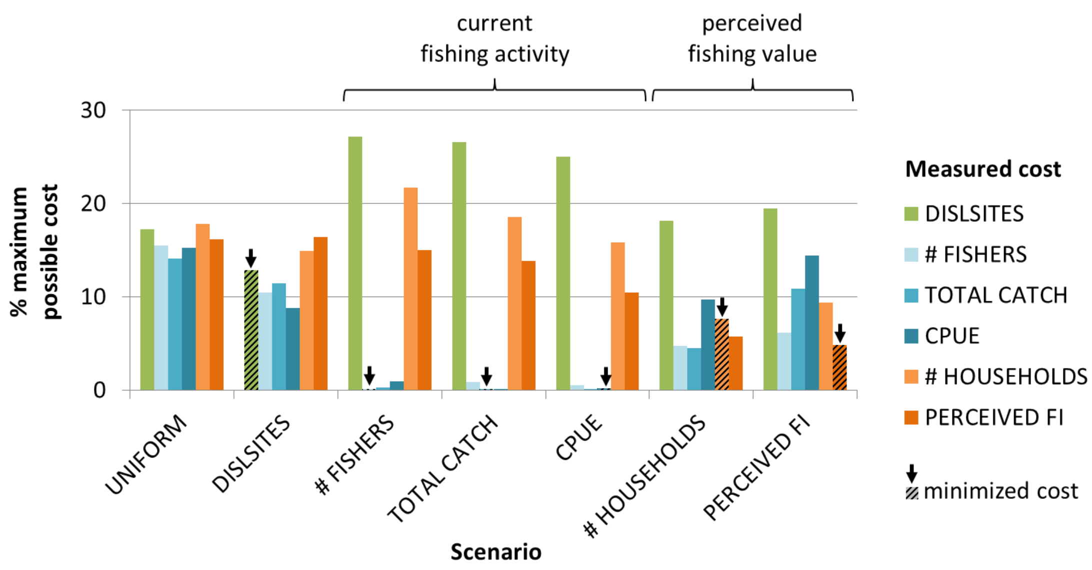


Figure 4.8. Incidental costs of the best (lowest-cost) solution for each scenario. For each scenario, coloured bars indicate the total cost of the solution for both the cost being minimised (hatched, with arrows) and the costs incurred incidentally. Maximum possible cost, for a given cost layer, is the sum of costs for all places.

Discussion

This chapter addresses a key challenge in systematic conservation planning: incorporating appropriate socioeconomic data in reserve design to reduce negative impacts on resource-dependent communities. I designed reserve systems to represent marine habitats while minimising opportunity costs based on: 1) proximity to landing sites, a common proxy of fishing effort; 2) the utilitarian value of fishing grounds, estimated empirically from data on current fishing effort; and 3) the importance of fishing grounds as perceived by communities. I found that these types of costs and the planning units selected to minimise them were not spatially correlated, and that they were not good proxies for one another.

All data collection was done over 20 days, allowing me to survey 52 households, and 68 fishers. While survey time span and sample sites may be considered short and small respectively by some, such data are not unusual in conservation planning. A longer and more comprehensive survey on current fishing activities may have yielded different results. However, the key goal of this chapter is not to show whether data on current fishing activity do or do not match the perceived importance of fishing ground by fishers, but that both datasets can provide very different pictures, mostly because of the typical variability in space and time of fishing data. I demonstrate why conservation planners need to be more wary about the type of data they base their decisions upon, especially in developing countries such as Papua New Guinea, where communities are heavily reliant on the marine environment for their livelihoods.

Opportunity costs based on current fishing activity, perception data, or proximity were different because they provide different types of information, but also because of their inherent characteristics and limitations. All three datasets can be valid for systematic conservation planning but the questions they help planners answer are different.

Data on current fishing activity are empirical, indicating the importance of planning units in providing fish, or the number of people who fish them, and serving as a proxy for the amount of protein-rich food and/or income where fishing is the dominant income-generating activity. However, such data are typically spatially patchy, with many zero or low values, and, when collected for planning purposes, do not always account for the diversity and high spatiotemporal variability of reef fisheries. In summary, minimising costs related to current fishing activity attempts to ensure short-term food or income security.

My data on perceived fishing importance indicate the subjective importance of fishing grounds to fishers. They identify not only fishing planning units important for providing fish, but also those valued for emotional attachment, aesthetics, safety, or ease of access. Because my surveys involved several parts, including the participatory mapping of valued fishing grounds (**Appendix 4**), I did not investigate the reasons why fishers valued each of their delineated fishing grounds. Overall, perception data were less aggregated spatially and, because they were less focused on short-term catches, they could also better reflect spatiotemporal variability of catch. However, values are subjective and may change with people's experiences. Minimising costs based on perceived fishing importance attempts to ensure that planning units perceived as important for diverse fishing-related reasons remain accessible. This is probably useful in avoiding negative social impacts of protection, improving social acceptance, compliance, and encouraging engagement in planning (Walmsley and White, 2003).

Proximity data are easily derived and assumed to indicate the importance of planning units in terms of the number of people who fish them. Minimising proximity costs attempts to ensure that fishing activities close to shore are maintained, but is not relevant in all contexts (Weeks et al., 2010). The complexity, small-scale, and variable nature of reef fisheries limits the use of such coarse proxies (Deas et al., 2014). More sophisticated proximity variables, such as one that considers accessibility of specific areas within the Lagoon, could be derived and tested. However, it is likely that these variables would not be relevant elsewhere, which would ultimately undermine their use as a proxy.

Designing reserves with the wrong cost layer has potentially serious implications both in terms of biodiversity protection and impacts of conservation on human populations. In my study area, proximity to landing sites was unrelated to fine-resolution patterns of resource use and had high incidental costs in terms of fishing activity and perceived importance. Considering only data on fishing activity could fail to address the broader spectrum of factors determining the fishing importance of planning units for local communities, as exemplified by my perception data. Failing to account for the range of potential impacts of reserves on people would risk poor social acceptance of and low compliance with conservation actions, reducing their effectiveness (West et al., 2006, Mascia et al., 2010). In cases where data collection is limited by a restricted budget, it will be especially critical to ensure that the type of data to be collected will effectively help achieve objectives for the planning exercise.

An alternative approach to choosing one cost layer would be to combine different "truths" to provide a more comprehensive view of how reserves can directly impact local fishing communities. Although proximity was not appropriate in my context, integrating costs based on current fishing activity and perceived fishing importance could help maintain food and income while minimising other fishing-related social impacts of conservation management. The variables used would have to be chosen carefully according to specific socioeconomic objectives. Analytically, recent software development can accommodate for multiple costs (Pauly et al., 2000, Watts et al., 2009) but integrating multiple costs is still an area of conservation planning that requires more research. The main challenges lay in integrating costs of different units (e.g. subjective value and catch per unit effort, in my case), and estimating the relative importance of each cost layer (i.e. weight) (Ban and Klein, 2009, Gurney et al., 2015). A practical approach would be to return to affected communities after data collection and preliminary reserve design, and consult with them about the different types of costs and related designs to determine the best strategy (i.e. Multiple costs? Which costs? Weight for cost layers?).

Indirectly, the process of collecting data on perceptions also promotes more comprehensive stakeholder participation and engagement through involvement of whole households and longer periods spent with respondents. Early involvement of stakeholders is key to effective planning, with benefits that include understanding concerns and requirements of affected communities and building trust (Pressey and Bottrill, 2009). Importantly, successful planning must ensure that communities understand and approve socioeconomic data that affect them, before the data are used. Stakeholders' feedback can help correct sampling biases, encourage consensus, and facilitate negotiations on implementation (Cash et al., 2003).

Finally, it is important to note that my approach was a theoretical exercise. For the purpose of this study, I designed marine protected areas with simplistic conservation objectives, and literally avoiding protection of the main fishing areas.

Ideally and more realistically, more conservation features would be incorporated, as well as information such as connectivity or larval spillover. More comprehensive socioeconomic assessments would also have been conducted, and multiple zones with a range of restrictions, rather than full reserves would be used.

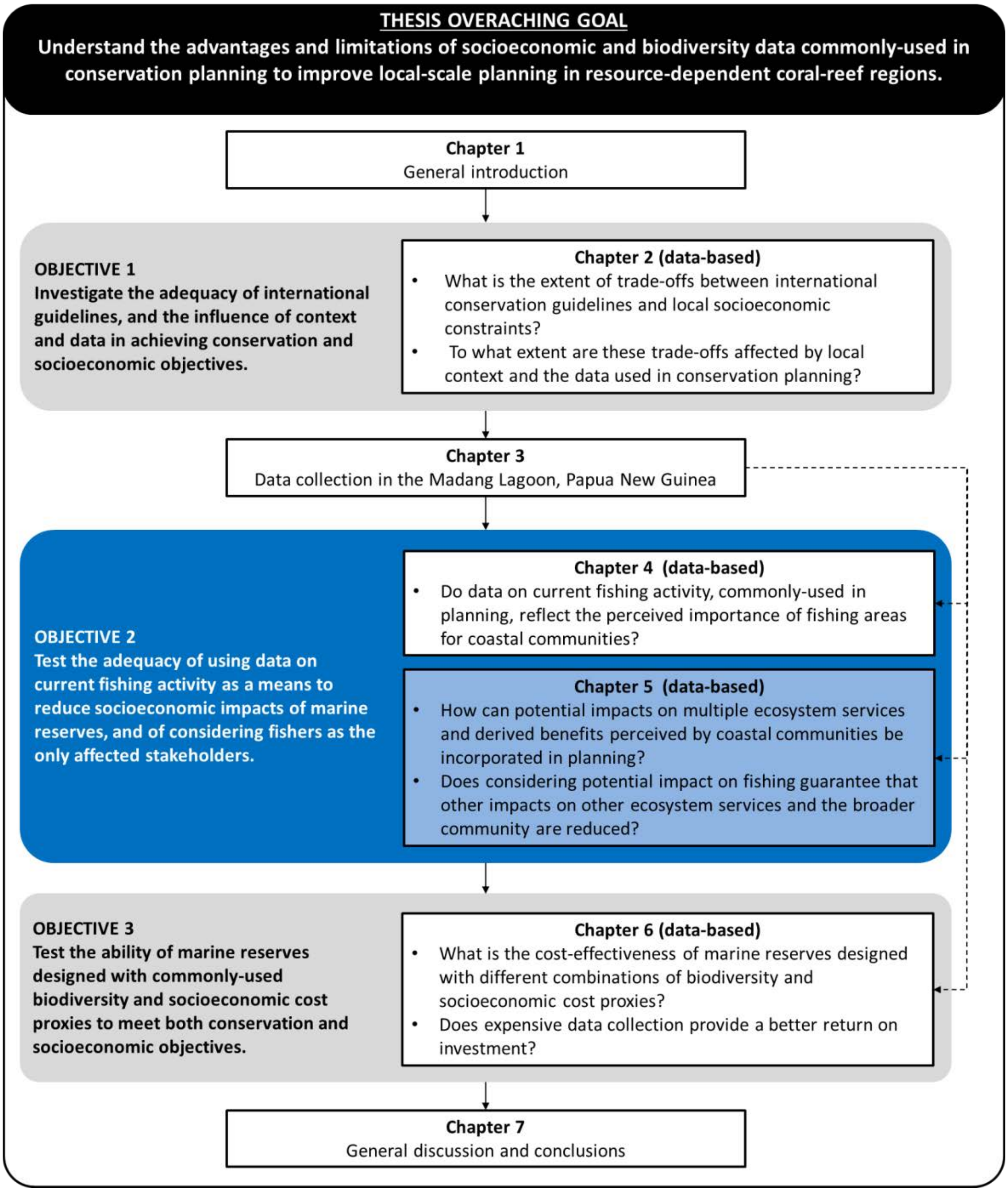
Defining and integrating socioeconomic costs of reserves in systematic conservation planning is challenging, especially when attempting to understand all possible costs in a given social-ecological system. This chapter shows that different yet valid spatial cost variables can be derived for a single system, but that planning with costs simplistically can provide a false sense of achievement, lead to conservation mistakes, and displace costs unnecessarily. These risks emphasize once more the importance of clearly defining conservation and socioeconomic objectives to inform the choice of cost data for particular groups of stakeholders.

Chapter 5.[1]

Accounting for the importance of ecosystem services and benefits to local communities in systematic conservation planning

In this chapter, I investigate the adequacy of using only data on fishing in marine conservation planning, as a proxy of how people value their environment. Putting aside important areas would limit access to or harvest in these places, and therefore incur a cost to the broader community, not just fishers. Here I develop the first approach for systematic conservation planning to incorporate information on the perceived importance of places for local communities to access a range of ecosystem services and benefits, using participatory mapping techniques.

[1] This chapter was slightly modified from the submitted version to avoid redundancies. The submitted version is currently under review: **M. A. Hamel**, R. L. Pressey, S. Andréfouët, L.S. Evans, C. Hicks (in revision) Accounting for the Importance of Ecosystem Services and Benefits to Local Communities in Systematic Conservation Planning: It's Not All About Fisheries. Plos One. Details on data collection are described in <u>Chapter 3</u>, and some of the supplementary material was directly included in the main text.


THESIS OVERACHING GOAL
Understand the advantages and limitations of socioeconomic and biodiversity data commonly-used in conservation planning to improve local-scale planning in resource-dependent coral-reef regions.
Chapter 1
General introduction
OBJECTIVE 1
Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.
Chapter 2 (data-based)
• What is the extent of trade-offs between international conservation guidelines and local socioeconomic constraints?
• To what extent are these trade-offs affected by local context and the data used in conservation planning?
Chapter 3
Data collection in the Madang Lagoon, Papua New Guinea
OBJECTIVE 2
Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders.
Chapter 4 (data-based)
• Do data on current fishing activity, commonly-used in planning, reflect the perceived importance of fishing areas for coastal communities?
Chapter 5 (data-based)
• How can potential impacts on multiple ecosystem services and derived benefits perceived by coastal communities be incorporated in planning?
• Does considering potential impact on fishing guarantee that other impacts on other ecosystem services and the broader community are reduced?
OBJECTIVE 3
Test the ability of marine reserves designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.
Chapter 6 (data-based)
• What is the cost-effectiveness of marine reserves designed with different combinations of biodiversity and socioeconomic cost proxies?
• Does expensive data collection provide a better return on investment?
Chapter 7
General discussion and conclusions

5. Accounting for the importance of ecosystem services and benefits to local communities in systematic conservation planning

Abstract

Incorporating meaningful social and economic information into conservation planning is challenging yet critical to minimising impacts of conservation actions on livelihoods, thereby maximizing compliance with restrictions on resource use. The social impacts of conservation reserves are often reduced only to socioeconomic considerations typically included in planning through opportunity costs. In many places where people are strongly connected to marine ecosystems, opportunity costs are often only measured for fishers. However, the services and benefits people gain from their marine environments go beyond just food and income from fishing. People access and value marine ecosystems for benefits that include recreation, aesthetic enjoyment, spiritual connections, medicine, and culture. Therefore, designing reserves with the aim to minimise only lost fishing opportunities might not minimise other lost opportunities associated with the marine environment. I explored how conservation planning can be informed and optimised with data on how people value coral-reef ecosystem services. I first developed a method to identify and map places of value (including fishing) to households, which involved engaging actively with the Riwo community of the Madang Lagoon, Papua New Guinea. I deployed this method through surveys with heads of households (n=52). Then, I used a novel way to incorporate the multiple benefits of the Madang Lagoon into spatial prioritisation. I found that different places in the Madang Lagoon are valued for different reasons, and that designing reserves based only on opportunity costs to fishing will likely have incidental impacts on the other ways people benefit from their marine environment such as spiritual and cultural uses. I also found that incorporating information on all benefits is the most effective way to minimise the loss of all benefits. I demonstrate how planners can move beyond accounting only for socioeconomic costs of conservation actions toward more comprehensive approaches which include a broader range of stakeholders, and ecosystem benefits.

Introduction

Successful conservation requires planners to understand and account for the needs and aspirations of people (Knight et al., 2008, Polasky, 2008). Conservation planners aim to protect areas with the highest benefit for biodiversity, and typically with the least impact on people's use of natural resources (Pressey and Bottrill, 2009). Despite claims of numerous long-term benefits of reserves for local communities and other stakeholders (e.g. Gell and Roberts, 2003, Russ et al., 2004b, Mascia et al., 2010, McClanahan, 2010), restrictions on human activities can also compromise access to ecosystem services, affect livelihoods and well-being (Adams et al., 2004, West et al., 2006, Adams and Hutton, 2007, Cinner et al., 2014), and ultimately compromise the core goal of reserves (Walmsley and White, 2003). It is now widely recognised by economists and social and conservation scientists that, to be effective, planning should consider whole social-ecological systems, but effective approaches to assess and incorporate relevant social and economic data into ecosystem-based planning are still developing (Ban et al., 2013, Kittinger et al., 2014, Le Cornu et al., 2014).

By minimising negative impacts of reserves on people, planners hope to improve social acceptance and compliance, thereby potentially improving the effectiveness of these reserves (Bergseth et al., 2013). Typically, planners consider the negative impacts of reserving places of significance for biodiversity in the form of "costs" of conservation actions to affected stakeholders (Naidoo et al., 2006, Ban and Klein, 2009). Potentially affected stakeholders include people who depend on the resources within reserves for commercial or non-commercial uses. The most straightforward way to assess these costs is by estimating opportunity costs or forgone access to resources (Ban et al., 2013). These costs are usually measured through proxies of the economic value of candidate areas for reservation, and are assumed to reflect broader socioeconomic impacts on people.

In marine systems, intended conservation actions can conflict with fishing activities important for economies and livelihoods, leading conservation planners to typically focus on the opportunity costs of conservation decisions to fishers (e.g. Ban and Klein, 2009, Weeks et al., 2010, Adams et al., 2011, Deas et al., 2014).

However, in most countries with extensive coral reefs, coastal communities gain more from their marine environment than just fish for food or income. People often access and value the marine realm for a range of other ecosystem services. In the following, “valuing” means “considering important or beneficial”. For example, Hicks and Cinner (2014) recently identified important types of services - two provisioning (fishery, materials), two regulating (coastal protection, sanitation), one supporting (habitat control), and four cultural (culture, education, recreation, and bequest) - explicitly valued by stakeholders across 28 coral-reef fishing communities in four countries.

Minimising opportunity costs to fishers by implementing conservation actions away from the most important fishing places is a first step towards incorporating people’s needs into conservation plans. However, as well as encouraging the creation of residual reserves[1] if applied at the wrong scales (Devillers et al., 2014), this approach can incidentally displace restrictions on access and resource extraction to places or activities that are important for other benefits. This could provide a false sense of achievement, hiding unexpected negative impacts on the wider community and compromising conservation effectiveness through lack of acceptance and compliance. Key research gaps here are therefore assessing and understanding how communities (not just fishers) perceive and value different places they access in their environment for services and benefits, including extractive uses, and then using this information explicitly in conservation planning. Such an approach would help minimise potential broader impacts of proposed conservation plans; identify opportunities to maximise both conservation and socioeconomic benefits; foster stakeholder engagement; and help make decision-making more transparent.

Following the publication of the Millennium Ecosystem Assessment (Millennium Ecosystem Assessment, 2003), literature is burgeoning on assessing economic, social, and cultural services and benefits of marine ecosystems to people (e.g. Egoh et al., 2007, Klain and Chan, 2012), with many studies recommending their

[1] The term “residual reserves” refers to the trend of terrestrial (and marine) protected areas in being “residual” to commercial uses. See DEVILLERS, R., PRESSEY, R. L., GRECH, A., KITTINGER, J. N., EDGAR, G. J., WARD, T. & WATSON, R. 2014. Reinventing residual reserves in the sea: are we favouring ease of establishment over need for protection? *Aquatic Conservation: Marine and Freshwater Ecosystems*. for more details.

incorporation into conservation planning. Despite calls from both the social sciences and the conservation-planning community to explicitly assess and incorporate such information into conservation plans (Fisher and Brown, 2014), only a few published studies have done so with tools for systematic conservation planning. These studies were terrestrial, undertaken over large extents (tens of thousands of square kilometres), and used biophysical proxies to value ecosystem services with no link to people's perceptions (see Egoh et al., 2010, Chan et al., 2011).

Approaches are available for collecting spatial information on ecosystem services and benefits to people across smaller extents and based on people's perceptions, but not in the marine literature. Brown (2004) reviewed and proposed methods to incorporate into natural-resource management what he calls "perceived landscape values", in other words the perceived services and benefits provided by landscapes. To do this, he adapted a typology of ten landscape values (life support, economic, scientific, recreation, aesthetic, wildlife, biotic diversity, natural history, spiritual, intrinsic) initially developed by Rolston and Coufal (1991). Brown (2004) then used participatory GIS techniques to map and measure the importance of places to people for their landscape values. Although he promoted the use of this method for conservation planning, the study did not provide information on how this might be done.

Raymond and Brown (2009) revised Brown's method to map what they called "community values" for natural capital and ecosystem services. Although the results were accounted for in the local conservation plan (by including "people" as an asset alongside land, water, biodiversity, and atmosphere), the spatial dataset created was not used explicitly to inform systematic conservation planning scenarios. Only recently, Whitehead et al. (2014) incorporated spatial data on "social value", collected with the approach of Raymond and Brown (2006), into systematic conservation planning. It is the only published study, to my knowledge, to do so. The study areas for all these approaches were terrestrial, in developed countries for which mail surveys were relevant, and often covering large areas. One challenge is now to further adapt this work to tropical marine contexts, local-scale conservation planning, and developing countries where communities rely strongly on natural resources for their livelihoods and have little alternative opportunities.

In this chapter, I develop a practical method for collecting, understanding, and incorporating spatial information on the perceived importance of multiple ecosystem services and their benefits into conservation planning scenarios. My goal is to demonstrate the potential broad impact on people of no-take areas where extraction of live and dead material is prohibited; and no-go areas where access is fully prohibited to everyone. I use the coral-reef ecosystems of the Madang Lagoon, Papua New Guinea, as a case study.

Materials and methods

Overview

The approach employed is summarised in **Figure 5.1**, outlining how to plan for the perceived value of places to people for the ecosystem services and benefits they provide, along with field methods, data processing, and spatial analyses. I present my method within the general framework for landscape- and seascape-scale conservation planning developed by Pressey and Bottrill (2009) and described earlier in Chapter 1. The widely-used framework consists of 11 steps ranging from scoping and costing the planning process through selecting reserves, to maintaining and monitoring them.

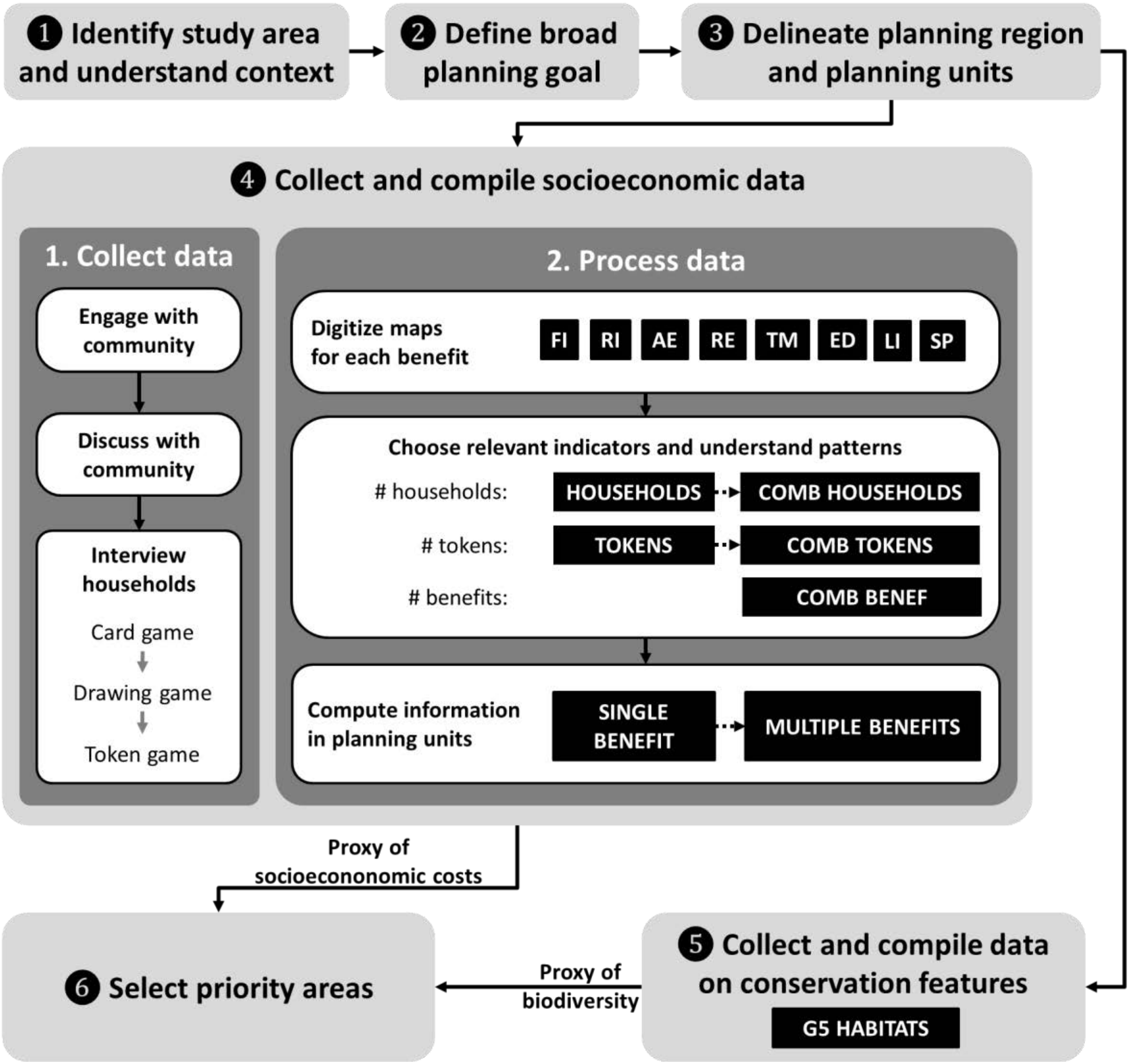


Figure 5.1. Overview of my method to incorporate information on ecosystem services and derived perceived benefits into systematic conservation planning. Each step is described in the main text. Benefits include fishing (FI), perceived biological richness (RI), aesthetics (AE), recreation (RE), traditional medicine (TM), education (ED), lime for betel nut chewing (LI), and spirituality (SP). Planners can decide to focus on individual benefits only (SINGLE BENEFIT), and/or on a combination of benefits (MULTIPLE BENEFITS). The multiple benefits cost layer requires the creation of single benefit cost layers.

Step 1: Identify the study area and understand the context

The first step of a conservation planning exercise involves identifying the study area, the stakeholders, and understanding its social and ecological context (stages 1 to 3 in Pressey and Bottrill, 2009). I chose to work in the Madang Lagoon, Papua New Guinea, located in the Coral Triangle region (**Figure 3.1**). The Madang Lagoon covers approximately 40 km^2. My choice of study area had two motivations. First, my socioeconomic work was combined with an extensive biological survey (more details in Chapter 3), not only facilitating logistics but also providing data on habitats and biodiversity for future planning (e.g. Fricke et al., 2014). Second, similar to most coral-reef countries (Bell et al., 2009), coastal communities of the Madang Lagoon rely on threatened coral-reef resources and habitats for their day-to-day life, subsistence, and income (Marnane et al., 2002, Kinch et al., 2005, Jenkins, 2011), providing a new and challenging situation for incorporating spatial information on the perceived importance of coral-reef ecosystem services and their benefits into conservation plans.

I focused on Riwo (*Ziwo*), the largest coastal community in the Madang Lagoon, because of its central location (**Figure 3.1**), large population (National Statistical Office of Papua New Guinea, 2002), and history of involvement in conservation and resource management. Three of the four Wildlife Management Areas (WMAs) in the lagoon were established by clans of Riwo, with the support of local and international NGOs (Jenkins, 2002a, Jenkins, 2002b). I engaged with the local coastal community of Riwo as stakeholders who could potentially be affected by conservation actions in their marine tenure.

Step 2: Define broad planning goals

Defining the broad planning goal(s) is an essential step in the planning process (stage 4 in Pressey and Bottrill, 2009), allowing planners to scope the type of data required. Broad goals are typically refined into quantitative conservation objectives later in the process. Here, my goal was to design reserves that protect marine habitat types within the Riwo waters, while minimising loss of access by the community to important ecosystem services and benefits. Therefore, spatial data on habitat types present in the region were necessary, as well as maps of the benefits provided by the Madang Lagoon as perceived by the community.

I planned for two types of reserves. The first, no-take areas, prohibited extraction of live and dead material. This restricted, for example, artisanal fisheries, collection of seaweed and shellfish for therapeutic purposes, and collection of coral or shellfish to make lime to chew with betel nut. The second type, no-go areas, prohibited access to everyone, regardless of the activity involved, including recreation, aesthetic enjoyment, and visits for spiritual purposes. No-go areas (corresponding to IUCN protected areas category Ia) ensure the natural integrity and values of places, and have been implemented elsewhere in Oceania (Great Barrier Reef Marine Park Authority, 2004, Bertaud, 2011, Deguignet et al., 2014). Although this type of extreme protection is not common, I used this category to demonstrate what would happen with restrictions on access for activities other than resource harvesting that could adversely affect biodiversity (e.g. recreation, tourism).

Step 3: Delineate planning region and planning units

Pressey and Bottrill (2009) defined their planning region in their stage 1 and refined boundaries with data in their stages 5 and 6. Here, the planning region was first broadly defined as the extent of marine habitats and islands within the section of the Madang Lagoon used by the Riwo community (roughly their customary tenure). I later refined the boundaries of my planning region by overlaying a grid of planning units (to define "places") on a map of marine habitats and islands in the study area (see step 5), and the spatial footprint of my benefits dataset (see step 4). The 319 300 m x 300 m planning units intersecting both marine habitats and benefits constituted the planning region (see Chapter 3 for more details on the choice of planning unit sizes).

Step 4: Collect and compile socioeconomic data

This step corresponds to stage 5 in Pressey and Bottrill (2009), and is the main focus of this chapter.

Step 4.1: Collect spatial data on perceived ecosystem services and benefits

Here the aim was to collect the necessary information to create accurate maps, across the planning region, of the degree of importance of social, economic, and cultural benefits derived from the Madang Lagoon's ecosystem services (hereafter "benefits"). Since many conservation actions restrict access to certain places for specific users or activities, I assume that losing access equates to losing the ability to derive benefits. To collect relevant data, I adapted the participatory GIS methods described in Raymond et al. (2009) to the context of the Madang Lagoon. The methods I developed to collect such data were fully described in Chapter 3 ("Household surveys").

Step 4.2: Process data to measure the importance of places for different benefits

All delineated regions from the drawing game were digitised manually in ArcGIS (ESRI 2010) in the form of polygons. One layer was created for each benefit, containing all polygons drawn by all households for the relevant benefit. To each polygon, I assigned the corresponding household, identified benefit, and standardised number of tokens (from 0 to 100) allocated in the token game. An "area" was defined by the presence of only one polygon, or the overlap between two or more polygons. For example, two partially overlapping polygons produced three areas: the non-overlapping parts of the first and second polygon, and the intersection of the two (**Figure 5.2**). I then needed to create indicators, at the community level, of the relative importance of accessing areas for the ecosystem services and benefits they provide, reflecting spatial patterns as accurately as possible.

Table 5.1. Cost variables, based on indicators of the importance of each area/planning unit for a single benefit or combination of benefits. Cost x was calculated for all areas/planning units in the planning region from variables associated with delineated polygons (see **Table 5.2** for a description of variables, and **Figure 5.2** for a schematic representation). All indicators were normalised as percentage of maximum to standardise units and allow direct comparisons ($0 \leq x \leq 100$).

Calculated for	Type	Indicator	Formula	Description
An area (raw data) **Figure 5.2.A**	Single benefit	HOUSEHOLDS	$x_1 = n$	Number of households who visited the area for the benefit of interest.
		TOKENS	$x_2 = \sum_{n=0}^{N} i_{HH_n} . w_{HH_n}$ with $w_{hh_n} = \frac{a_{HH_n}}{A_{HH_n}}$	Importance of an area for the benefit of interest. Sum of the number of tokens associated with polygons for each household $\boldsymbol{n}$ within the area $\boldsymbol{i_{hh_n}}$, weighted by the proportion of the area of each polygon intersecting the area $\boldsymbol{w_{hh_n}}$.
	Multiple benefits	COMB BENEF	$x_3 = b$	Importance of an area for a combination of benefits of interest. Number of unique perceived benefits assigned to the area.
		COMB HOUSEHOLDS	$x_4 = \sum_{b=0}^{B} n_b$	Importance of an area for a combination of benefits of interest. Number of unique households who visited the area, all benefits combined.
		COMB TOKENS	$x_5 = \sum_{b=0}^{B} j_b . x_2$	Importance of an area for a combination of benefits of interest. Sum of all calculated $\boldsymbol{x_2}$ (TOKENS) for all benefits of interest, weighted by the relative importance $\boldsymbol{j_b}$ of each benefit.
A place (planning unit) **Figure 5.2.B**	Single benefit	TOKENS	$x_6 = \sum_{n=0}^{N} i_{HH_n} . w_{HH_n}$ with $w_{hh_n} = \frac{a_{HH_n}}{A_{HH_n}}$	Importance of a place for the benefit of interest. Sum of the number of tokens associated with polygons for each household $\boldsymbol{n}$ within the place $\boldsymbol{i_{hh_n}}$, weighted by the proportion of the area of each polygon intersecting the place $\boldsymbol{w_{hh_n}}$.
	Multiple benefits	COMB TOKENS	$x_7 = \sum_{b=0}^{B} j_b . x_6$	Importance of a place for a combination of benefits of interest. Sum of all calculated $\boldsymbol{x_6}$ (TOKENS) for all benefits of interest, weighted by the relative importance $\boldsymbol{j_b}$ of each benefit.
	Reference	UNIFORM	$x_8 = 100$	Uniform importance (each planning unit, including trimmed ones had the same cost).

Table 5.2. List of variables used to calculate the importance of areas (raw data), and the costs of reserving each planning unit.

Variable name	Description
n	number of households "using" the place ($0 \leq \boldsymbol{n} \leq \boldsymbol{N}$)
N	total number of households surveyed
i_{HH_n}	importance (number of tokens) of the polygon used by household $\boldsymbol{n}$ for the benefit of interest
a_{HH_n}	area (m^2) of polygon used by household $\boldsymbol{n}$ within the place
A_{HH_n}	total area (m^2) of polygon used by household $\boldsymbol{n}$
w_{HH_n}	proportion of the polygon used by household $\boldsymbol{n}$ within the place
b	number of perceived benefits provided by the place ($0 \leq \boldsymbol{b} \leq \boldsymbol{B}$)
B	total number of perceived benefits in the planning region
n_b	number of households "using" the place for the benefit of interest $\boldsymbol{b}$ ($0 \leq \boldsymbol{n_b} \leq \boldsymbol{N}$)
j_b	importance of benefit $\boldsymbol{b}$ for the community

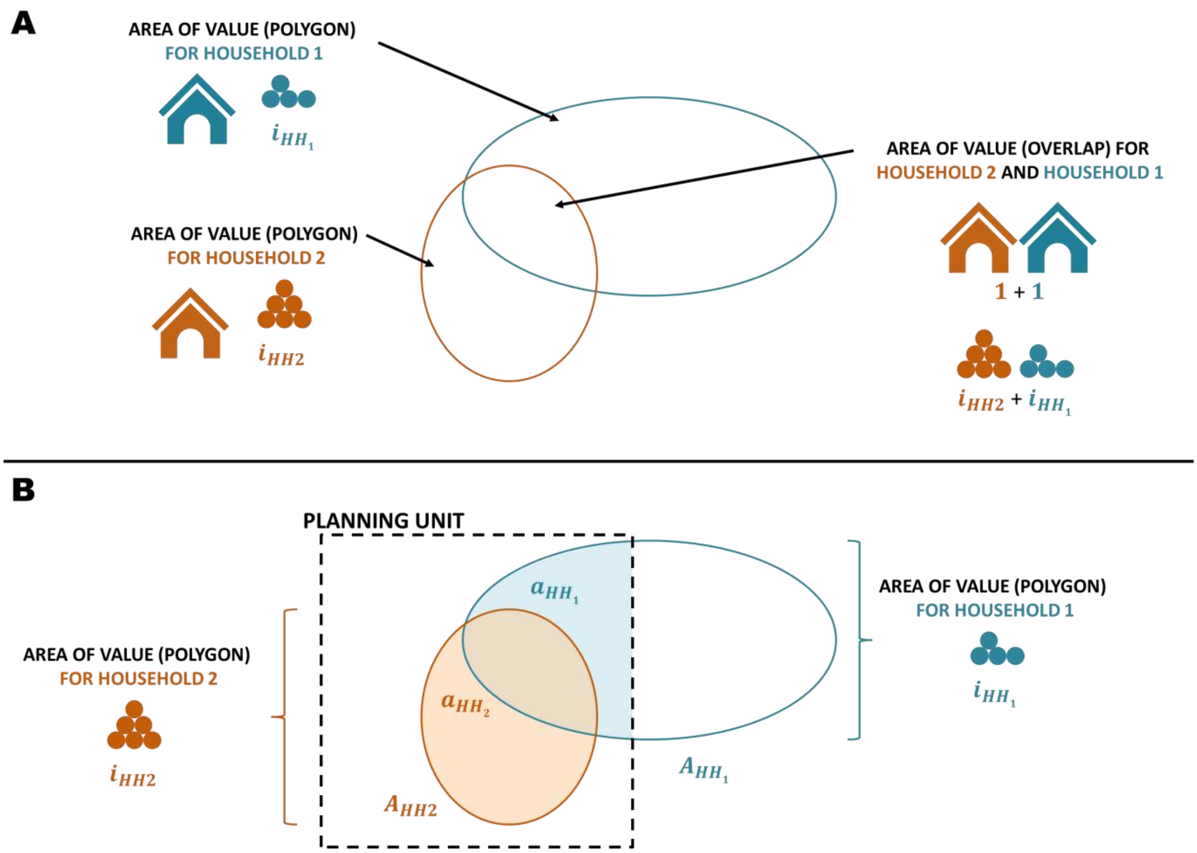


Figure 5.2. Schematic representation of the variables used to derive spatial datasets for each benefit of interest (e.g. fishing, recreation, aesthetics), showing the importance of areas (A) and places (planning units) (B). In both panels, the same two polygons are presented, each delineated by a different head of household representing his/her household (1, in blue, and 2, in red). Each polygon is associated with its importance in regard to the benefit of interest (respectively i_{HH1} and i_{HH2} based on the number of tokens), and its total area (respectively A_{HH1} and A_{HH2}). In (A), the importance of each area is calculated as the number of households visiting the area (# HOUSEHOLDS), or as the total number of tokens (# TOKENS) associated with all overlapping polygons (sum of all i_{HH_n},), depending on the method chosen. In (B), the importance of a place (or planning unit) is calculated as described in **Table 5.1** and **Table 5.2**. One place is represented, covering partially area a_{HH1} of the polygon delineated by household 1 and entirely the polygon delineated by household 2 ($a_{HH2} = A_{HH2}$).

I created two indicators to measure the importance of areas for a given benefit (details in **Table 5.1**, **Table 5.2**, and **Figure 5.2**): 1) the number of households identifying an area as important (HOUSEHOLDS); and 2) the total number of tokens assigned to the area (TOKENS). For a given benefit, HOUSEHOLDS was calculated as the number of overlapping polygons constituting each area in the relevant benefit layer (minimum 0, theoretical maximum 52). TOKENS was the sum of the standardised number of tokens assigned to polygons within the area, weighted by the proportion of the surface of each polygon intersecting the area (minimum 0, theoretical maximum 52*100). To compare the importance of areas as shown by both indicators, I standardised, for a given benefit, each indicator from 0 to 100, 100 being the maximum value of an indicator in each benefit layer. Note that this standardisation is different to the one used to normalise the tokens sets allocated to households during the household surveys. I investigated differences in spatial patterns between benefits visually. I created difference maps to reveal differences between indicators by converting final polygon layers to raster (**Appendix 7: Figure 1 and methods**).

I created three indicators to measure the importance of areas for all benefits combined (details in **Table 5.1**, **Table 5.2**, and **Figure 5.2**): 1) the number of benefits assigned to each area (COMB BENEF); 2) the number of households valuing each area regardless of the number of benefits identified (COMB HOUSEHOLDS); 3) the total number of tokens assigned to the area, all benefits and households combined (COMB TOKENS). I calculated the three indicators for the following combinations of benefits: all benefits and benefits corresponding to extractive and non-extractive uses (see **Table 3.1** for use categories). To calculate COMB BENEFITS and COMB HOUSEHOLDS, I first merged all polygons from all households for the relevant combination of benefits to a single layer using my raw data. COMB BENEFITS and COMB HOUSEHOLDS were calculated by dissolving the merged layer by benefits or households respectively, then counting the number of overlapping polygons. These two measures involved weighting individual benefits equally. To calculate COMB TOKENS, I weighted, in each benefit layer created for TOKENS, the sum of TOKENS for each area by the proportion of people in my sample to whom the benefit was perceived as relevant, as assessed in the card game. For example, when 98% of households indicated fishing as a benefit, all

TOKENS calculated for all areas in the fishing cost layer were multiplied by 0.98. I then summed all eight "weighted" benefit layers together. I compared the importance of areas as per the three indicators, and considering the different combinations of benefits.

Once indicators calculated for areas were compared, and fine-resolution spatial patterns identified, the next step involved choosing the most relevant indicator and computing information on costs for each place (planning unit) within the planning region. Regardless of the positive effects that reserves might have on ecosystem services and benefits, I assumed that reservation meant losing rights to access or harvest, thereby incurring a "cost". I defined this cost of restricting access or harvest in a given place as how highly valued this place was for a particular benefit. In other words, the more a place was valued for its benefits, the higher the cost of putting this place aside. I chose to use TOKENS as a cost indicator because it gave the most detail on how people valued access to areas. First, I created a separate cost layer for each of the eight benefits by computing a measure of TOKENS for each place (**Figure 5.3**) using similar equations to those used for areas (**Table 5.1**). Second, I built a cost layer combining all benefits. For the same reasons as above, I chose to use COMB TOKENS, and computed a measure of this indicator for each place (**Figure 5.3**). Finally, for comparison, I also derived a uniform cost where all places, including trimmed ones, had a cost of 100. All spatial cost layers were created using a combination of ArcGIS (ESRI, 2010) and R (R Development Core Team, 2008).

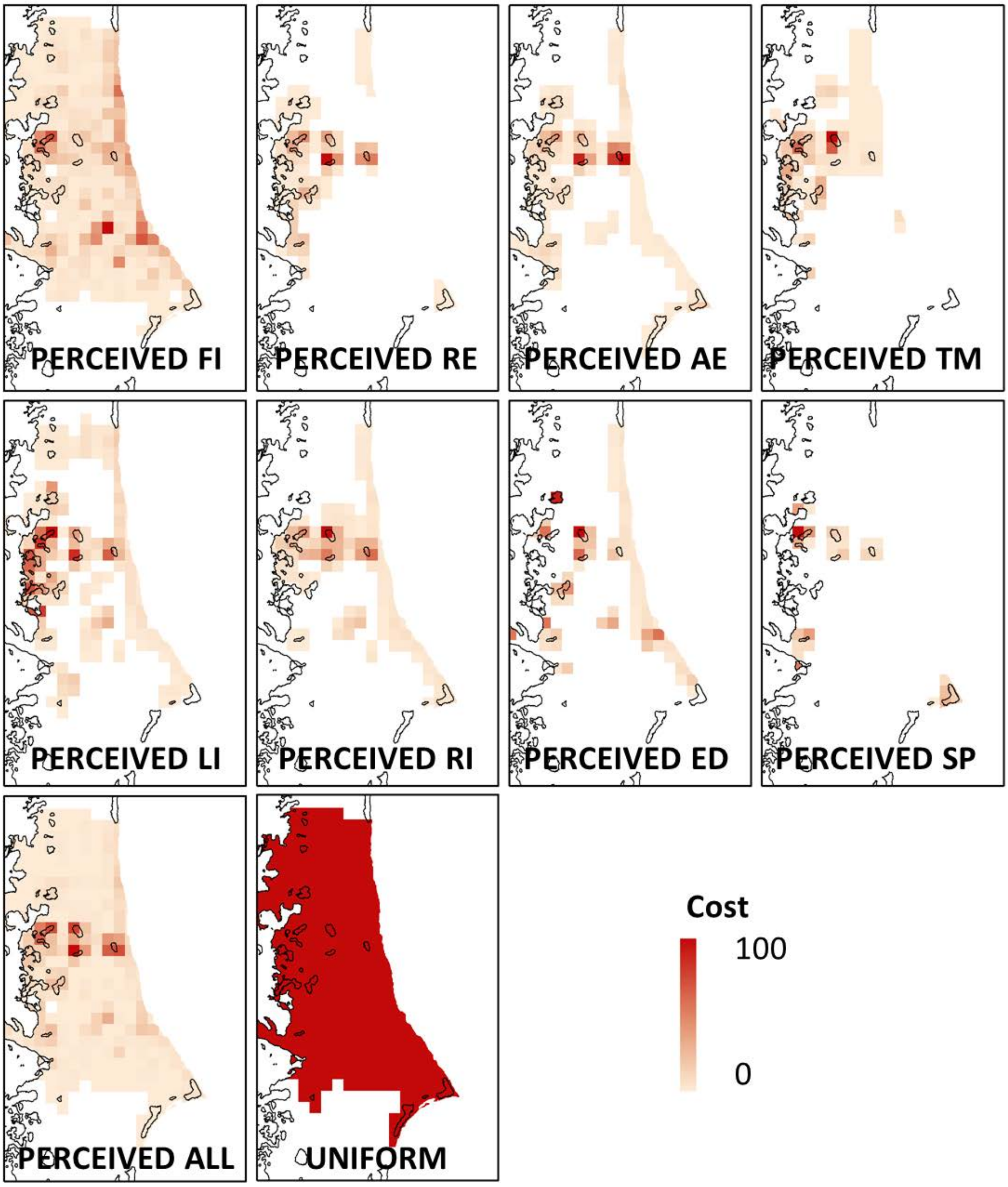


Figure 5.3. Standardised cost layers used in spatial prioritisation. For each benefit, a cost layer was derived using the TOKENS indicator, based on the importance of each drawn polygon to access the benefit of interest. Benefits are: fishing (FI); recreation (RE); aesthetics (AE); traditional medicine (TM); collecting material to make lime for betel nut chewing (LI); perceived biological richness (RI); education and knowledge sharing (ED); and spiritual value (SP). The cost layer combining all benefits (ALL) was derived using the COMB TOKENS indicator: the sum of TOKENS calculated for each benefit, weighted by the importance of each benefit for the community (see main text for calculations). A uniform cost layer (UNIFORM) is also shown.

Step 5: Collect and compile data on conservation features

This step corresponds to stage 6 in Pressey and Bottrill (2009). I created a habitat map of the Madang Lagoon from a high-resolution (2 m) Worldview satellite image (see "user approach" in Andréfouët 2008 for details on method). The resulting hierarchical habitat typology for the Madang Lagoon describes 28 geomorphic types (**Appendix 1**), including 22 within the planning region. I computed the extent of each habitat type in each of the 319 places.

Step 6: Select priority areas

The compiled data on habitats and socioeconomic costs was used in systematic conservation planning scenarios through Marxan planning software (Ball et al., 2009). The scenarios proposed reserve systems that limited activities posing a threat to coral-reef ecosystems in the form of no-go or no-take areas, achieving representation objectives for habitats at minimum cost. My scenarios addressed two questions: 1) What is the incidental cost incurred by the reservation of places valued for the full range of services and benefits when only cost to fishing is minimised? and; 2) What are the potential advantages and limitations of incorporating all benefits in a single prioritisation exercise, by minimising the cost related to all benefits combined?

Multiple scenarios were run to answer each question (**Table 5.3**). I refined conservation goals defined earlier (step 2) by setting a quantitative objective for habitat protection: a proposed reserve system must include at least 20% of the total extent of each habitat in the study area, following a precautionary interpretation of Aichi target #11 (UNEP/CBD/COP/10/X/2, 2010) as in Chapter 2. In each scenario, the cost of reserving places was minimised, using the different cost layers from step 4 (**Table 5.1** and **Figure 5.3**). Marxan was run 1000 times for each scenario.

Table 5.3. Scenarios used for spatial prioritisation.

Question	Scenario names	Conservation objective	Socioeconomic constraint or objective	Costs minimised	Costs measured
1. What are the incidental costs of reducing impact on fishing only?	- FISHING	Strict representation of 20% of the extent of each habitat type	Constraint: minimise loss of access to places valued for fishing	PERCEIVED FI (indicator: TOKENS)	- Costs to benefits related to extractive uses in the context of no-take reserves: PERCEIVED FI, TM (E), and LI. - Costs to all benefits in the context of no-go reserves: PERCEIVED FI, RE, AE, TM, TM (E), TM (N-E), LI, RI, ED, SP.
	For comparison: - RECREATION - AESTHETICS - TRADITIONAL MEDICINE - TRADITIONAL MEDICINE (E) - TRADITIONAL MEDICINE (N-E) - LIME - BIOLOGICAL RICHNESS - EDUCATION - SPIRITUAL	As above	Constraint: minimise loss of access to places valued for benefits other than fishing: recreation, aesthetic enjoyment, all uses, extractive or non-extractive uses of traditional medicine, lime, perceived biological richness, education, spirituality benefits, respectively	PERCEIVED FI, RE, AE, TM, TM (E), TM (N-E), LI, RI, ED, SP (indicator: TOKENS)	
	For comparison: - UNIFORM (reference scenario)	As above	Constraint: minimise the total extent of reserves	UNIFORM	
2. What are the advantages and limitations of combined benefits?	- ECOCENTRIC (conservation objective and socioeconomic constraint)	As above	Constraint: minimise loss of access to places valued for all benefits.	PERCEIVED ALL (indicator: COMB TOKENS)	Costs to all benefits, in the context of no-go reserves: PERCEIVED FI, RE, AE, TM, TM (E), TM (N-E), LI, RI, ED, SP.
	- SOCIAL-ECOLOGICAL (conservation objective, socioeconomic objective and socioeconomic constraint)	As above	Objective: no reservation of areas of high value for combined benefits. Constraint: minimise loss of access to places valued for all benefits	PERCEIVED ALL (indicator: COMB TOKENS)	

For question 1 (**Table 5.3**: "what are the incidental costs of reducing impact on fishing only?"), the FISHING scenario minimised costs only in terms of fishing. I looked at how implementing the candidate reserve systems as either all no-go or all no-take would likely affect access to places providing benefits (including fishing) to the Riwo community. For comparison, I also ran scenarios (RECREATION, AESTHETICS, TRADITIONAL MEDICINE, TRADITIONAL MEDICINE (E), TRADITIONAL MEDICINE (N-E), LIME, BIOLOGICAL RICHNESS, EDUCATION, SPIRITUAL) that aimed to minimise loss of each non-fishing benefit separately. I also ran a scenario (UNIFORM) with a uniform cost of 100 for all places. For each scenario, I measured two things: 1) the proportion of places, valued for each benefit other than fishing, whose access/harvest would be lost with implementation of candidate reserves; 2) the percentage incidental cost, for each benefit including fishing, of the best candidate reserve systems in relation to the maximum possible cost for each benefit (i.e. if all 319 places were reserved).

For Question 2 (**Table 5.3**: "what are the advantages and limitations of combined benefits?"), scenarios were set to design systems of no-go areas that also achieve the conservation objectives above, but this time with minimal impact on all benefits combined. To do this, I tested two different scenarios, both using the same combined cost layer created in step 4. In the "ecocentric" scenario (ECOCENTRIC), I designed reserves using the conservation objective while minimising costs for all benefits combined, including fishing. In the "social-ecological" scenario (SOCIAL-ECOLOGICAL), I excluded *a priori* the most valued places from the pool of possible reserves (i.e. all places with a cost for combined benefits above the upper quartile), as a way to consider the reduction of socioeconomic impact as an objective rather than as a constraint only. Then I designed reserves, excluding these places, with the same objectives as in the ECOCENTRIC scenario. I measured all incidental costs for all benefits, and the number of planning units reserved in both scenarios.

Results

The card game: relevant ecosystem services and benefits

All benefits listed in the cards were mentioned by at least three of the four focus groups, except education, only mentioned by non-fishers. Households visited and valued their marine environment (marine habitats and islands) for fishing (98% of all surveyed households), followed by recreation (79%), aesthetics (65%), traditional medicine (60%), lime material for betel nut chewing (56%), perceived biological richness (50%), education (33%), and spirituality (21%). Tourism and two benefits not in the cards (transportation and friendship) were identified by only three or fewer households so were excluded from analyses.

The drawing and token games: areas providing ecosystem services and benefits

Many of the areas people valued for fishing, recreation, aesthetics, and biological richness overlapped between households, indicating some consensus on important areas (**Appendix 7: Figures 2 and 4**). In contrast, areas valued for therapeutic, lime production for betel nut chewing, education, and spiritual benefits differed more strongly between households.

Overlaps of areas delineated for each benefit showed ten distinct geomorphologic entities of importance for the Riwo community: six islands (*Guzem*, *Duad*, *Duad Tinan*, *Tabad*, *Sinub* and *Wongad)*; three patch reefs (*Yazi Tinan*, *Yazi Natun* and *Mitzegwadan)*; and the barrier reef (**Appendix 7: Figure 3**). In general, reefs were mostly valued for fishing, while islands were appreciated for a greater range of uses and benefits, including fishing. In particular, *Tabad* and *Wongad* islands were identified as having great recreation and aesthetic benefit. *Tabad*, *Sinub* and *Wongad* islands had the greatest number of benefits (see **Figure 3.1** for a map of these locations). The barrier reef had the highest fishing benefit.

I found small differences between the importance of areas according to the number of households (HOUSEHOLDS) and the number of tokens (TOKENS). Compared to TOKENS, HOUSEHOLDS tended to emphasise the importance of very specific areas relatively far from the coast for fishing, aesthetics, and lime, but reduced the apparent importance of some places closer to shore for recreation (**Appendix 7: Figure 1 and methods**). HOUSEHOLDS tended to slightly emphasise the therapeutic and education benefits of all areas compared to TOKENS. Both indicators were very similar for biological richness and spiritual benefit.

Combining all benefits showed the overall importance of areas of the lagoon for the Riwo community according to different indicators of importance. The three measures (COMB BENEF, COMB HOUSEHOLDS and COMB TOKENS) showed similar spatial patterns with some differences (**Appendix 7: Figure 5**). In general, more benefits (COMB BENEFITS) were assigned to islands, mainly because of the wide range of uses they support. More people (COMB HOUSEHOLDS) valued *Wongad* and *Tabad* Islands, especially for non-extractive uses. The reefs around *Wongad* Island provided all eight types of benefits. Areas valued for extractive uses (fishing, collecting material for lime, therapeutic uses) were mainly the barrier reef, *Yazi Tinan*, and *Yazi Natun.* Accounting for the way people weight each area for a given benefit, and the benefits themselves (COMB TOKENS) altered the importance of some areas when compared to COMB HOUSEHOLDS. For example, many households valued *Wongad* and *Tabad* for non-extractive uses but placed a relatively low importance on these islands in terms of the benefits they provided. Fewer people valued *Duad* and *Duad Tinan* overall but these islands and surrounding reefs were very important to these people in terms of the benefits they provided.

Incidental costs of reducing impact on fishing only

Designing reserves that aimed to minimise only costs to fishing limited access or harvest in places of high social, cultural, and economic benefit, generating substantial incidental costs. With the FISHING scenario, a high proportion of places valued for extractive uses, including fishing itself, incidentally tended to be candidates for reservation (**Figure 5.4**). In the best solution that achieved conservation objectives at the smallest possible cost and for the smallest reserve system out of 1000 runs, candidate reserves included 19% of places valued for fishing, but also incidentally included 25% of places for extractive aspects of traditional medicine and 19% for lime material. The cost of this scenario to fishing was relatively low at 3% of maximum possible cost. Major incidental costs were for spirituality (19% of maximum possible cost), extractive uses for traditional medicine (18%) and lime material (10%). Overall, higher costs of the FISHING scenario were incurred for restrictions on access rather than only on harvest. Although the implementation of proposed reserves as no-take had a strong impact on extractive benefits other than fishing, implementing these reserves as no-go not only incurred costs to all extractive uses, but also particularly affected benefits related to Riwo people's culture and traditions (e.g. spirituality, traditional medicine, lime for betel nut chewing). In contrast, there were minor incidental costs for non-extractive traditional medicine (1% of valued places in candidate reserves). Interestingly, only 10% of places valued for their biological richness were candidate reserves in this scenario.

Designing reserves to minimise costs to each benefit separately highlighted how accounting for one benefit can incur significant incidental costs to others (**Figure 5.4**). Notably, the best solution for the FISHING scenario incurred incidental costs to all benefits (average of all incidental costs of 7%) but these costs were lower than best solutions for scenarios focused on other individual benefits (between 9 and 34%). Not including spatially-explicit costs in the reserve design (UNIFORM scenario) incurred the second highest average incidental cost (best solution: 25%).

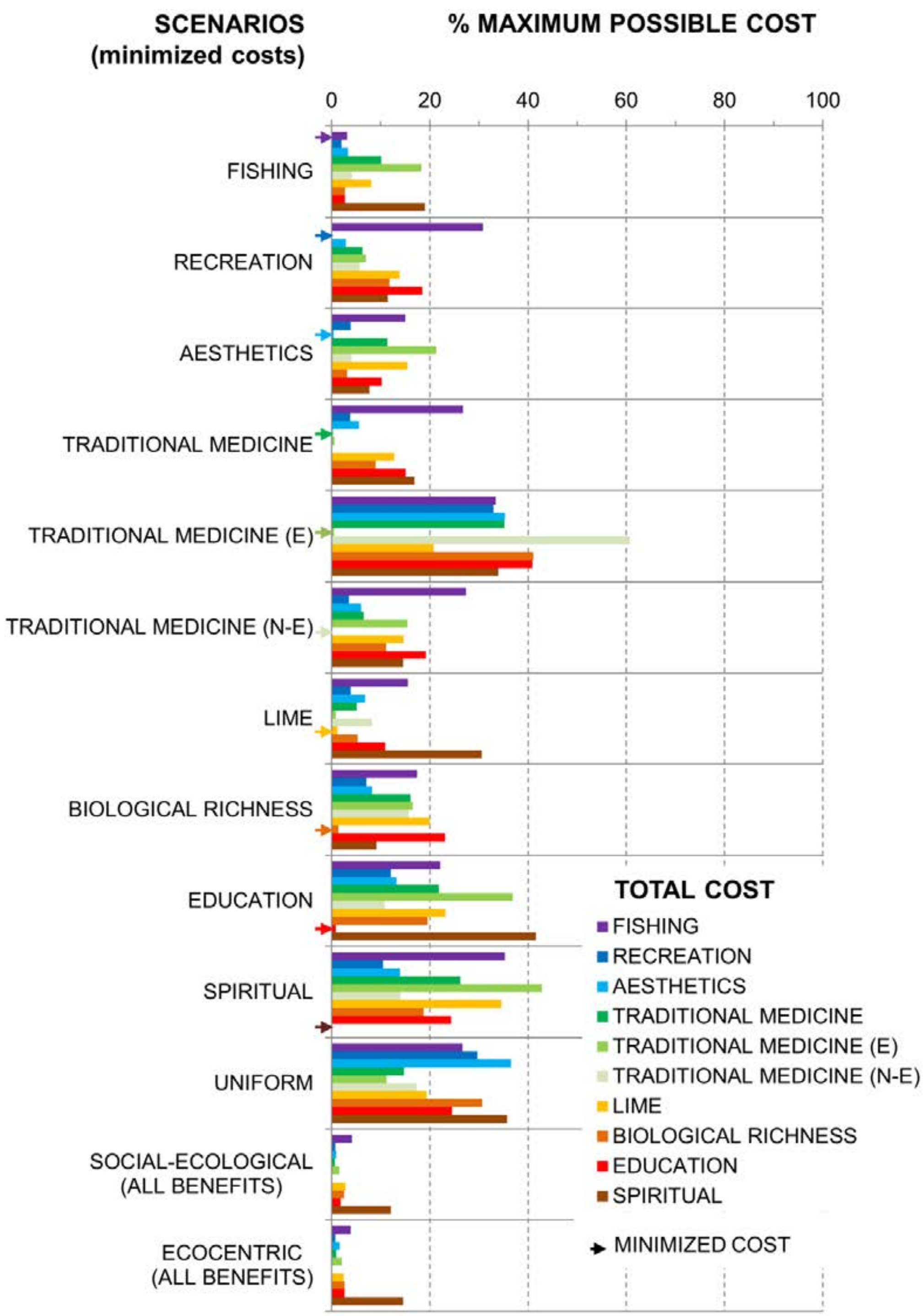


Figure 5.4. Incidental costs from all scenarios. Scenarios are listed on the y-axis. Each incidental cost for the best solution of each scenario, on the x-axis, was calculated as the percentage of the maximum possible cost for the corresponding benefit (i.e. the total cost when all places were selected for reserves). Scenarios for which costs to individual benefits were minimised are identified by the name of the cost that was minimised, with minimised costs indicated with arrows. (E) and (N-E) refer to “extractive” and “non-extractive” aspects of traditional medicine, respectively. For the UNIFORM scenario, a uniform cost layer was used, where all places in the planning region had a cost of 100. The ECOCENTRIC and SOCIAL-ECOLOGICAL scenarios both aimed to minimise a cost representing the combination of all benefits. All scenarios had the objective of representing 20% of the extent of each habitat type in the planning region.

Advantages and limitations of combined benefits

The two combined benefits scenarios (ECOCENTRIC and SOCIAL-ECOLOGICAL) yielded very similar results, regardless of the way I incorporated socioeconomic information (**Figure 5.5**). The best reserve system proposed for the ECOCENTRIC scenario required 67 places to be protected, while the SOCIAL-ECOLOGICAL scenario required 72 places (aside from those locked out as candidates because of their social benefits). The two resulting reserve systems were very similar spatially, sharing 51 candidate reserves. The ECOCENTRIC scenario only selected two places that were locked out in the SOCIAL-ECOLOGICAL scenario. Both the best ECOCENTRIC and SOCIAL-ECOLOGICAL reserve systems were successful at lowering incidental costs for all benefits with an average incidental cost of only 3% each, with individual costs ranging from 0 to 4%, except for spiritual benefit with a cost of 15% and 12% respectively (**Figure 5.4**). Only two of the eight benefits had lower incidental costs in scenarios focused on other individual benefits: traditional medicine (extractive uses only) had the lowest incidental cost with the LIME scenario (1% against 2% for the ECOCENTRIC and SOCIAL-ECOLOGICAL scenarios, respectively); spirituality had the lowest incidental cost with the AESTHETICS scenario (8% against 12% and 15% for the SOCIAL-ECOLOGICAL and ECOCENTRIC scenarios, respectively). The combined benefits scenarios also required fewer planning units than other scenarios to achieve conservation objectives.

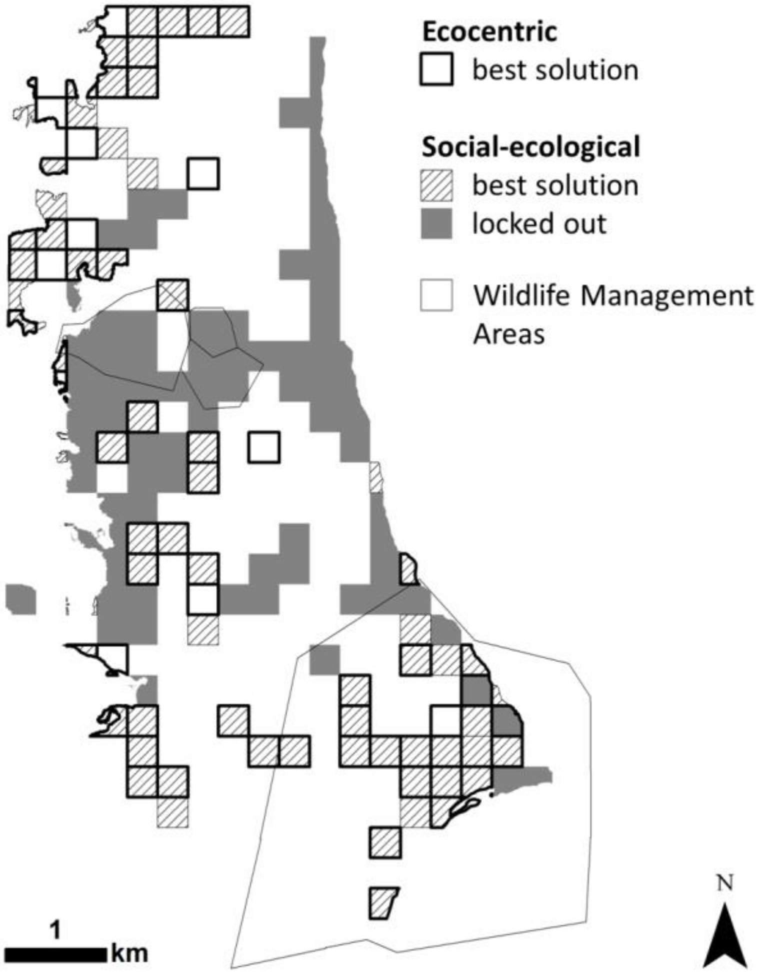


Figure 5.5. Best reserve systems for the ECOCENTRIC and the SOCIAL-ECOLOGICAL approaches. The ECOCENTRIC approach aimed to design a system that represented 20% of the total extent of each habitat type in the planning region, while minimising social, economic, and cultural costs to the Riwo community. The SOCIAL-ECOLOGICAL scenario had the same aim, but selecting only from places with relatively low benefits for the Riwo community by *a priori* locking out places with higher benefits (i.e. places for which the aggregated benefit was greater than the third quartile; grey places). Actual Wildlife Management Areas are shown with thin black outlines.

Discussion

Despite calls from both the social and conservation sciences to incorporate social, economic, and cultural benefits into conservation planning, few studies provide practical ways to do so. This is particularly important in regions such as Oceania where people are strongly connected to highly-threatened marine ecosystems such as coral reefs for a range of ecosystem services and benefits. In this chapter, I developed a method to assess and understand spatially how people value different places in a reef system for diverse benefits derived from ecosystem services, and proposed practical ways to incorporate this information into conservation planning. I also developed several planning scenarios to show how the consideration of places important for the various benefits they provide can reduce the potential social impact of proposed reserves.

I found that specific places were valued for a wide range of benefits by the coastal community of Riwo. Considering impacts of conservation actions only on fishing can displace reserves to incidentally affect access to other important benefits, thus potentially negatively affecting people. I also found that considering the whole range of benefits combined is the most efficient way to design reserve systems that reduce broad impacts, because considering benefits separately can incur unnecessarily large costs to other benefits. Below, I highlight several key insights gained from this exercise for systematic conservation planning.

Certain places provide specific ecosystem services and benefits enjoyed and valued by people

In the Madang Lagoon, specific places were valued to different degrees by the Riwo community for fishing, but places were also regularly visited and valued for other reasons. If access or uses are restricted in these areas by conservation actions such as no-take or no-go reserves, many benefits to communities would be diminished. I assumed that, when local coastal communities lose access and harvest rights in places, costs are borne, proportionally to the importance of these places in terms of the benefits they provide.

Ecosystem services and derived benefits are dynamic in space and time, and the way people perceive them is complex. More research would be required to understand the subtleties of each benefit and how different conservation actions might affect them (positively or negatively). For instance, new benefits could appear or be positively affected in specific places through reserve implementation (e.g. new reserves can be valued for their biodiversity, the aesthetics of restored habitats, or their spirituality), or displaced (e.g. places around new reserves could have increased aesthetic or fishing benefits, or people might find new places for recreation) (Graham et al., 2011b). No-take areas, no-go areas, and other types of conservation actions (e.g. resource harvesting quotas or size restrictions, restrictions on certain activities only, education and communication) could also change the dynamics of benefits in various ways. In my context, I found higher costs overall for restrictions on access (assuming benefits related to extractive and non-extractive uses are affected by no-go areas) rather than only on harvest (assuming only benefits related to extractive uses are affected by no-take areas).

Implications for conservation planning

Conservation planners often focus on the economic benefits of places, and concentrate efforts on reducing opportunity costs to economically-engaged stakeholders in the hope of facilitating implementation and compliance (e.g. Naidoo et al., 2006, Ban and Klein, 2009, Adams et al., 2010, Schröter et al., 2014). These stakeholders are not necessarily representative of whole communities, so overlooking wider community preferences can lead to inequity and frustration. I showed that reducing impacts of conservation actions to people is not all about fishing, since avoiding closures in areas valued for fishing can incidentally affect access to other benefits important to communities.

We can promote the support of communities for conservation by ensuring preferences and perceptions are accounted for, with community participation and by engaging communities in planning. My approach can help planners achieve this. First, I measured benefits and incorporated them into planning scenarios in a transparent way. I showed that explicitly considering a combination of benefits is the most effective way to reduce costs for each individual benefit, compared to scenarios that consider cost related only to single benefits such as fishing. Second, I tested communication and engagement methods to make my approach as well received as possible by the community. I split my relatively long surveys into "games", and used participatory GIS which helped engage respondents and whole households in the activities, especially important in regions such as Madang with apparent research fatigue. The time spent with the community (one month in total) was short but longer than the average duration of previous surveyors in the region as reported by community members, which considerably helped to build trust and gain insight into the local social-ecological context. Should this study be replicated as part of an actual planning project (as opposed to a research exercise), I recommend that planners ensure they engage with local people during the whole conservation planning process, not just during one socioeconomic survey. Importantly, planners should return to the communities to seek feedback on data and spatial analyses, and to discuss opportunities to incorporate this feedback.

Accounting for a range of local social, cultural, and economic benefits in systematic conservation planning is difficult but possible

This is a first step towards integrating ecosystem services into local-scale systematic conservation planning, which means it has obvious caveats, outlined below, that should be considered in subsequent studies and planning projects.

My work was largely exploratory and used various indicators of the importance of places, mostly showing similar spatial patterns but with slight variations. There is no ideal indicator, and how to choose the most relevant one is at the discretion of planners and will depend on objectives and constraints. Applying COMB BENEF gives an indication of the range of uses an area supports but is simplistic because it does not account for the different degrees to which different people value different benefits. Using HOUSEHOLDS (for one specific benefit) or COMB HOUSEHOLDS (for a combination of benefits) effectively indicated areas of importance and the degree of their importance at the community level but did not account for the possible variation in perceptions among households. In practice, TOKENS (for one specific benefit) or COMB TOKENS (for a combination of benefits) might give additional information on the degree of importance of each area for individual households but biases are likely, depending of sampling method used. Furthermore, challenges arise in determining the best way to standardise the number of tokens, and requirements for data collection significantly increase the duration of surveys. Using a combination of indicators would likely give a fuller picture.

Another caveat concerns the weighting of each benefit when aggregating benefits. I used the proportion of people to whom the benefit was relevant because my rank data did not allow me to assess the importance of each benefit in relation to others. However, other weighting methods are possible. The way I aggregated benefits provided the best possible reserve systems (see ECOCENTRIC and SOCIAL-

ECOLOGICAL scenarios), in terms of reduced costs across all benefits, compared to all other scenarios focused on individual benefits.

My data allowed me to go further into testing the way socioeconomic information is included in planning: as a constraint only or as an objective. Although the ECOCENTRIC and SOCIAL-ECOLOGICAL scenarios yielded very similar results due to the biophysical and socioeconomic context of the Madang Lagoon, this might not be the case elsewhere or if the quantitative objectives and constraints were different. This highlights the importance of clearly setting objectives at the start of a conservation-planning exercise, and contributes to the debate about what should be prioritised: biodiversity, human well-being and livelihoods, or both.

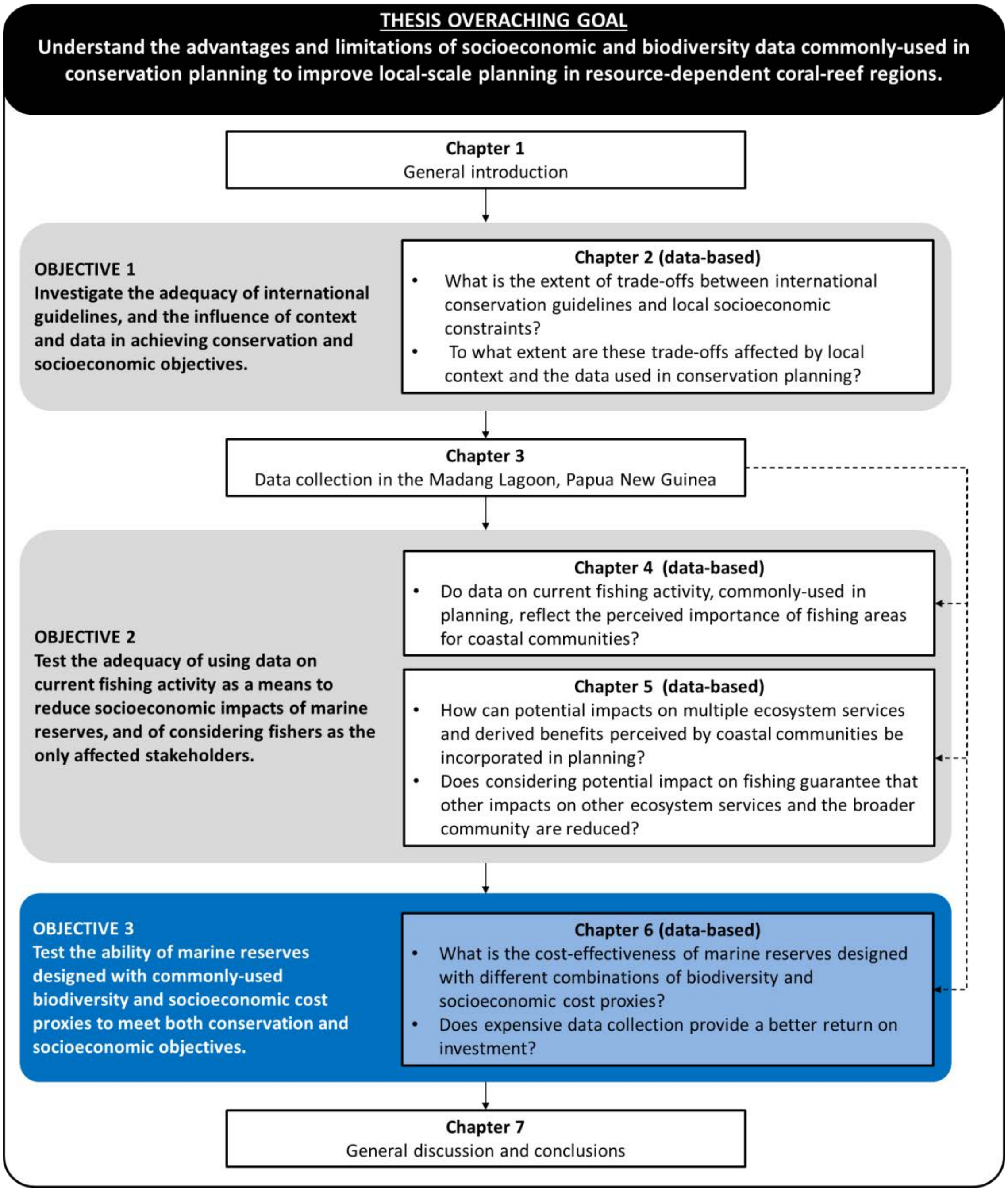
Here I provided the first local-scale conservation planning framework that explicitly includes benefits derived from ecosystem services in systematic conservation planning to account for important places to people. The approach has potential to be replicated in other realms. Importantly, I showed that incorporating local perceptions does not necessarily compromise conservation objectives or increase impact on the community. Instead, combined with an open, transparent, and ongoing engagement process, the approach has potential to improve the likelihood of success of conservation plans in similar settings.

Chapter 6.[1]

Benefits and costs of using proxies of biodiversity value and socioeconomic impacts in coral-reef conservation planning

In this chapter, I test the ability of marine protected areas designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives. I compile several commonly-used proxies of biodiversity (habitat maps of different thematic resolutions created in Chapter 3) and several commonly-used proxies of socioeconomic costs (created in Chapter 4) and calculate the dollar costs of collecting them. Then I design protected areas that aim to protect biodiversity while minimising socioeconomic costs using all possible combinations of proxies. I develop an index to measure the effectiveness of resulting reserve systems to represent a reference biodiversity dataset (species) while minimising a reference cost (developed in Chapter 5).

[1] This chapter was slightly modified from the version prepared for publication to avoid redundancies. The manuscript prepared for publication is: **M. A. Hamel**, R. L. Pressey, S. Andréfouët (in preparation) Benefits and costs of using surrogates for biodiversity value and socioeconomic impacts in coral-reef conservation planning. Conservation Biology. Details on data collection are described in Chapter 3, and some of the supplementary material was directly included in the main text.


THESIS OVERACHING GOAL
Understand the advantages and limitations of socioeconomic and biodiversity data commonly-used in conservation planning to improve local-scale planning in resource-dependent coral-reef regions.
Chapter 1
General introduction
OBJECTIVE 1
Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.
Chapter 2 (data-based)
• What is the extent of trade-offs between international conservation guidelines and local socioeconomic constraints?
• To what extent are these trade-offs affected by local context and the data used in conservation planning?
Chapter 3
Data collection in the Madang Lagoon, Papua New Guinea
OBJECTIVE 2
Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders.
Chapter 4 (data-based)
• Do data on current fishing activity, commonly-used in planning, reflect the perceived importance of fishing areas for coastal communities?
Chapter 5 (data-based)
• How can potential impacts on multiple ecosystem services and derived benefits perceived by coastal communities be incorporated in planning?
• Does considering potential impact on fishing guarantee that other impacts on other ecosystem services and the broader community are reduced?
OBJECTIVE 3
Test the ability of marine reserves designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.
Chapter 6 (data-based)
• What is the cost-effectiveness of marine reserves designed with different combinations of biodiversity and socioeconomic cost proxies?
• Does expensive data collection provide a better return on investment?
Chapter 7
General discussion and conclusions

6. Benefits and costs of using proxies of biodiversity value and socioeconomic impacts in coral-reef conservation planning

Abstract

Conservation plans are all based on proxy measures of biodiversity as a whole and of the socioeconomic value of different places, especially in data-poor regions. The use of proxies is often supported by the common beliefs that they are usually easier, quicker and cheaper to collect than comprehensive data, and that they adequately represent the information of interest. Rarely tested, these assumptions can lead to unrealistic expectations and ineffective conservation actions. Here I used Marxan to design marine reserve systems that maximise biodiversity while minimizing costs to the local community in the Madang Lagoon (Papua New Guinea). I used 4 commonly-used biodiversity proxies in combination with 6 commonly-used proxies of socioeconomic costs. I measured the cost-effectiveness of reserve systems based on the 24 combinations of proxies. Effectiveness was the ability of a system to represent species and to minimise loss of important ecosystem services for the local community. I developed and used a novel index of proxy effectiveness for this context. Cost was the total monetary cost of data acquisition for each combination of proxies used to design the system. Cost-effectiveness varied greatly with the different combinations of biodiversity and cost proxies. The effectiveness of different proxy combinations also varied with the taxa for which species representation was assessed. The total cost of each combination of proxies had little to do with effectiveness of reserves designed with these proxies. Lack of comprehensive reference data, one of the major challenges when testing for proxy effectiveness, could explain the absence of clear trend. However, using a world-best coral-reef dataset at local scale, my results confirm that assessments of proxy effectiveness are highly context-dependent and disprove the assumption about return on investment in data collection. Implications are important for conservation planning and it is recommended that practitioners systematically assess risks in using the data upon which they base important conservation decisions.

Introduction

One of the foundations for the success of protected areas designed using systematic conservation planning is relevant, comprehensive, and up-to-date spatial datasets: on biodiversity to maximise its protection, and on socioeconomic costs to minimise potential negative impacts of protection on resource-dependent people (Pressey and Bottrill, 2009). However, comprehensive, high-resolution information on biodiversity and socioeconomic costs is seldom available for a whole region of interest. To overcome the lack of resources to collect such data, and recognising the urgency of implementing conservation actions, conservation planners typically use data on proxies (also termed 'surrogates') of the features of primary concern (Margules and Pressey, 2000). Commonly used proxies of biodiversity include information on specific taxa, habitats types, and abiotic environmental variables. Perhaps less well recognised but equally common is the use of proxy variables for socioeconomic costs of conservation actions, for similar reasons (Pressey et al., 2013). Proxies of opportunity costs to stakeholders, for example, are typically quantified through the economic value of the areas of interest, their potential yield, or the effort given to resource extraction, sometimes measured for specific stakeholder groups. In theory, proxies are chosen based on their ability to represent the features of primary interest, and because data on proxies are quicker, easier, and cheaper to collect. In practice, conservation planners use proxies based on untested (and sometimes unstated) assumptions about their cost-effectiveness (Rodrigues and Brooks, 2007, Carwardine et al., 2008, Caro and Girling, 2010).

In effect, systematic conservation planning designs protected areas to achieve objectives for proxies of biodiversity while minimising proxy metrics of socioeconomic costs. It follows that understanding the cost-effectiveness of using both kinds of proxies to design reserves is crucial to ensure actual protection of biodiversity with minimal impact on affected communities. Below, I refer to the cost-effectiveness of proxies as the ability of protected areas designed with these proxies at protecting biodiversity and/or minimising socioeconomic costs (effectiveness), in relation to the costs of collecting data on these proxies (cost).

A large amount of work has been conducted on measuring the ability of proxies (and of protected areas based on proxies) to represent biodiversity features of interest (see Chapter 1 for examples). However, the literature shows scattered and conflicting results. General characteristics of "good" proxies are hard to identify because many factors can influence the effectiveness of these proxies (Sarkar and Margules, 2002, Caro and Girling, 2010, Grantham et al., 2010, Lewandowski et al., 2010, Mellin et al., 2011, Andréfouët et al., 2012a). Factors influencing the apparent effectiveness of proxies include: the type of proxy and the metrics (also termed 'indicators') used to measure it; the reference data on the feature of primary interest (for testing the proxy) as well as the metrics used to measure it; biogeographic and socioeconomic aspects of the study region; the spatial extent of the study region; data resolution; and the method used to measure effectiveness.

In contrast, the effectiveness of different socioeconomic cost proxies at minimising socioeconomic impacts of conservation actions on stakeholders has been little investigated (but see Adams et al., 2010, Weeks et al., 2010) and similar inconsistencies are expected. For example, socioeconomic costs can be estimated through opportunity costs (or the costs of foregone opportunities) (Naidoo et al., 2006). Such costs can be measured for a variety of involved stakeholders (e.g. commercial fishers, recreational fishers, tourism operators in a marine setting) and in a variety of ways (e.g. lost revenue, area, intensity of use).

Also little explored are comparisons of the effectiveness of proxies in relation to the cost of collecting them. The costs associated with conservation planning have increasingly received attention in the literature, mainly focusing on land acquisition, management, transaction, damage and opportunity costs (Naidoo et al., 2006). Surprisingly, however, the cost of data collection for conservation planning and related return on investment have been little studied. Few studies have investigated the cost-effectiveness of different biodiversity datasets for conservation planning (Gardner et al., 2008, Grantham et al., 2008, Hermoso et al., 2015). These have mostly focused on different types of species data. To my knowledge, no published study has looked at the cost-effectiveness of different proxies for species or of socioeconomic cost. Additionally, I am not aware of studies which considered interactions between proxies of biodiversity and socioeconomic costs in measuring the effectiveness of protected areas at representing biodiversity while minimising costs. Testing the effectiveness of reserves based on the combination of both proxies, as opposed to either the biodiversity or the socioeconomic proxy, is critical. Indeed, protected areas are increasingly designed to account for socioeconomic costs to human communities. Therefore, the effectiveness of biodiversity proxies is likely influenced by constraints in the design from cost proxies, and vice versa.

In coral reef ecosystems, collecting comprehensive data on biodiversity and socioeconomic costs to plan for conservation is a daunting yet essential task, given the ever-increasing pressures they face. The difficulty is compounded by the location of most of the world's coral reefs in developing countries. In these countries, constraints on logistics, time, and money are respectively exacerbated by remoteness, the urgent need to protect coral-reef ecosystems given threats, the high dependence of local communities on associated resources, and the global absence of funds for biodiversity and ecosystem conservation. These constraints make the use of and need for "good" proxies particularly acute. However, coral-reef conservation planning is still at its infancy, prompting planners to mimic approaches to data collection used in other contexts and realms, often based on untested assumptions. For instance, habitat maps are often used and promoted as cost-effective proxies of biodiversity in terrestrial conservation planning (Ferrier, 2002, Grantham et al., 2010), mainly due to the increasing availability and decreasing costs of modern remote-sensing products. Reserves are then designed to represent a wide variety of habitats under the assumption that distinct species assemblages will be similarly protected. However, the effectiveness of such reserves at representing biodiversity is still poorly understood in most ecosystems, including coral reefs (e.g. Dalleau et al., 2010, Van Wynsberge et al., 2012, Shokri and Gladstone, 2013). In addition, in marine planning generally, proxies of fishing opportunity costs such as distance from candidate reserves to the coast or landing sites, fishing effort, total catch, or catch per unit effort, are commonly used as proxies for socioeconomic costs (Ban and Klein, 2009). However, little work has been done to assess the relevance and cost-effectiveness of these socioeconomic proxies to forgone benefits to people in coral-reef planning (but see Weeks et al., 2010).

In this chapter, my goal is to assess, for local-scale coral-reef conservation planning, the cost-effectiveness and adequacy of different proxies of biodiversity and socioeconomic costs commonly used in other contexts. To do so, I design reserves based on combinations of proxies of biodiversity and socioeconomic costs. I first investigate the sensitivity of proxy effectiveness to the type of biodiversity proxy, type of socioeconomic proxy, set of planning units used in the analyses, and, when applicable, the reference dataset used to measure effectiveness. Second, I test the assumption that a greater investment in proxy data collection guarantees effectiveness of protected areas designed with different combinations of proxies. I use the Madang Lagoon, Papua New Guinea, as a case study.

Methods

Planning region and planning units

The study area for this chapter was the Madang Lagoon. For the following analyses, the planning region was defined as the extent of marine habitats and islands within the section of the Madang Lagoon used by the Riwo community. I used a grid of 319 square planning units, each 300 m x 300 m or 9 ha, covering the planning region. I considered the implementation of full reserves where access is fully prohibited, as an indicative conservation action. More details on the planning region and planning units can be found in **Figure 3.1** and in Chapter 3, respectively.

General approach

In this subsection, I summarise the general approach for this chapter (**Figure 6.1**). Details are found in the next subsections.

The aim of this chapter was to test the ability of different combinations of proxies for biodiversity and socioeconomic cost to help design marine reserves which achieve good species representation (biodiversity benefit), with a low perceived socioeconomic impact on affected communities (socioeconomic cost), using systematic conservation planning. The three biodiversity proxies consisted of four different classifications of marine habitats in the Riwo waters (form coarse to very detailed), derived from a habitat map covering the Madang Lagoon. For socioeconomic proxies, I created six spatial datasets: five with different types of information on potential costs to the Riwo community, and a spatial dataset with uniform cost, for comparison. I developed 24 conservation planning scenarios to explore all possible combinations of biodiversity and socioeconomic proxies. Each scenario aimed at designing a reserve system that protects a set amount of a given proxy of biodiversity, while minimising a given proxy of socioeconomic costs. I ran each scenario using different sets of planning units available for reservation. To measure the effectiveness of resulting reserves at representing biodiversity, and/or minimising socioeconomic costs to communities, I developed a reference species dataset and a reference socioeconomic dataset. I developed an index (the Proxy Effectiveness Index, or PEI) to measure biodiversity benefits (PEI_{bio}) and socioeconomic cost (PEI_{sec}) of each scenario, based on the reference biodiversity and cost datasets, respectively. I measured PEI_{bio} and PEI_{sec} for all 24 scenarios ran for each set of planning units, except when I used all planning units, in which case I could only measure PEI_{sec} because the reference biodiversity dataset was not available for the whole pool of planning units. I also measured the costs, in Australian dollars, of collecting each proxy dataset.

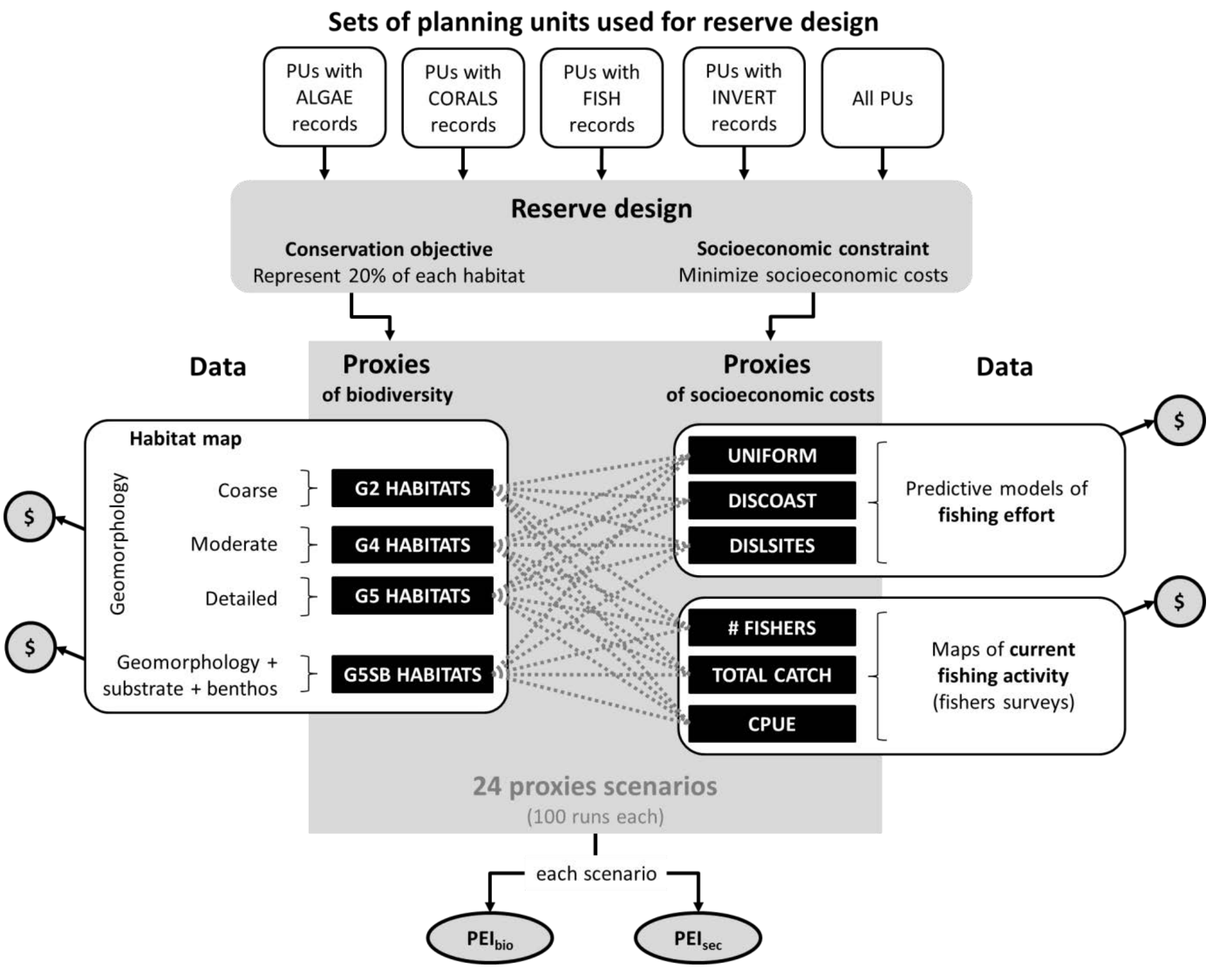


Figure 6.1. General approach for this chapter. Proxies of biodiversity include the types and extent of each habitat for habitat maps G2, G4, G5 and G5SB described in Chapter 3. Proxies of socioeconomic costs are measured based on a uniform cost (UNIFORM), the distance from each planning unit to land (DISCOAST), to the main landing sites (DISLSITES), the number of fishers (# FISHERS), the total catch (TOTAL CATCH), and the average catch per unit effort (CPUE).

Datasets

I compiled four datasets for this chapter: biodiversity proxies (extent of habitat types within planning units for four habitat classifications), proxies of socioeconomic costs (five measures of fishing opportunity costs in each planning unit, and one that considered uniform costs for all planning units), reference biodiversity (species presence in selected planning units for macro-algae, corals, fish, and invertebrates), and reference socioeconomic cost (the perceived importance of planning units for the ecosystem services and benefits they provide). Details for each dataset are described below.

The four biodiversity proxies consisted of four different classifications of marine habitats in the Riwo waters, derived from a habitat map covering the Madang Lagoon. Chapter 3 gives details on the creation of the habitat map, typology, and classification. I used the three following geomorphologic habitat classifications: G2 (coarse thematic resolution, 4 habitat classes in the planning region); G4 (moderate thematic resolution, 13 habitat classes); and G5 (fine thematic resolution (22 habitat classes). I also created G5SB, a classification with very fine thematic resolution (107 habitat classes) that combined geomorphology G5, substratum stability S, and benthic cover B. For each of the four classifications, I derived, information on the extent of all habitats (in square metres) in each of the 319 planning units This resulted in four different sets of biodiversity proxies, hereafter G2, G4, G5, and G5SB respectively.

As a reference biodiversity dataset, I used records of marine species from the biological expedition, with surveyed sites chosen to cover a diversity of marine habitats. Chapter 3 gives details on data collection. I used 172 species of macro-algae recorded from 29 different sites across the planning region, 258 coral species from 19 sites, 472 fish species from 35 sites, and 226 macro-invertebrate species from 20 sites. These taxa would benefit from marine reserves through, for example, regulating extractive activities and/or destructive human uses within their boundaries. To each planning unit, I allocated all species recorded within 20 m of its boundary. Because any planning unit could contain several surveyed sites, this allocation resulted in 26 planning units with a list of macro-algae species, 18 for corals, 30 for fish, and 16 for macro-invertebrates. Out of a total of 319 planning units, this meant that direct records of species in any of the four taxonomic groups were relatively sparse. However, due to the size and expense of the expedition, the species data were fairly unique as taxonomic inventories generally include fewer biological groups (often fish and corals only), or groups that have been sampled in different sites throughout the targeted area in the course of non-related research programs (but see Cleary et al., 2008, Dalleau et al., 2010, Jimenez et al., 2012).

The six proxies of socioeconomic costs consisted of five spatial datasets to represent possible socioeconomic costs to stakeholders, and one that considered spatially homogenous costs, for comparison. Chapter 3 gives details on the creation of the cost layers. For all 319 planning units, I measured socioeconomic costs based on commonly used predictive models of fishing effort: 1) the distance to the nearest land (DISCOAST); and 2) the distance to any landing site (DISLSITES). Such measures are used in conservation planning, on the assumption that protecting areas closer to the coast, landing sites, or ports, where fishing effort is assumed to be concentrated, will have a higher impact on fishing communities (e.g. Green et al., 2009, Weeks et al., 2010, Giakoumi et al., 2013, Mazor et al., 2014). Three additional cost layers were derived from actual data on current catch and fishing effort in the Riwo waters based on fisher surveys (details in Chapter 3). For each of the 319 planning units, I derived: 1) information on the number of fishers visiting for fishing (# FISHERS); 2) the total catch recorded (TOTAL CATCH); and 3) the average catch per unit effort recorded (CPUE). Each of the five datasets for socioeconomic proxies was scaled to 100, with 100 being the maximum recorded value within the planning region for the respective proxy. A spatially uniform cost layer (UNIFORM), where all 319 planning units had the same cost of 100, was also created.

As a reference socioeconomic cost dataset, I used information on the importance of each planning unit for the ecosystem services and derived benefits they provide, as perceived by Riwo people. During the expedition, I conducted socioeconomic surveys that included the collection of spatial data on the importance of areas in the lagoon that people access for a variety of social, economic, and cultural benefits: fishing, recreation, aesthetics, spirituality, traditional medicine, collecting material to make lime to chew with betel nuts, education and knowledge sharing, and perceived biological richness. A cost layer representing the importance of each of the 319 planning units for all these benefits combined was created. Chapter 3 gives details on data collection, and Chapter 5 gives details on the creation of this spatial cost layer (hereafter "PERCEIVED ALL"). Here, I assumed that reserving a planning unit will incur a cost proportional to its importance for the derived benefits perceived by the coastal community.

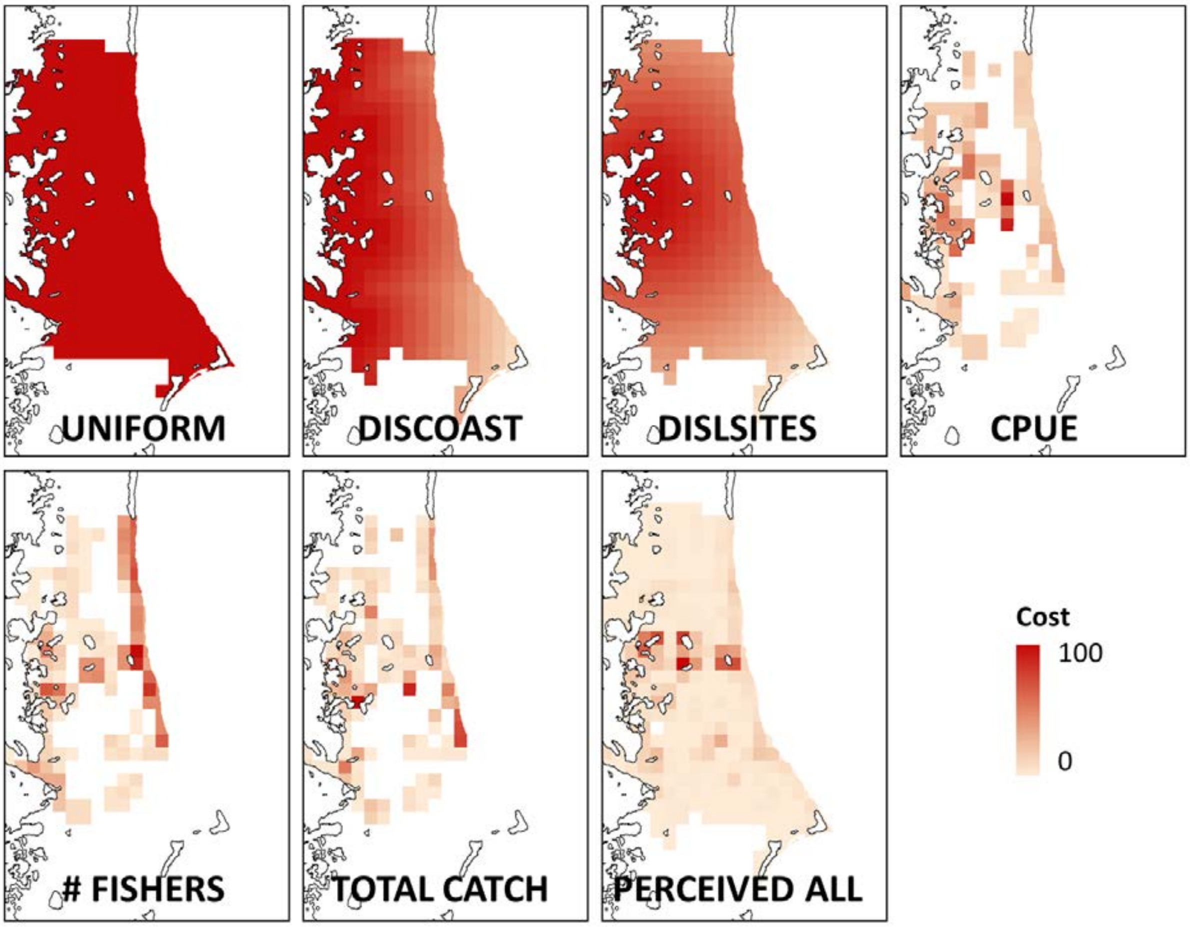


Figure 6.2. Spatial layers of socioeconomic cost proxies used in the present study. Socioeconomic cost proxies are: no cost (UNIFORM), cost inversely proportional to the distance from land (DISCOAST), cost inversely proportional to the distance from landing sites (DISLSITES), cost proportional to the number of fishers (# FISHERS), cost proportional to the total catch (TOTAL CATCH), and cost proportional to the average catch per unit effort (CPUE). The reference cost used to measure the performance of scenarios (PEIsec) is proportional to the importance of areas for the perceived ecosystem services they provide to the broader community (PERCEIVED ALL).

Conservation planning with proxies

To develop the conservation planning scenarios, I needed planning units with data on: all types of biodiversity proxies, all types of socioeconomic proxies, reference biodiversity data (species lists), and reference socioeconomic data (importance for derived ecosystem benefits). Data on all biodiversity proxies, socioeconomic proxies, and the reference socioeconomic data were available for all 319 planning units, but only a subset of planning units had reference biodiversity data. Modelling species distributions across all 319 planning units was not an option given the lack of independent environmental data available at fine enough spatial resolution. Moreover, using the same habitat data for distribution modelling and testing for effectiveness of proxies would have biased the results. Therefore, in my analyses, I used only planning units for which species lists were available. This considerably reduced my sample size but avoided likely bias in my measure of biodiversity benefit. Different sets of planning units were therefore used for each of the four taxa of interest: 26 for macro-algae, 18 for corals, 30 for fish, and 16 for macro-invertebrates.

To select candidate reserves based on biodiversity and socioeconomic cost proxies, I used the Marxan software (Ball et al., 2009). The required input data for Marxan typically consist of biodiversity features in each planning unit (here, biodiversity proxies or reference biodiversity data) and on the cost of protecting each planning unit (here, socioeconomic cost proxies or reference cost data). With this information, the software uses a simulated annealing algorithm to find optimal or near-optimal solutions: systems of protected areas that achieve conservation objectives for all biodiversity features for a minimum total cost. In the 24 scenarios using all possible combinations of biodiversity and socioeconomic proxies, the objective was to represent at least 20% of the extent of each biodiversity proxy (habitat types) contained in the set of planning units available for reservation, while minimising the potential socioeconomic impacts of protection on the Riwo community, according to socioeconomic proxies (**Figure 6.1**). Because the sets of planning units with species lists were different for the four taxa of interest, all 24 scenarios were repeated for each of the four different sets. Each scenario involved

100 repeat runs of Marxan to provide multiple solutions that all achieve objectives. I also ran all 24 scenarios 100 times using all 319 planning units.

Proxy effectiveness index

To measure the effectiveness of reserves designed with each combination of proxies, I needed: 1) a metric to assess scenario performance at representing species within reserve systems (biodiversity benefit, as per the reference biodiversity dataset); and 2) a metric to assess scenario performance at minimising costs to the Riwo community in terms of loss of access to ecosystem benefits (socioeconomic cost, as per the reference socioeconomic cost). These metrics had to be comparable between scenarios. I developed the Proxy Effectiveness Index (PEI) to measure the effectiveness of protected areas designed with an optimisation algorithm and input data on proxies for biodiversity and socioeconomic cost. PEI was based on the Species Accumulation Index (SAI), a commonly-used standardised approach (Rodrigues and Brooks, 2007) developed by Ferrier and Watson (1997) for protected areas designed with a sequential selection algorithm.

PEI can be calculated to measure the effectiveness (or performance) of scenarios at maximizing biodiversity benefits (PEI_{bio}) based on reference biodiversity data, or at reducing socioeconomic costs (PEI_{sec}) based on reference costs data.

For a given scenario, and a given set of planning units used for reserve design, PEI is calculated as:

$$PEI = \frac{s - r}{o - r}$$

With:

- s : biodiversity benefit (s_{bio}) or socioeconomic cost (s_{sec}) of the best protected-area system designed with a “proxies scenario”. The proxies scenario optimises the representation of a given biodiversity proxy while minimising a given socioeconomic cost proxy. The letter “S” stands for “surrogate”, to use the same terminology as in Ferrier and Watson (1997) ;

- r : biodiversity benefit (r_{bio}) or socioeconomic cost (r_{sec}) of protected areas selected randomly. The letter "R" stands for "random". Comparing proxy scenarios to random selections is important because it provides information on the extent to which using data on proxies improves effectiveness of reserves over and above using no data; this comparison is not informed by how closely proxy scenarios approach best-possible outcomes (below);
- o : biodiversity benefit (o_{bio}) or socioeconomic cost (o_{sec}) of the best-possible protected-area system for the representation of biodiversity, or for the reduction of socioeconomic costs, respectively. The best-possible system for biodiversity benefits is designed with an "optimum biodiversity scenario", whereas for socioeconomic costs, it is designed with an "optimum socioeconomic scenario". The letter "O" stands for "optimum".

I measured the variables s_{bio}, s_{sec}, r_{bio}, r_{sec}, o_{bio} and o_{sec} as described below for each of the 24 proxy scenarios run on one of the five possible sets of planning units (i.e. planning units containing sites for either macro-algae, corals, fish or macro-invertebrates and all planning units).

To measure s_{bio}, I identified the best reserve system produced by Marxan. The best reserve system out of the 100 was defined as the system with the smaller number of planning units that achieved the representation objectives for all habitats, with a minimum socioeconomic cost. In case of ties, I selected the reserve system produced in the first run of the list of runs with ties. Then I measured, for the best system, how many species belonging to the taxon corresponding to the chosen set of planning units were represented (biodiversity benefit, based on the reference dataset). For example, if the initial set of planning units was that with algae species records, I measured the number of algae species represented in the best system.

To measure r_{bio}, I first recorded u, the number of planning units selected in the best reserve system identified above. Then, using the same initial set of planning units as used in the corresponding proxies scenario, I selected randomly u planning units 100 times. I measured the mean number of represented species belonging to the taxon corresponding to the chosen set of planning units (biodiversity benefit, based on the reference dataset). Using the same number of planning units was critical because not keeping this variable constant would bias

PEI measures. First, the more planning units selected, the more effective for biodiversity benefits a random scenario will appear, as it will ultimately allow more species to be protected. The reverse would apply if fewer planning units were selected at random that in the scenario being tested. Second, a random scenario will appear less effective at minimizing socioeconomic costs with more planning units selected because additional cost will be incurred. Sensitivity of proxy effectiveness to the number of planning units selected is even more relevant in cases with small numbers of planning units. More information on the bias caused by the number of planning units available can be found in **Appendix 8**.

To measure o_{bio}, I defined an optimum scenario for biodiversity benefits. An ideal optimum scenario for biodiversity benefits would seek to use the reference biodiversity data to provide maximum biodiversity benefits, while minimising each given socioeconomic cost proxy, for the same number of planning units u as the one used in both the proxies and random scenarios. Indeed, because u planning units are used in both the proxies scenario and the random scenario, the optimum scenario must be constrained similarly to allow for an unbiased comparison of scenarios. However, Marxan typically solves the "minimum set" reserve design problem, which means it is not designed limit selections to a given number of planning units unless costs are uniform. Therefore, here I defined the optimal scenario for biodiversity benefits as one that aims for the representation of as many species from the reference dataset as possible in the protected-area system, without socioeconomic cost constraints, and using u planning units. This allowed me to use a uniform cost (UNIFORM), and therefore control the maximum number of planning units u to reserve. Because in the uniform cost layer, each planning unit had a cost of 100, I set a cost threshold of 100 times the number of planning units recorded for the proxies scenario to control the maximum number of planning units to reserve.

I ran the optimum biodiversity scenario 100 times, with a ceiling on the maximum number of planning units u that can be reserved. I identified the best protected-area system produced by Marxan as the one that represented the largest number of species belonging to the taxon of interest. In case of ties, I selected the reserve system produced in the first run of the list of runs with ties. I then recorded the number of represented species belonging to the taxon corresponding to the chosen set of planning units (biodiversity benefit, based on the reference dataset).

To measure s_{sec} and r_{sec}, I followed the same steps as for s_{bio} and r_{bio}, except that I measured the total socioeconomic cost of the reserve system (based on PERCEIVED ALL, indicating lost access to important areas for ecosystem services and benefits as perceived by the Riwo community, as per the reference socioeconomic dataset) instead of the number of species represented. The reference socioeconomic cost PERCEIVED ALL was available for all planning units of the planning region.

To measure o_{sec}, I defined an optimal scenario for socioeconomic costs. An ideal optimum scenario for socioeconomic costs would seek to achieve conservation objectives for the biodiversity proxies, while minimising the reference cost, for the same number of planning units as the one used in both the surrogate and random scenarios. However, for similar reasons as the optimum scenario for biodiversity benefits (see **Appendix 8** for details on bias), the limitations of Marxan prevented this analysis. To our knowledge, no better algorithm was available at the time of the analysis. Therefore, here I defined the optimum scenario for socioeconomic costs as one that aims for the protected areas with the lowest possible socioeconomic costs (i.e. lost access to important areas for ecosystem services and benefits as perceived by the Riwo community, as per the reference socioeconomic dataset), without considering any conservation objectives. This allowed me to control the number of planning units to reserve.

I ran the optimum socioeconomic scenario using u planning units. There was only one best solution, corresponding to the u planning units with the smallest total cost. I then measured the total socioeconomic cost of this reserve system (i.e. lost access to important areas for ecosystem services and benefits as perceived by the Riwo community, as per the reference socioeconomic dataset).

For all 24 proxy scenarios run for each of the sets of planning units with records of macro-algae, corals, fish, and macro-invertebrate species, I measured proxy effectiveness PEI_{bio} and PEI_{sec}. For the 24 proxy scenarios run with the full set of 319 planning units, I measured only proxy effectiveness PEI_{sec} because species records (the reference biodiversity dataset) were not available for all planning units. This resulted in a total of 96 measures of PEI_{bio} (24 for each taxa) and 120 PEI_{sec}(24 for five different sets of planning units). The protocol used to calculate the above-mentioned elements of PEI_{bio} and PEI_{sec} is summarised in **Figure 6.3**. All spatial layers compilation, scenario preparation, Marxan runs, and subsequent analyses were performed using R (R Development Core Team, 2008).

PEI varied from any negative value to 1. A PEI of 1 indicated that the effectiveness of reserve systems based on a given proxy scenario was close to an optimum scenario for either biodiversity or socioeconomic costs. A PEI close to 0 indicated that reserve systems performed close to a random selection of reserves. A negative PEI indicated that reserves systems performed worse than random.

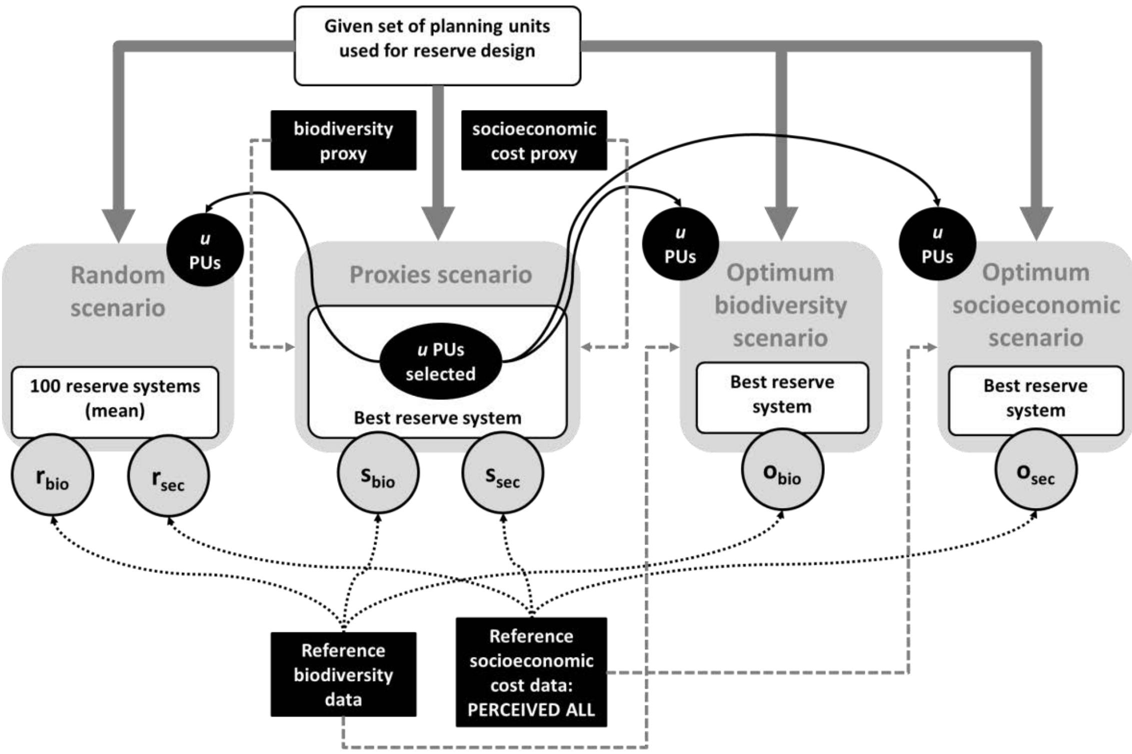


Figure 6.3. Protocol used to calculate all elements of the Proxy Effectiveness Index (PEI). Variables are described in the main text (section “Proxy effectiveness index”).

Factors influencing proxy effectiveness

A common assumption about proxies is that using a proxy which has been shown (or is assumed) to be effective in a specific situation, will be effective in most contexts. Here I aim to investigate the sensitivity of proxy effectiveness to the type of socioeconomic cost proxy, type of biodiversity proxy, set of planning units used in the analysis, and, when applicable, the taxon of reference. To assess the presence of significant effects of these interacting factors on the effectiveness of biodiversity proxies and of socioeconomic proxies, I performed an analysis of variance on PEI_{bio} (model: PEI_{bio} ~ socioeconomic cost proxy + biodiversity proxy + taxon), and PEI_{sec} (model: PEI_{sec} ~ socioeconomic cost proxy + biodiversity proxy + taxon), respectively, for scenarios based only on planning units with species records. I looked at effects and interactions between all factors. If differences were identified, I performed a posthoc Tukey test to identify which pairs of proxies were statistically different. To assess the presence of significant effects of these interacting factors on the effectiveness of socioeconomic proxies for scenarios that used all 319 planning units, I performed an analysis of variance on PEI_{sec} (model: PEI_{bio} ~ socioeconomic cost proxy + biodiversity proxy). I compared results for PEI_{sec} scenarios using reduced sets of planning units and those using the full set so assess the sensitivity of $PEI_{sec}$to the number of planning units used in the analysis.

Cost of collecting biodiversity and socioeconomic data

I calculated the costs of creating each spatial layer used in this study for all proxies and reference datasets. Data collection requires three main categories of resources: money, time, and technical expertise (Gardner et al., 2008). To measure monetary costs of the reference datasets for biodiversity, I assumed brand new field survey equipment for each dataset. Estimates of time spent collecting and processing taxonomic data, as well as equipment costs and salaries were obtained through a short questionnaire sent directly to taxonomists (**Appendix 9**). I converted time and expertise in monetary costs by multiplying the time spent for each staff member by the relevant salary scale. Approximate salaries, including in-kind contributions of respective institutions when applicable, were obtained for all staff at the time of the study. For habitat maps (methods detailed in Chapter 3), I calculated, in monetary value, the time and expertise required to process field data and digitise the maps for a graduate student (as in this study) and an expert, and found that both situations produced equivalent costs.

To ensure my cost structure was as consistent as possible, some costs were excluded for the creation of all datasets. Capital costs, such as accommodation, travel, and medicals, were not considered since they vary with the location of origin of each staff member, and the study location. Because the habitat mapping and taxonomic surveys were done from a research vessel as part of a large international biodiversity expedition, which could highly bias my costs, I calculated representative standard boating costs per day for the region instead. Other capital costs such as office and laboratory equipment and diving equipment were excluded because they were provided by the respective institutions to staff members. Hidden costs such as project management and planning or logistic support were also excluded.

Results

Effectiveness of reserve systems at representing species

Reserves designed to represent the four proxies of biodiversity (G2, G4, G5 and G6SB) while minimising the six proxies of socioeconomic cost varied greatly in their effectiveness at representing species (PEI_{bio}) between and within taxa.

Considering all 24 scenarios for each taxon separately, the effectiveness of protected areas at representing species of macro-algae and macro-invertebrates was often similar to random selections (PEI_{bio} slightly higher than zero), or even worse than random (PEI_{bio} lower than zero) (**Table 6.1** and **Table 6.2**). In contrast, the effectiveness of protected areas at representing species of corals and fish was better than random and closer to that of protected areas designed to represent the species themselves (PEI_{bio} closer to 1).

The habitat classification used as a proxy of biodiversity, the type of cost proxy used, and the taxon for which effectiveness was measured, as well as all interactions between these factors, all influenced the performance of scenarios in representing species (PEI_{bio}, adjusted R^2=0.7, p-value<0.05), and in minimising socioeconomic costs to the Riwo community (PEI_{sec}, adjusted R^2=0.7, p-value <0.05). The effect of both proxies and taxa was even stronger for scenarios that used the whole set of planning units (PEI_{sec}, adjusted R^2=0.9 p-value <0.05).

Acknowledging the effects of all factors, the *post hoc* Tukey test revealed significant differences in PEI_{bio} between taxa when considering all 24 scenarios and all taxa together. Pairwise comparisons between taxa revealed that PEI_{bio} was significantly higher for fish and corals compared to algae and invertebrates, and that there was no significant differences between taxa within each of these two pairs. Significant differences between pairs of biodiversity proxies were also observed. Considering the effects of other factors, G5 and G5SB, as well as G4 and G5 were not different, but PEI_{bio} was significantly better when targeting the representation of G4 rather than G5SB or G2. The effects of different cost proxies on PEI_{bio} were only significantly different between scenarios using CPUE and UNIFORM, and between those using DISLSITES and UNIFORM. Indeed, using a UNIFORM cost (i.e. no constraint on cost) was significantly better at producing scenarios that represent species well, than using CPUE or DISLSITES.

Table 6.1. Effectiveness of each scenario at representing species of algae, corals, fish, and invertebrates measured with the Surrogacy Effectiveness Index PEI_{bio}. A scenario aims for the representation of a biodiversity proxy while minimising a socioeconomic proxy. For each scenario ran on a set of planning units with records on a given taxa, PEI_{bio} is measured using the reference biodiversity data for this taxa. One planning unit measures 9 hectares.

Set of planning units (PUs)	Socioeconomic cost proxy	Biodiversity proxy G2	G4	G5	G5SB	Reference biodiversity data
PUs with algae records	UNIFORM	0.2	-0.3	-0.1	-0.3	Algae species
	DISCOAST	-0.7	-0.3	-0.1	-0.4	
	DISLSITES	-0.5	-0.4	-0.1	-0.1	
	# FISHERS	-0.1	0.1	0.2	-0.1	
	TOTAL CATCH	-0.2	0.0	-0.1	-0.3	
	CPUE	0.0	0.1	-0.7	-0.1	
PUs with coral records	UNIFORM	0.4	0.5	0.2	0.4	Coral species
	DISCOAST	0.1	0.6	0.4	0.1	
	DISLSITES	-0.2	0.1	-0.1	0.1	
	# FISHERS	-0.5	0.2	-0.2	0.4	
	TOTAL CATCH	0.4	0.2	0.4	0.2	
	CPUE	-0.5	0.2	-0.1	0.1	
PUs with fish records	UNIFORM	0.4	0.3	0.4	0.2	Fish species
	DISCOAST	0.3	0.4	0.6	0.2	
	DISLSITES	0.0	0.2	0.7	0.2	
	# FISHERS	-0.3	0.6	0.4	0.1	
	TOTAL CATCH	-0.1	0.3	0.2	0.2	
	CPUE	0.3	0.2	0.3	0.2	
PUs with invertebrates records	UNIFORM	0.2	0.6	0.1	-0.3	Invertebrate species
	DISCOAST	-0.4	0.6	0.2	-0.3	
	DISLSITES	-0.7	0.2	0.1	-0.3	
	# FISHERS	-0.6	0.2	0.1	-0.3	
	TOTAL CATCH	-0.7	0.3	0.2	-0.3	
	CPUE	-0.7	0.1	-0.4	-0.3	

Table 6.2. Effectiveness of each scenario at minimising loss in terms of access to places perceived as important for the provision of ecosystem services and benefits for people, measured with the Surrogacy Effectiveness Index PEI_{sec}. A scenario aims for the representation of a biodiversity proxy while minimising a socioeconomic proxy. For each scenario ran on a set of planning units with records on a given taxa, or on all planning units, PEI_{sec} is measured using the reference socioeconomic cost. One planning unit measures 9 hectares.

Set of planning units (PUs)	Socioeconomic cost proxy	Biodiversity proxy				Reference socioeconomic cost data
		G2	G4	G5	G5SB	
PUs with algae records	UNIFORM	0.0	-0.6	-0.2	0.1	Perceived importance of places for ecosystem services and benefits
	DISCOAST	0.4	0.2	-0.5	-0.1	
	DISLSITES	0.7	0.7	0.4	0.1	
	# FISHERS	0.6	0.8	0.0	0.2	
	TOTAL CATCH	0.5	0.8	0.0	-0.2	
	CPUE	0.6	0.7	0.4	0.1	
PUs with coral records	UNIFORM	0.6	-0.7	-0.3	-0.4	Perceived importance of places for ecosystem services and benefits
	DISCOAST	0.1	-0.6	0.0	-0.4	
	DISLSITES	0.8	0.5	0.4	-0.4	
	# FISHERS	0.7	0.5	0.4	-0.4	
	TOTAL CATCH	0.7	-0.2	-0.1	-0.4	
	CPUE	0.7	0.4	0.4	-0.4	
PUs with fish records	UNIFORM	0.2	0.1	0.4	0.2	Perceived importance of places for ecosystem services and benefits
	DISCOAST	0.5	0.2	0.2	0.2	
	DISLSITES	0.9	0.5	0.4	0.2	
	# FISHERS	0.9	0.5	0.3	0.2	
	TOTAL CATCH	0.9	0.6	0.4	0.2	
	CPUE	0.7	0.5	0.3	0.2	
PUs with invertebrates records	UNIFORM	0.2	-1.0	-0.4	0.0	Perceived importance of places for ecosystem services and benefits
	DISCOAST	0.4	0.4	-0.7	0.0	
	DISLSITES	0.8	0.7	-0.6	0.0	
	# FISHERS	0.0	0.6	0.2	0.0	
	TOTAL CATCH	0.1	0.3	-0.6	0.0	
	CPUE	0.8	0.7	0.2	0.0	
All PUs	UNIFORM	0.0	0.2	0.0	0.1	Perceived importance of places for ecosystem services and benefits
	DISCOAST	-0.2	0.1	-0.1	0.1	
	DISLSITES	0.3	0.4	0.4	0.1	
	# FISHERS	0.7	0.7	0.7	0.4	
	TOTAL CATCH	0.7	0.8	0.7	0.2	
	CPUE	0.7	0.7	0.7	0.3	

Effectiveness of reserve systems to minimise loss of access to ecosystem services and benefits

Reserves designed to represent the four proxies of biodiversity while minimising the six proxies of socioeconomic cost (UNIFORM, DISCOAST, DISLSITES, # FISHERS, TOTAL CATCH, CPUE) varied highly in their effectiveness to minimise loss of access to ecosystem services and benefits to the broader community (PEI_{sec}).

Whether reserves were designed using only planning units with species records (from 18 to 30 planning units, depending on the taxon) or all planning units (n = 319), significant differences in performance were found between scenarios that used each of the socioeconomic cost proxies. In the scenarios that used different sets of planning units, as well as those that used all planning units, the following was observed. Significant differences between pairs of cost proxies were observed: using a UNIFORM cost (i.e. no constraint on cost) was significantly worse than any other cost layer, except DISCOAST, at minimising socioeconomic costs to the Riwo community. Similarly, using DISCOAST was worse than most other cost layers (DISLSITES, CPUE, #FISHERS) but no significant difference was observed when compared with TOTAL CATCH and with UNIFORM. All biodiversity proxies used in scenarios had significantly different effects on their effectiveness at minimising socioeconomic costs, except G5 and G5SB which were not different.

In scenarios using all 319 planning units, additional differences in PEI_{sec} were observed: minimising TOTAL CATCH was significantly better than DISCOAST, and #FISHERS was significantly better than DISLSITES. However, the difference between UNIFORM and DISLSITES observed when using smaller sets of planning units was not significant.

Comparing differences in the influence of taxa on $PEI_{sec}$was only relevant to identify potential effects of the set of planning units used for each scenario since different sets were used for each taxon. Significant differences in PEI_{sec} were observed between coral and fish, and between fish and invertebrates, but not within each of these two pairs.

Cost of data collection for proxies of biodiversity and socioeconomic costs

Excluding capital and hidden costs, accommodation and travel costs which vary with the staff performing the work and the location of the study, the costs of mapping coral reef habitats for a region like the Madang Lagoon (~40km2) with 40 sampling sites was estimated around AU$5,000 for a coarse (G2), moderate (G4), or fine (G5) classification of geomorphology because it could be done without ground truthing and required only one full-time staff. A very fine description (G5SB) required field data collection and three full-time staff, resulting in a total cost of AU$56,000. The costs of creating spatial layers for socioeconomic proxies were estimated at AU$300 for UNIFORM, DISCOAST, and DISLSITES provided the required spatial layers on coastline and landing sites were readily available. The costs of creating the spatial layers based on fisheries data (68 fishers) were estimated at AU$7,000 for # FISHERS, TOTAL CATCH, or CPUE, including one local field assistant and one full-time staff. Costs of data collection are summarised in **Table 6.3** .

Table 6.3. Summary of costs of collecting each dataset.

Type	Dataset	Proxy	Estimated proxy Cost (AU$)
Biodiversity proxy	Habitat map	G2, G4, G5 (each)	5,000
		G5SB	56,000
Reference biodiversity	Macro-algae	NA	125,000
	Corals	NA	30,000
	Fish	NA	59,000
	Macro-invertebrates	NA	25,000
Socioeconomic cost proxy	Models of fishing effort	UNIFORM, DISCOAST, DISLSITES (each)	300
	Map of current fishing activity	#FISHERS, TOTAL CATCH, CPUE (each)	7,000
Reference socioeconomic cost	Perceived importance of places for ecosystem services and benefits	NA	14,000

Cost-effectiveness of reserves using different combinations of proxies

The relationship between the effectiveness of a scenario at protecting species (PEI_{bio}) and its effectiveness at minimising socioeconomic costs (PEI_{sec}) varied between taxa, and between scenarios for each taxa (**Figure 6.4**). For corals, for instance, there was a clear trade-off between biodiversity benefits and socioeconomic costs: scenarios that provided high biodiversity benefits (high PEI_{bio}) incurred relatively high socioeconomic costs (low PEI_{sec}), and *vice versa*. A similar trade-off was observed for invertebrates, but appeared stronger (very high PEI_{bio} were associated with very low PEI_{sec}). For fish, all scenarios were similar in their performance at both providing biodiversity benefits and minimising socioeconomic costs (small differences in PEI_{bio} and PEI_{sec}). All scenarios for fish appeared to perform relatively well for both indices. In contrast, no particular pattern was identified for algae, for which the performance of scenarios was generally poor and variable for both indices.

The total cost of each combination of proxies had little to do with effectiveness of scenarios using these proxies. For example, for invertebrates, the scenario that provided the best biodiversity benefits for the lowest socioeconomic costs (highest PEI_{bio} and PEI_{sec}) is the one that aimed for the representation of G4 habitats while minimising the DISCOAST coast, one of the least expensive combination of proxies in terms of data collection cost. For corals, the scenario that provided the best biodiversity benefits for the lowest socioeconomic costs aimed to represent habitats G2 while not minimising any socioeconomic cost (UNIFORM cost). For fish and invertebrates, the most expensive combinations of proxies (e.g. protection of G5SB habitats while minimising CPUE or TOTAL CATCH) were among the worst scenarios.

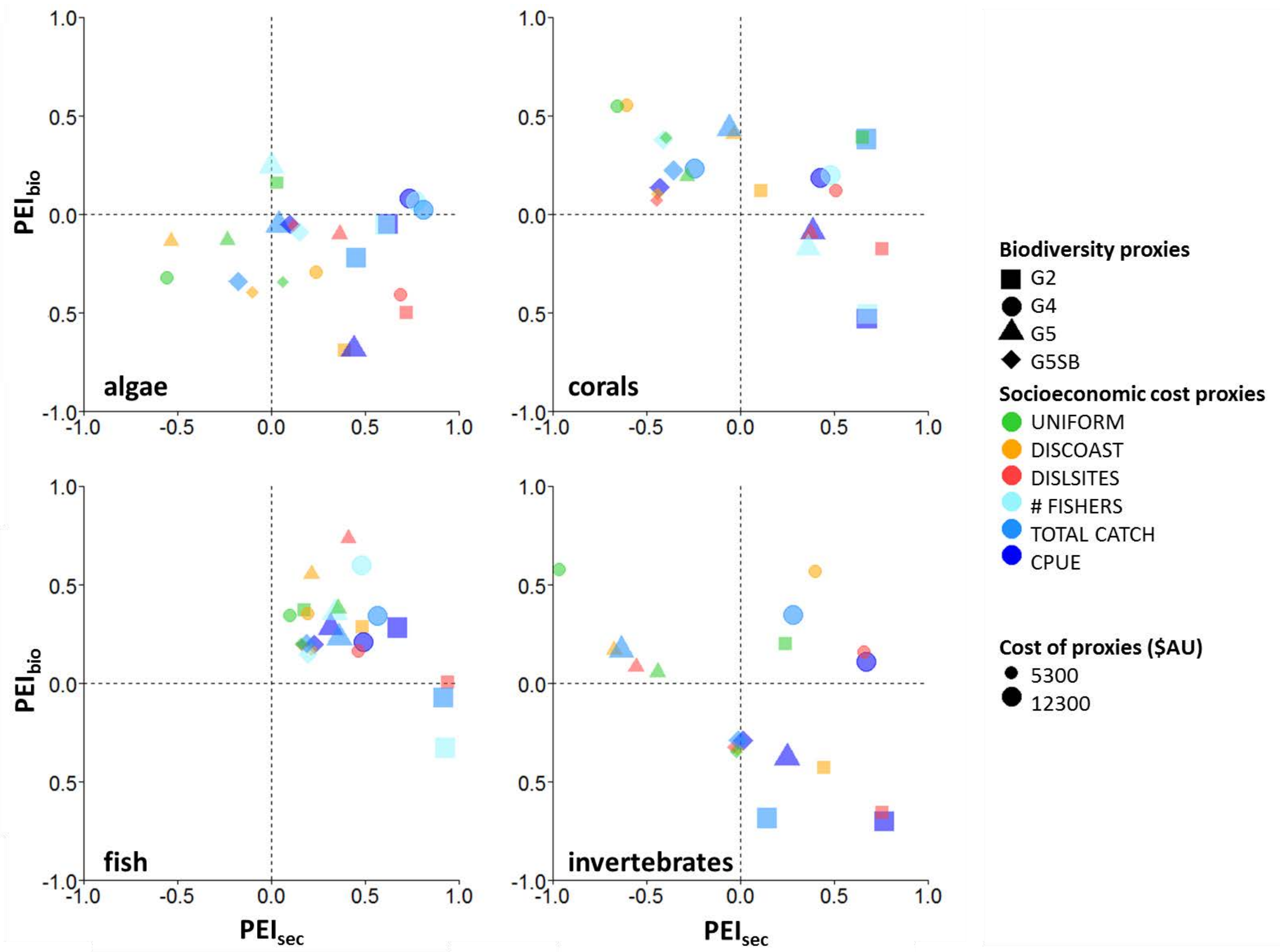


Figure 6.4. Costs and benefits of using proxies of biodiversity and socioeconomic data for algae (top-left), corals (top-right), fish (bottom-left) and invertebrates (bottom-right) conservation planning. The performance of scenarios at representing species (PEI_{bio}) for each taxa is shown on the y-axis, against the performance of scenarios at minimising socioeconomic costs of protected areas to the broader community (PEI_{sec}) shown on the x-axis. Biodiversity proxies (i.e. habitat classifications) that the scenarios aimed at protecting are represented by different symbols. The colour of symbols varies with the socioeconomic cost proxies minimised to design the reserves. A symbol in the top right corner corresponds to a scenario that performs well at protecting a maximum of biodiversity (high PEI_{bio}) for low socioeconomic costs (high PEI_{sec}). The size of symbols is proportional to the total cost (AU$) of the biodiversity and socioeconomic proxies used to design reserve systems (larger symbols are more expensive).

Discussion

Proxies of biodiversity and socioeconomic costs are now used in many conservation projects. A proxy is used, in theory, based on its effectiveness at representing a feature or variable of interest, and on its low data collection cost, compared to directly measuring the variable of interest. In data-poor regions where conservation actions are urgent and funding is limited, knowing which proxy is the most cost-effective can prove essential.

In this study, I investigated the effectiveness of marine reserves designed with different combinations of biodiversity proxies and socioeconomic proxies at representing biodiversity and minimising a reference socioeconomic cost, and I related it to the cost of collecting data on these proxies. In the following "effectiveness" refers to both the representation of biodiversity and the minimisation of socioeconomic costs. I found that some biodiversity or socioeconomic proxies always outperformed others, but overall, the effectiveness of reserves designed to represent a biodiversity proxy while minimising a socioeconomic cost proxy varied with numerous interacting factors as shown in an increasing number of studies (Sarkar and Margules, 2002, Rodrigues and Brooks, 2007, Caro and Girling, 2010, Grantham et al., 2010, Lewandowski et al., 2010, Mellin et al., 2011, Andréfouët et al., 2012a). The biodiversity and the socioeconomic proxies themselves, the taxa used to measure effectiveness at representing species, and the context of the planning region (or initial set of planning units) all influenced the performance of reserves. I also showed that the most expensive combinations of proxies do not guarantee more effective reserves. My findings invalidate several common assumptions about proxies and highlight the need for conservation planners to be cautious about the data they use to make important conservation decisions.

Measuring proxy effectiveness in spatial planning

Actual protected areas whose designs weresupported by systematic conservation planning tools such as Marxan, are increasingly implemented on the ground (The University of Queensland, 2015). Marxan uses a particular algorithm which proposes reserve systems that meet targets for a conservation feature, for the least socioeconomic cost. This approach is based on the premise that socioeconomic costs must be incorporated in reserve design to minimise negative impacts of conservation actions on human communities. Proxies of biodiversity and socioeconomic costs are now used for spatial planning with Marxan, yet, studies which have investigated the effectiveness of proxies in a spatial prioritisation context have mostly focused on biodiversity proxies (e.g. Payet et al., 2010, Januchowski-Hartley et al., 2011). Intuitively, the ability of a reserve designed with Marxan to protect biodiversity is typically constrained by the spatial arrangement of costs in regard to the spatial arrangement of biodiversity features. Similarly, the ability of the reserve to minimise costs is driven by the arrangement of conservation features, which itself drives reserve selection (Naidoo et al., 2006). Quantifying the effectiveness of proxies that will be used for reserve design using a particular tool or algorithm must account for the objectives and constraints of this tool. Therefore, in modern conservation planning, the question is not to know which biodiversity or socioeconomic proxy is best at representing a feature of interest, but which combination of biodiversity or socioeconomic cost provides the best reserve design outcomes. This calls for a shift in approaches to measuring effectiveness than those commonly used (i.e. from pattern-based to selection-based approaches, as discussed in Chapter 1). This study was the first to recognise the need to assess the effectiveness of reserves designed using proxies based on proxy combinations , to propose an approach, and demonstrate its use with exclusively empirical data. In the following, I highlight how this chapter advances research in the field of surrogacy testing, the limits of my approach, opportunities for improvement, and the implications for conservation planning in practice.

The myth of a "one-size-fits-all" proxy

The quest for the most cost-effective biodiversity proxy, has been on since the beginnings of the discipline. Many studies investigated the subject but results are conflicting and the effectiveness of different proxies for conservation planning is still highly debated (see Chapter 1). More recently, several reviews and meta-analyses raised concern about the high context-dependence of proxy effectiveness. Many factors prevent agreements between these studies on the effectiveness of a given proxy at representing a given feature, including the type of proxy and the way it is measured, the type of reference data for the feature and by the way it is measured, as well as the approach to measure effectiveness.

My results confirm that proxy effectiveness is highly context-dependent, both for biodiversity proxies and socioeconomic proxies. I showed that reserves based on the representation of habitats using a map of geomorphologic types with moderate thematic resolution (G4) perform significantly better at representing species, than when using very coarse (G2) or very fine (G5SB) maps, regardless of the taxa used for reference. I also found that designing reserves that minimise uniform costs (UNIFORM) or distance to land (DISCOAST) perform significantly worse than most other costs, regardless of the initial set of planning units used for reserve design. However, I also found that other factors have a significant effect on effectiveness. Biodiversity benefits were not only influenced by the type of biodiversity proxy, but also by the type of socioeconomic cost proxy and the taxa used for reference. Similarly, socioeconomic costs were not only influenced by the type of socioeconomic cost proxy, but also by the biodiversity proxy, and the initial set of planning units used for reserve design. Interactions between these factors were also found to significantly affect effectiveness of reserves designed with proxies.

By definition, a "good proxy" performs relatively well at representing a feature of interest, while being less constraining to collect than the feature it is meant to represent. This study not only investigated the effectiveness of proxies, but also put these results into the context of limited funding for data collection. The cost-effectiveness of proxies was measured as the effectiveness of each combination of proxies to represent biodiversity while minimising socioeconomic cost, in relation to the cost of collecting data on the proxy. Results show that the cost-effectiveness of combinations of proxies is also context-dependent (**Figure 6.4**), which precludes any generalisation. Hence, the choice to not discuss here which proxy is more effective and why, and how these results could help conservation planning in other data-poor coral-reef regions is deliberate. Instead, I focus on how this study on a local coral-reef context can help the broad field of conservation planning move forward.

Towards a standard measure of proxy effectiveness?

Measuring proxy effectiveness in a way that can be compared with other studies is challenging for several reasons, two of which are explained here. First, different studies investigating surrogacy effectiveness use different approaches that ultimately answer different questions about proxy effectiveness. When comparing studies, it is important to examine these questions and their answer carefully. Here, the objective of each combination of biodiversity and socioeconomic cost proxies was to protect as many species as possible, for the smallest possible socioeconomic cost to local communities, based on their access to important ecosystem services and benefits. Although the approach I developed can help answer this specific question in different spatial planning contexts, my results should not be taken out of context. Second, testing the effectiveness of proxies must be done against a reference dataset, representing the features of interest. This is probably one of the major reasons why proxy effectiveness is more often assumed than tested. Indeed, by definition a proxy is used to avoid comprehensive data collection (Rodrigues and Brooks, 2007). But testing their effectiveness involves collecting data on the proxy, as well as on the feature it is meant to represent. What makes the dataset used in this study unique is the fact that I had the (rare) opportunity to collect and have access to comprehensive and consistent empirical data on proxies and reference information for the same study area, over the same timeframe.

With these considerations, I developed the Proxy Effectiveness Index (PEI), a new index adapted from the Species Accumulation Index (SAI), a commonly-used standardised approach (Rodrigues and Brooks, 2007) developed by Ferrier and Watson (1997) for protected areas designed with a sequential selection algorithm. The PEI can be used with reserve optimisation algorithms to measure the effectiveness of a combination of proxies used in reserve design. It is based on information on a biodiversity proxy, a socioeconomic proxy, a reference biodiversity dataset and a reference socioeconomic cost dataset. It considers the effectiveness of reserves at representing a biodiversity feature as per the reference biodiversity dataset, or at minimising the socioeconomic cost of interest, by comparing reserves designed with proxies with randomly selected reserves, and the best possible reserves. Using the PEI, I assessed the effectiveness of 24 different reserve design scenarios (based on 24 different combinations of biodiversity and socioeconomic proxies).

Opportunities for improvement

Because this study was one of the first of its kind, there are a number of caveats to consider. First, I only used one conservation target (representation of 20% of the extent of each biodiversity proxy feature, or habitats). Further research would be needed to understand the sensitivity of effectiveness to conservation targets. Another caveat is related to the nature of empirical species observations, which are typically patchy. This constrained my study to considerably reduce the number of planning units in the initial sets used for reserve design, which likely augmented the sensitivity of my results. However, although results may also be influenced by the species survey design, I used empirical observation of species only, which could be argued to be more representative of the truth as opposed to distribution models. Further studies dedicated to surrogacy testing should carefully consider the implications of their sampling design, although context-dependent constraints on data collection might preclude this. A third limit lies in the calculation of PEI, which used an optimal scenario that ultimately could never be achieved with the same constraints as the proxy scenario (e.g. optimising the representation of the reference biodiversity data, without considering constraints on socioeconomic costs, for the same number of planning units as the best reserve system selected for the proxy scenario). This was the chosen option because the software's limitations prevented me to design an ideal optimal scenario. Further research should attempt to either modify the PEI or the Marxan algorithm to allow for these constraints.

My results shed light on the increasingly demonstrated fact that proxy effectiveness is context-dependent. An excellent opportunity for further research lies in examining what elements of context change this effectiveness, and how. In other words, what are the ecological or social patterns that can drive reserve selection towards proxy effectiveness? For example, would proxy effectiveness be increased or reduced in a planning region with more rare or common species or habitats? What is the effect of spatial scale of analysis and spatial resolution of data?

Implications for conservation planning practice

This study showed that, in a context of conservation planning, measuring proxy effectiveness is difficult, there is likely no "one-size-fits-all" proxy, and studies cannot be compared (or only with great care). Although such results may not provide the expected answer to conservation planners, they emphasize that there is still much to learn about surrogacy effectiveness, despite the hundreds of studies already on the subject. The risks in using proxies that have not been tested in the same context are extreme: allocating limited conservation resources to the wrong places, ineffectively protecting the features to be protected, ineffectively reducing costs to stakeholders, and a general false sense of achievement. **Figure 6.5** shows, conceptually, how the results of this Chapter could be used by conservation practitioners and the level of uncertainty associated with the use of proxies in different common situations. To avoid the severe consequences of using inadequate proxies, managers and planners must ensure they test the proxies they are using in a similar context, find literature that does, or by default, elicit the rationale for using such proxy explicitly in conservation plans.

Figure 6.5. (next page) Conceptual diagram showing uncertainty in conservation outcomes in relation to the type of data used in spatial planning. Datasets are assumed to have been carefully chosen for their adequacy to help achieve conservation objectives. Typically, conservation practitioners collect data on biodiversity and socioeconomic costs, which are then used to feed spatial planning analyses. If datasets for both biodiversity and socioeconomic costs include both proxies and references, proxy effectiveness can be tested as in this chapter. Whether or not the proxy effectiveness test results are "adequate" remain at the discretion of the practitioners. Cases where datasets are incomplete (i.e. data only on biodiversity, only reference data, or only proxy data) are also detailed. An indicative level of uncertainty in conservation outcomes and where this uncertainty lies are shown for each case. For simplicity, this diagram assumes that the data are available for the whole region of interest, which I acknowledge is rarely the case. In particular, reference data will virtually never be available for a whole region. In a case where data are available for only part of the region, planners will need to identify the best upscaling strategy (e.g. to interpolate the data or fill spatial gaps using modelling techniques), or will use a combination of datasets.

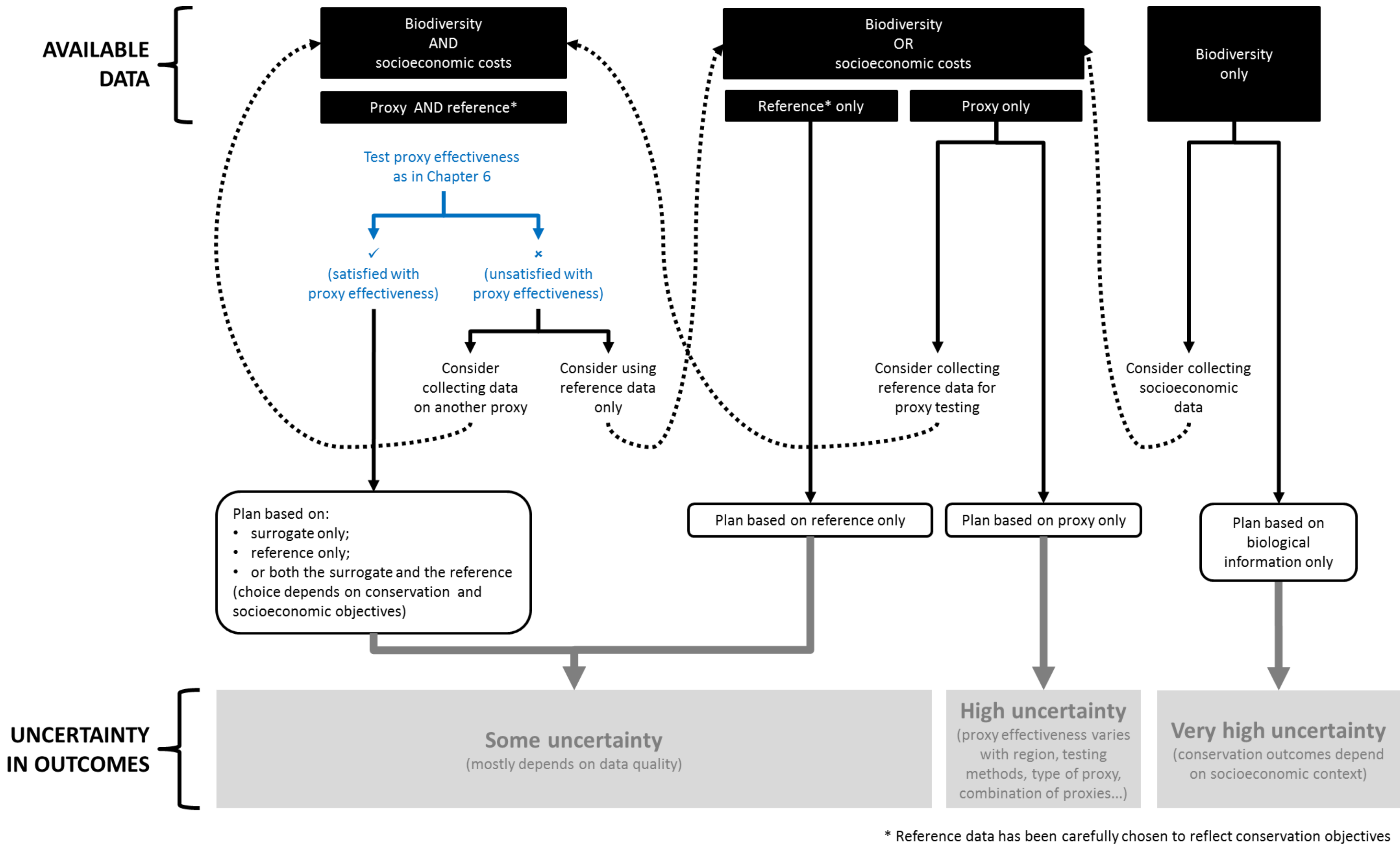
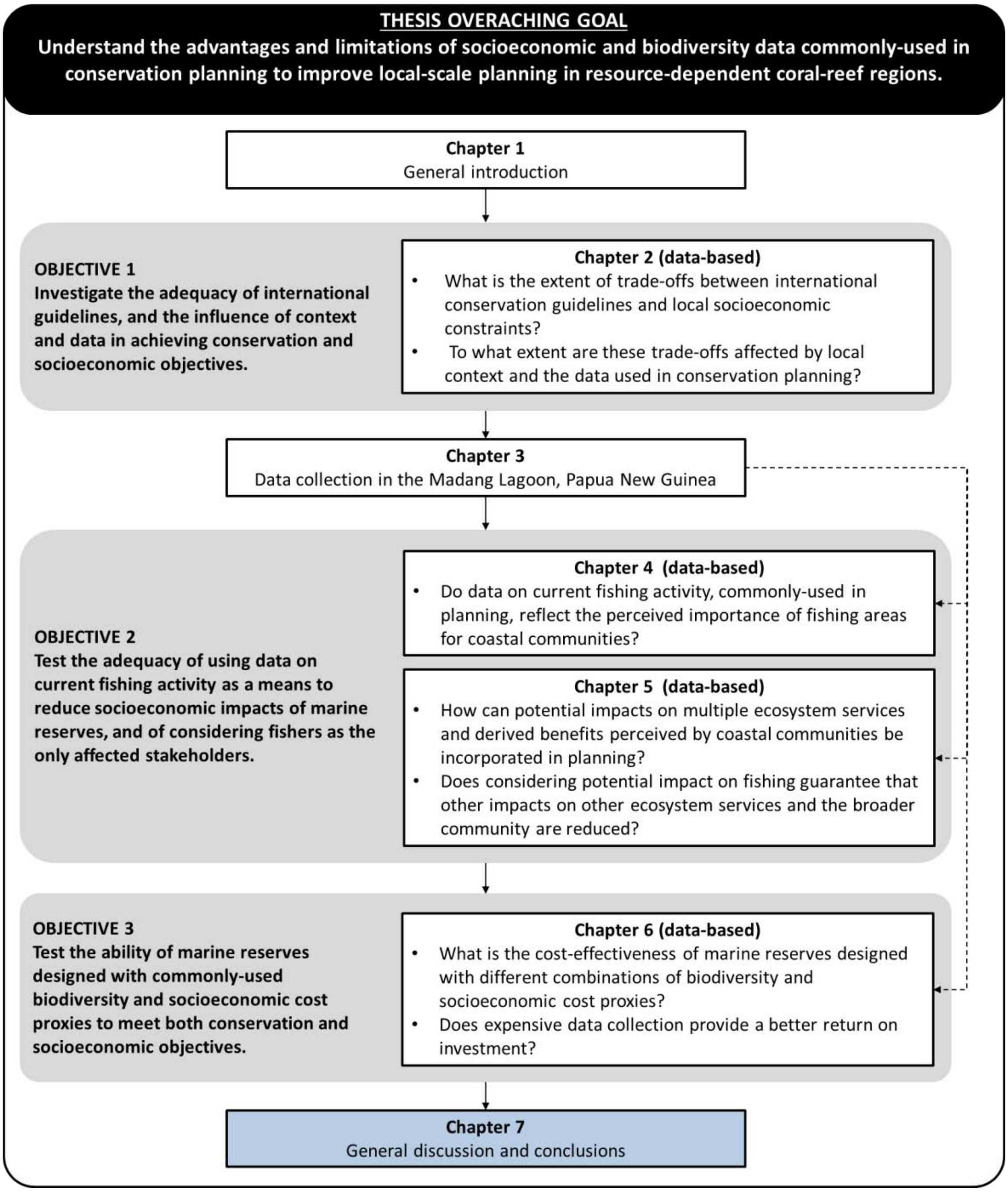


Figure 6.5. Conceptual diagram showing uncertainty in conservation outcomes in relation to the type of data used in spatial planning.

Chapter 7.
General discussion

In this chapter, I conclude with a general discussion on my findings, their limitations, management recommendations, and identify opportunities for future work.


THESIS OVERACHING GOAL
Understand the advantages and limitations of socioeconomic and biodiversity data commonly-used in conservation planning to improve local-scale planning in resource-dependent coral-reef regions.
Chapter 1
General introduction
OBJECTIVE 1
Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.
Chapter 2 (data-based)
• What is the extent of trade-offs between international conservation guidelines and local socioeconomic constraints?
• To what extent are these trade-offs affected by local context and the data used in conservation planning?
Chapter 3
Data collection in the Madang Lagoon, Papua New Guinea
OBJECTIVE 2
Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders.
Chapter 4 (data-based)
• Do data on current fishing activity, commonly-used in planning, reflect the perceived importance of fishing areas for coastal communities?
Chapter 5 (data-based)
• How can potential impacts on multiple ecosystem services and derived benefits perceived by coastal communities be incorporated in planning?
• Does considering potential impact on fishing guarantee that other impacts on other ecosystem services and the broader community are reduced?
OBJECTIVE 3
Test the ability of marine reserves designed with commonly-used biodiversity and socioeconomic cost proxies to meet both conservation and socioeconomic objectives.
Chapter 6 (data-based)
• What is the cost-effectiveness of marine reserves designed with different combinations of biodiversity and socioeconomic cost proxies?
• Does expensive data collection provide a better return on investment?
Chapter 7
General discussion and conclusions

7. General discussion

Using systematic conservation planning in coral-reef regions with high resource-dependence

Systematic conservation planning typically helps to identify candidate protected areas to represent biodiversity for a minimum cost. Conservation costs are complex and include planning and management costs, as well as potential negative impacts on stakeholders. One of the foundations for the success of protected areas designed using a systematic conservation planning approach is relevant, high-quality spatial datasets: on biodiversity and on the expected costs incurred when implementing protected areas. However, as discussed in Chapter 1, collecting comprehensive data is challenging and expensive, leading planners to use proxies of ideal information on biodiversity or costs. Another important influence on the success of such protected areas is setting relevant and adequate conservation and socioeconomic objectives.

In tropical coastal countries adjacent to coral reefs, the particular ecological, social, and political context of most countries is likely to require different approaches than common top-down conservation planning. Notably, many human communities have a close relationship with coral reefs, which contributes to their livelihoods and wellbeing. To protect coral reefs through marine reserves, it is therefore critical to adequately account for their potential impacts (or "socioeconomic cost") on people who depend on these ecosystems. However, in coral-reef regions, conservation planning is still in its infancy, leading planners to adapt approaches used in other realms, spatial scales, and socio-political contexts. These approaches used for coral reefs (and other realms) are often based on a number of untested and unstated assumptions. For example, ambitious international conservation guidelines are interpreted and applied at local scale, when they have been designed for regional planning. Another common approach to minimise the difficulty and cost of data collection is to use habitats maps to derive proxies for biodiversity. Forgone resource extraction or economic opportunities are also often used as the only proxies for socioeconomic costs of reserves to people. Because the ecological success and socioeconomic impacts of marine reserves designed using systematic conservation planning depends on the data and objectives used to design them, there is a need to understand better how common approaches can be adapted to the context of coral reefs.

Consequently, the overarching goal of my thesis was to improve our understanding of the advantages and limitations of the proxies of biodiversity and socioeconomic costs commonly used in conservation panning, with the intention to improve local-scale planning in resource-dependent coral reef regions. Specifically, I achieved this goal through three broad research objectives:

- **Objective 1:** Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.
- **Objective 2:** Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders.
- **Objective 3:** Test the ability of MPAs designed with commonly-used biodiversity and socioeconomic cost proxies to achieve both conservation and socioeconomic objectives.

I summarise the outcomes of my thesis related to each objective below.

Thesis summary and outcomes

The broad outcome of my thesis is that commonly-used conservation and socioeconomic objectives, as well as commonly-used biodiversity and socioeconomic cost proxies can lead to ineffective conservation and negative impacts on people if they are used in local coral reef contexts with high resource-dependence. In achieving my three objectives, I shed light on the advantages and limitations of these commonly-used approaches in a context of local coral-reef conservation planning. It is important to note, however, that I investigated the influence of different proxies on one of the many steps of the conservation planning process: reserve selection. Actual reserves implemented on the ground are often significantly different than initial proposed reserves selected with decision-support tools. This point is discussed in more detail in the next section.

Table 7.1 summarises the broad objectives of my thesis, the research questions developed in each data chapter to achieve objectives, and the associated key findings.

Objective 1: Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives

To achieve objective 1, I developed an approach to quantify the extent of trade-offs between conservation objectives and local socioeconomic constraints in designing marine protected areas (Chapter 2). I applied this approach to Wallis, Futuna, and Alofi islands in the central South Pacific. I designed marine reserves that aimed to represent 20% of each habitat type, following a precautionary interpretation of the CBD target for marine habitats (current at the time of writing Chapter 2) without impinging on any of the subsistence fishing grounds identified by the local environment agency. I used two types of habitat maps and two different sizes of planning units. I found that international guidelines and local constraints are incompatible and that compromises are needed on both sides. The trade-offs between conservation and socioeconomic objectives were enhanced by the small size of regions, causing substantial overlaps between fishing grounds and habitats to be represented in reserves. I also found that the difficulty in meeting both objectives increased with the spatial and thematic resolution of habitat maps, and with the size of planning units. By demonstrating that the ability to achieve conservation and socioeconomic objectives and the apparent need for trade-offs are largely data- and context-dependent, this chapter highlighted the need to question the effect on and relevance to objectives of different biological and socioeconomic data used in local-scale coral-reef conservation planning.

To investigate the advantages and limitations of biodiversity and socioeconomics data commonly-used in conservation planning in a context of local coral reef planning, as highlighted in Chapter 2, I chose another study area, the Madang Lagoon in Papua New Guinea, for which I had the opportunity to collect the necessary multidisciplinary dataset. Chapter 3 detailed the data collection process in the Madang Lagoon. Four main spatial datasets were created: data on current fishing activity (proxies of socioeconomic costs), data on the perceived importance of places for the ecosystem services and benefits people derive (reference socioeconomic cost), habitat maps of various thematic resolutions (biodiversity proxies), and species lists for macro-algae, corals, fish, and invertebrates at locations across the Lagoon (biodiversity reference data). I used these datasets in Chapter 4, Chapter 5, and Chapter 6, to address the next two objectives summarised below.

Objective 2: Test the adequacy of using data on current fishing activity as a means to reduce socioeconomic impacts of marine reserves, and of considering fishers as the only affected stakeholders

To achieve objective 2, I investigated the implications of using different socioeconomic proxies based on common assumptions about local-scale coral-reef conservation decisions.

First, in Chapter 4, I investigated the relevance of proxies of current fishing opportunities obtained with fisher surveys, as indicators of the importance of fishing places for fishers, and thus of potential socioeconomic costs. I compared maps of the importance of places for fishing based on each of these proxies, with a map of the importance of planning units for fishing as perceived by the community obtained with household surveys. I designed marine reserves that aimed for habitat protection while minimising costs to fishing, using each proxy derived from current fishing activity, and investigated the incidental reservation of places perceived as important for fishing through these scenarios. I found that proxies derived from current fishing activity do not necessarily reflect the importance of fishing grounds as perceived by fishers themselves. Consequently, designing reserves that aim at minimising costs based on such proxies can incur significant hidden socioeconomic costs by incidentally reserving places perceived as important for fishing.

Second, in Chapter 5, I tested the adequacy of using only the importance of places for fishing as a socioeconomic cost of reserve implementation. Reservation can incur a cost to the broader community by revoking harvest or even access rights in areas providing ecosystem services and derived benefits other than fishing. I developed a promising approach for systematic conservation planning to incorporate information on the perceived importance of places for local communities to access a range of ecosystem services and benefits, using participatory mapping techniques. I looked at how designing reserves that minimised cost to fishing only, as well as to each other benefit separately can incur other costs to the broader community. I found that locating reserves to minimise costs to fishing only, or to other single benefits, is likely to incur significant costs to the wider community by displacing reserves into areas where access to other benefits is important. Incorporating the range of benefits in spatial prioritisation with the proposed approach resulted in systems of reserves that were better at minimising loss of access and harvest in important places, and at equitably minimising costs related to all benefits.

Objective 3: Test the ability of MPAs designed with commonly-used biodiversity and socioeconomic cost proxies to achieve both conservation and socioeconomic objectives

In conservation planning, it is often assumed that designing reserve systems that encompass a greater diversity of habitats will incidentally protect a higher number of species. It is also assumed that finer-resolution and more complex data are more representative of "the truth", implying that representation of "true" biodiversity and reduction of "true" socioeconomic costs increases with effort in data collection. In Chapter 6, I tested these two assumptions. I designed reserves that aimed to represent a set extent of each habitat type (using four different habitat classifications described in Chapter 3 as proxies of biodiversity), while minimising socioeconomic costs (using six cost proxies derived in Chapter 4). I developed the Proxy Effectiveness Index (PEI) to measure the effectiveness of each scenario at representing species (biodiversity reference), and at minimising socioeconomic costs in terms of access to ecosystem services (socioeconomic cost reference, from Chapter 5). With information on the dollar cost of collecting each dataset, I analysed the cost-effectiveness of scenarios using all possible combinations of biodiversity proxies and socioeconomic cost proxies. I found that, overall, using too detailed or too coarse habitat maps performs poorly at representing more species in candidate reserves. Of the commonly-used proxies of fishing opportunity costs, socioeconomic costs based on data on current fishing activity performed best to reduce the reference cost when compared to coarser proxies (although they may not be the best proxies of fishing opportunities, as seen in Chapters 4 and 5). Importantly, I found that proxy effectiveness was highly variable between scenarios, and that the most expensive combinations of biodiversity and cost proxies did not necessarily provide the most cost-effective reserve systems.

Table 7.1. Key findings of my thesis.

Objectives	Chapters	Research questions	Key findings
1. Investigate the adequacy of international guidelines, and the influence of context and data in achieving conservation and socioeconomic objectives.	Chapter 2	What is the extent of trade-offs between international conservation guidelines and local socioeconomic constraints?	- New approach to measure trade-offs between conservation objectives and socioeconomic objectives, using the systematic conservation planning software Marxan. - International marine conservation guidelines appear incompatible with local socioeconomic constraints in small resource-dependent islands, and should be carefully adapted to context.
		To what extent are these trade-offs affected by local context and the data used in conservation planning?	- Aiming for the representation of habitats based on maps with fine spatial and thematic resolution may be more ecologically relevant but compromises the ability of socioeconomic and conservation objectives to both be met. - Using smaller planning units helps to achieve conservation objectives within socioeconomic constraints, but presents potential problems in managing small, dispersed areas. - Trade-offs are stronger where there is high overlap between conservation features (coral-reef habitats) and socioeconomic activities (fishing).
2. Question the adequacy of using data on current fishing activity and considering fishing as the main use of the marine environment to ensure low negative socioeconomic impacts of MPAs, and propose new approaches.	Chapter 4	Does data on current fishing activity, commonly-used in planning, reflect the perceived importance of fishing areas for coastal communities?	- Proximity to landing sites, as well as data on current fishing activity (number of fishers, total catch, and average catch per unit effort) are poor proxies for the importance of fishing grounds as perceived by fishers themselves. - Designing reserves that aim to minimise costs to fishing communities using proxies of the importance of areas for fishing such as proximity to landing sites, as well as data on current fishing activity can incur socioeconomic costs by incidentally reserving places important for fishing as perceived by local communities.
	Chapter 5	How can potential impacts on multiple ecosystem services and derived benefits perceived by coastal communities be incorporated in planning?	- Potential negative impacts of reserve implementation on the access to multiple ecosystem services and derived benefits can be incorporated in planning with a novel approach. The approach uses participatory mapping methods to measure the perceived importance, for the community, of benefits, and of the places accessed to derive these benefits.

Objectives	Chapters	Research questions	Key findings
		Does considering potential impact on fishing guarantee that other impacts on other ecosystem services and the broader community are reduced?	- Minimising only the costs of reserving fishing grounds perceived as important by the community can affect negatively the wider community's access to other important ecosystem services and benefits (e.g. recreation, aesthetics, traditional medicine…). - However, using the perceived importance of fishing grounds as a measure of the socioeconomic value of places for the wider community incurs less incidental costs to all other ecosystem benefits (e.g. recreation, aesthetics, traditional medicine…), than considering any of these benefits separately. - Aggregating all ecosystem benefits together as one cost is the scenario that allows the best compromises between conservation objectives and equitable access to all ecosystem benefits.
3. Test the ability of MPAs designed with commonly-used biodiversity and socioeconomic cost proxies to achieve both conservation and socioeconomic objectives.	Chapter 6	What is the cost-effectiveness of MPAs designed with different combinations of biodiversity and socioeconomic cost proxies?	- New approach to measure the effectiveness of reserves designed with systematic conservation planning and based on proxies of biodiversity and socioeconomic costs. - The effectiveness of MPAs at representing biodiversity when MPAs are designed with different combinations of biodiversity and socioeconomic cost proxies varies with the taxa used to measure biodiversity protection, the type of biodiversity proxy, and the type of socioeconomic proxy. - Similarly, the effectiveness of MPAs at minimising socioeconomic costs when MPAs are designed with different combinations of biodiversity and socioeconomic cost proxies varies with the type of socioeconomic proxy, and the type of biodiversity proxy. - Some biodiversity proxies or socioeconomic proxies always perform better than others. - The cost-effectiveness (effectiveness at representing biodiversity, while minimising costs, for a least dollar cost), varies with many interacting factors, including the types of proxies, and the taxa used to measure effectiveness.
		Does expensive data collection provide a better return on investment?	- Expensive combinations of biodiversity and socioeconomic cost proxies do not guarantee a better return on investment (i.e. better representation of biodiversity for the least socioeconomic cost). - Similarly, least expensive combinations can be the most effective in certain contexts.

Evaluation of the approach

Modern conservation planning and more broadly natural resource management, are clearly multidisciplinary, combining knowledge from natural and human sciences to find the best compromises between sustaining functioning ecosystems while allowing their use by people. Designing protected areas using systematic conservation planning approaches thus requires a fine understanding of social-ecological systems. However, the extent to which this knowledge can be incorporated into systematic approaches remains limited by the quantitative and spatial nature of the data required, the availability of this data, as well as by the tools at hand. These limitations must be fully acknowledged here to understand the implications of the present research.

My thesis was an academic exercise, and therefore I did not follow a typical full conservation planning process (Pressey and Bottrill, 2009). In reality, "planning is a collaborative effort between scientists, practitioners, communities, indigenous peoples, non-governmental organisations, individual landowners, large corporations, and international bodies." (Spoelder et al., 2015). Here, I focused on the influence of data and objectives on the spatial prioritisation stage of systematic conservation planning. I chose to use Marxan, the most widely-used prioritisation software, to design marine protected areas systems that help me investigate my research questions. Note, first, that that Marxan is a decision-support tool, not a decision maker (Ball et al., 2009). When systematic conservation planning is used on the ground, decisions about the best approach to spatial prioritisation, and ultimately conservation actions, should be made in consultation with all relevant stakeholders. Second, Marxan typically targets the representation of a certain amount of a conservation feature (in my thesis, 20% of each habitat type), for the least cost (in my thesis, the socioeconomic cost to resource users). These are, of course, simplistic objectives (Devillers et al., 2014) that do not account for the complexity of conservation planning on the ground. Designing effective marine reserves would ideally require the integration of more comprehensive ecological (e.g. connectivity, larval spillover) (Sale et al., 2005) and socioeconomic information, including non-spatial factors (e.g. governance and political context) (Lundquist and Granek, 2005). Third, although Marxan provides planners with critical information necessary to design their conservation plans, like all decision-support tools, the software has its own limitations. Limitations such as the choice of a target-based approach (Carwardine et al., 2009), the aggregation of multiple costs in one single variable (Adams et al., 2011), and the incorporation of socioeconomic factors as a constraint (Gurney et al., 2015) for example, are current frontiers of the rapidly-advancing field of conservation planning, and must be considered in interpreting my findings.

Additionally, throughout my thesis, I only designed one type of reserve with one type of restriction at a time (i.e. no access and no-take as opposed to multi-use reserves), leading to very conservative systems of protected areas. Recent advances in the field have seen the development of spatial prioritisation algorithms that allow the consideration of multiple-zones (Watts et al., 2009), which would be better at accommodating multiple uses of coral reef environments for example. The deliberate choice of using no access and no-take reserves in my analyses intended to: 1) illustrate two extreme conservation measures, providing complementary and contrasting results; and 2) reflect widely-used tools which effectively protect biodiversity (Juffe-Bignoli et al., 2014).

Acknowledging these limitations, I used the most widely-used approaches and tools to address key gaps in the field of systematic conservation planning, which makes my research relevant to a broad range of contexts.

Implications for conservation planning practice

In my thesis, I explored the adequacy of current approaches to systematic conservation planning to design marine reserves that aim to protect coral reef biodiversity while minimising impacts on local resource-dependent communities. As discussed in Chapter 1, for protected areas designed with systematic conservation planning to be successful, the robustness of three key foundations must be verified: 1. the ability to achieve conservation and socioeconomic objectives; 2. the adequacy of data on biodiversity; and 3. the adequacy of data on the socioeconomic context. However, these important foundations are rarely verified in practice and systematic conservation planning is thus often used based on a number of (sometimes unstated) assumptions:

- Assumption 1: International guidelines are adequate to all contexts.
- Assumption 2: The importance of areas for resource users (thus, the potential impact of restricting resource use in these areas) is well reflected by information on the intensity of their use.
- Assumption 3: Only considering the main socioeconomic use of resources in planning is enough to minimise impacts on people.
- Assumption 4: A good proxy of biodiversity or of socioeconomic cost is good in all contexts.
- Assumption 5: More expensive proxies produce more effective reserves.

Assumption 1 relates to the ability to achieve conservation and socioeconomic objectives. Assumptions 2 and 3 relate to the adequacy of data on the socioeconomic context. Assumptions 4 and 5 relate to the ability to achieve conservation and socioeconomic objectives, as well as the adequacy of data on biodiversity and on the socioeconomic context. If these common assumptions are not verified, planners face a number of risks. Risks include failure to actually protect biodiversity, and failure to reduce impacts of conservation actions on human communities, completed with a false sense of achievement of these objectives.

In my thesis, I tested these assumptions in the context of local-scale coral-reef conservation planning. My results have implications for biodiversity, human communities, scientists and managers in this context, but also more broadly for the discipline of conservation planning in general. Most of the theoretical implications of my findings for the discipline have already been mentioned in the relevant chapters. Therefore, in the following, I chose to discuss the implications of my key findings for conservation planning practice, and to propose recommendations for planners and practitioners. Opportunities for further research are outlined further. The titles of the following subsections refer to my findings regarding the above-mentioned assumptions.

International conservation guidelines are not adequate to all contexts

My research in Chapter 2 showed that assumption 1 is not verified in the context of small Pacific islands relying heavily on resources. One major underlying reason why ambitious guidelines such as the CBD targets are not adequate to all contexts is that international guidelines tend to be applied without flexibility as demonstrated in Chapter 2. International guidelines such as CBD, by definition, aim to guide actions to achieve a regional objective, which means planners must adapt these guidelines to their own context when setting local objectives (Lundquist and Granek, 2005, Carwardine et al., 2009). To my knowledge, however, there is no framework to guide governments or local organisations to do so. Reflecting on my findings, I derived a broad framework of recommendations for setting and adapting conservation and socioeconomic objectives in different contexts. This framework reflects the current shift in approaches to conservation planning which aims for more flexible and adaptable objectives based on longer-term achievement (Spoelder et al., 2015), and can be integrated with more adaptive conservation planning approaches (Pressey et al., 2013). The framework also follows the current trend considering socioeconomic factors as objectives, rather than as a constraint, along with conservation objectives (e.g. Gurney et al., 2015).

Conservation & socioeconomic objectives, especially in small islands with high resource-dependence, must be flexible, relevant, adapted, measurable, and explicit (FRAME). Conservation objectives, as well as socioeconomic objectives must be "flexible", or negotiable on both sides (as opposed to only modifying socioeconomic objectives to meet conservation objectives). These negotiations should be done in consultation with relevant stakeholders for conservation (e.g. scientists, NGOs), and for human uses (e.g. communities, resource user groups). Trade-off analyses such as the one demonstrated in Chapter 2 could be used to discuss compromises. Objectives must be "relevant" ecologically and socioeconomically. For example, does protecting 20% of each habitat type suffice to ensure the sustainability of the ecosystem in this context, and at this extent? Will avoiding reservation of all fishing grounds ensure reduced impacts on local communities, or ensure compliance, as well as protect biodiversity from threats (such as fishing)? These questions can only be answered with a thorough understanding of the social-ecological system. Objectives must be "adapted" to the local context and feasible. In small regions where most of the space is used for subsistence fishing, protecting a large portion of habitats while not impinging on fishing will not be feasible for example. Are resource users able to use alternative livelihoods? Is there capacity for enforcement? Is the objective achievable in the long-term with limited external resources? Understanding the social, economic and politic context will be critical to answer these questions. Objectives must be "measurable". In other words, conservation, as well as socioeconomic objectives must be quantitative, not qualitative to incorporate both. However, as I have shown, this can include objective and subjective data that are incorporated through indicators. Finally, objectives must be "explicit". This means they should include rationales based on evidence (e.g. scientific literature, scientific study) to explain why objectives were chosen.

The importance of areas for resource users (thus, the potential impact of restricting resource use in these areas) is not well reflected by information on the intensity of their use.

Fishers knowledge is increasingly considered in the fisheries sciences as a source of local spatial information (Martin, 2004, McCluskey and Lewison, 2008) and spatial patterns of fishing effort in particular (Léopold et al., 2014). Similarly, fishers' perceptions are increasingly used to evaluate the success of management measures (e.g. Leleu et al., 2012). However no study has incorporated fishers' subjective value of fishing grounds in conservation planning. My research in Chapter 4 showed that assumption 2 is not verified in the Madang Lagoon. Data on current fishing activity such as effort, total catch or catch per unit effort, as typically collected in conservation planning (e.g. short term, limited) provides a different picture of the importance of fishing places for fishers, compared to their own perceptions of important fishing grounds. My findings have serious implications for management. Considering only data on fishing activity - common in socioeconomic assessments for conservation planning (Adams et al., 2011) - could fail to address the broader spectrum of factors determining the fishing importance of planning units for local communities, as exemplified by perception data. This failure would risk poor social acceptance of and low compliance with conservation actions, reducing their effectiveness.

The discrepancies between scientific information such as empirical data on fishing catch and effort and fishers' knowledge have been recently investigated (Daw et al., 2011b). These differences, if not well understood, can prevent collective action to sustain resources. However, such disputes may also provide opportunities to expand the scope of knowledge available for resource management. There are several ways in which my findings could be addressed in practice, in this regard. First, "scientific" socioeconomic assessments should be more comprehensive. For example, data should be collected on the long-term to ensure they reflect appropriately the potential temporal variations. Data collection should also consist of larger representative samples to account for individual and other group variations (e.g. other communities, different genders, gear users). However, limitations of time and budget are common in conservation planning and data collection is often done in a rapid assessment fashion. This emphasizes the need to understand well the limitations of such data if planners were to use them. Second, data on current fishing activity and perceptions could be used in combination. However, more research would be needed to find the best approaches to do so in a systematic way. Alternative approaches would consist of consulting with local communities about the different datasets and their relevance to achieve socioeconomic objectives.

Only considering the main socioeconomic use of resources in planning is not enough to minimise impacts of reserves on people.

In marine reserve design, socioeconomic costs tend to be considered for fishers only (Adams et al., 2011), but recent approaches began to incorporate other users in the process (e.g. Klein et al., 2008, Giakoumi et al., 2011, Rojas-Nazar et al., 2012). My research in Chapter 5 showed that assumption 3 is not verified in the Madang Lagoon. Community members use, enjoy, and value the reef environment in various ways, other than through fishing. With the hypothesis that reserving an area valued for the ecosystem services it provides will incur a cost, I proposed an approach to incorporate information on the perceived importance of areas for different services and benefits into conservation planning. Using this approach, I showed that only minimising impacts in terms of fishing can potentially affect the broader community by restricting their use of areas important for other services.

In practice, my findings suggest that planners should consider and engage the broader community in planning, not just main resource users. This could have several advantages for conservation planning on the ground, for local coral-reef planning as well as in other contexts. Indeed, incorporating people's perceptions of important places, for example by using my approach, could help promote the support of communities for conservation by ensuring preferences are accounted for and by engaging community members in planning (Thornton and Scheer, 2012).

The effectiveness of proxies of biodiversity or of socioeconomic cost is context-dependent.

My research in Chapter 6 showed that assumption 4 is not verified in the Madang Lagoon. The effectiveness of proxies of biodiversity or of socioeconomic cost depends on many factors which will vary in different contexts. More importantly, this applies to the most commonly used proxies of biodiversity and socioeconomic costs in marine reserve design. The implications for conservation practice are high since using the wrong proxies can lead to failure of conservation actions, as well as high impacts on affected human communities. This is particularly critical in coral-reef regions, where coral reefs require urgent and robust protection and people are strongly connected to their environment on which their livelihoods and wellbeing depend. Since my findings are in accordance with other studies investigating the effectiveness of proxies of biodiversity (Sarkar and Margules, 2002, Caro and Girling, 2010, Grantham et al., 2010, Lewandowski et al., 2010, Mellin et al., 2011, Andréfouët et al., 2012a), my recommendation for practitioners would be to test proxies in a similar context to theirs, using the same systematic conservation planning objectives and software. In cases where testing is not possible, a thorough literature review would be best. By default, limitations associated with the proxies used should be explicitly acknowledged, and a risk assessment (following the example shown in **Figure 6.5**, Chapter 6) should be undertaken to avoid false expectations and perverse outcomes.

More expensive proxies do not necessarily produce more effective reserves

Evaluating the cost-effectiveness of conservation actions is critical in a climate of limited funding for environmental management (e.g. Grantham et al., 2008, Laycock et al., 2009). This is especially true in developing coral-reef countries were resources for conservation are scarce (Bottrill, 2011), biodiversity is severely at risk (Bellwood et al., 2004), and human communities depend strongly on resources for their livelihoods and well-being (Salvat, 1992, Moberg and Folke, 1999). My research in Chapter 6 showed that assumption 5 is not verified for the Madang Lagoon. I measured the effectiveness of biodiversity and socioeconomic assessments at producing reserves that represent biodiversity while minimising impact on local people, against the cost of conducting these assessments. It is generally assumed that gathering more data is a good investment for conservation planning, but return-on-investment (ROI) studies focusing on data collection costs in relation to the effectiveness of datasets to help achieve conservation objectives are lacking (Grantham et al., 2008, Boyd, 2012). Although the availability of data has increased substantially with the development of public biodiversity databases, data collection costs remain a large component of overall conservation costs, especially in data-poor regions. My approach was the first, to my knowledge, to investigate cost-effectiveness of (or return on investment on) data collection for marine conservation. I found that the most comprehensive (expensive) assessments did not necessarily perform better than others, because many interacting factors (described in the precedent section) influence effectiveness, thus cost-effectiveness of proxy combinations. This does not contradict my recommendation to undertake more comprehensive biodiversity and socioeconomic assessments (see other chapters). Rather, it emphasizes the need for planners to assess the relevance of proxies to achieving their conservation, social and economic objectives, rather than making important conservation decisions based on untested assumptions This could be achieved, ideally through testing the effectiveness of their data at achieving a given objective, or at least trying to formulate explicitly the rationales behind their choice of data and supporting them by scientific literature.

In conservation, return-on-investment is a relatively recent concept (Murdoch et al., 2007, Boyd, 2012), therefore, applications including the one presented in this thesis are often simplistic and contain obvious caveats. For example, it is important to note that the results of my analysis do not constitute generalizable recommendations, as the expenditures related to data collection and the effectiveness of proxies were specific to the context of the Madang Lagoon. Additionally, I conducted my return-on-investment analysis for a specific (simplistic) objective: representation of species while minimising the loss of access to important places for the ecosystem services and benefits they provide, as perceived by local communities. However, in practice, the methodology has potential to be replicated in any other context and with any other objectives. My approach not only allows evaluating which assessment(s) provide the best and worst trade-offs between conservation and socioeconomic objectives, but also shows the relative cost of these assessments.

Opportunities for future research

In future research work, I will be addressing several of the limitations and gaps identified throughout my thesis, which I outline below.

Effectiveness of surrogates for designing protected areas

In my thesis, I showed that results of published tests of surrogacy effectiveness are conflicting because many factors influence the effectiveness of reserves designed based on surrogates to achieve conservation and socioeconomic objectives (Sarkar and Margules, 2002, Caro and Girling, 2010, Grantham et al., 2010, Lewandowski et al., 2010, Mellin et al., 2011, Andréfouët et al., 2012a). Factors influencing the apparent effectiveness of proxies include: the type of proxy and the metrics (also termed 'indicators') used to measure it; the reference data on the feature of primary interest (for testing the proxy) as well as the metrics used to measure it; biogeographic and socioeconomic aspects of the study region; the spatial extent of the study region; data resolution; and the method used to measure effectiveness. The studies that identified these factors were often limited to a particular realm or a particular set of proxies as seen throughout the thesis. A large meta-analysis (or at least a systematic literature review) across realms and contexts is needed to investigate the different proxies used in conservation planning (theoretical vs. real-world), the rationales to use them, as well as evidence from the scientific literature if applicable. The review would consist in listing all elements of context that could potentially influence effectiveness (e.g. spatial extent of planning region, planning units, data type and resolution for proxy and reference, methods used to test effectiveness, initial objective), as well as key finding on effectiveness. A clear typology of proxies would be needed for this work. This review would allow a better understanding of how these factors interact, help identify research priorities for the future, and allow developing a set of recommendations regarding proxy testing.

Additionally, my thesis investigated only the influence of different proxies on spatial prioritization outputs, one of the many stages of a comprehensive conservation planning process. However, decision support tools such as Marxan are typically used to create “starting points” to engage stakeholders in the negotiation process (Pressey et al., 2013) but it would be highly unusual for spatial prioritization outputs to be directly implemented on the ground. An interesting research question would be to compare implemented reserve systems arising from spatial prioritization outputs such as those proposed in this thesis (i.e. based on several combinations of proxies). One way to do this would be to consult with several separate groups of experienced conservation planners who understand well the context of a given planning region to simulate a typical negotiation process that would convert initial designs into “applied” actions.. This could be done using my data and planning outputs for the Madang Lagoon. I could measure indicators of effectiveness of reserves like the ones in my thesis such as the amount of biodiversity protected in the final “applied” reserves, and the total socioeconomic cost incurred to the community of interest. I could also attempt to identify the factors in the decision-making process that drive changes in spatial outcomes following the typology of “reasons for changes in designs” in Pressey et al. (2013).

Can coral-reef habitats be used as a cost-effective proxy of coral-reef biodiversity?

In my thesis, I showed that creating habitat maps, increasingly used as a proxy of coral-reef biodiversity, is considerably less expensive than collecting sparse but comprehensive species data. In shallow coastal coral-reef environments, habitat mapping is mostly done via satellite imagery. Satellite imagery has several advantages for this application: it operates very well at depths shallower than 45 m (most prolific tropical coral reefs typically occur above 30 m depth); it provides exhaustive coverage of the area of study and can cover large areas; it is flexible, with data available for different extents of study areas and at different resolutions (cost increases with extent and resolution); and its acquisition is highly economical per unit area compared to extensive field surveys (see Green et al. (1996) and

Mumby et al. (1999) for the last reviews on the subject, which now require updating). The use of satellite imagery along with other remote sensing applications for coral-reef habitat mapping is now well studied and techniques are well established, thus making coral reef habitat mapping a well-established field that offers many possibilities in terms of products. Creating a habitat map through satellite imagery typically involves the following processes: 1/ definition of the study objectives and choice of imagery (sensor, spectral, and spatial resolution); 2/acquisition of remote sensing data; 3/ data processing and classification (supervised, unsupervised or manual, with or without integration of field data, local research, expert or community knowledge.); and 4/ ground-truthing and accuracy assessment.

In coral-reef countries, where there is a global lack of funds for conservation, planners see coral satellite imagery as a versatile cost-effective tool to collect coral reef habitat data, and thus potential information on biodiversity. Coral-reef habitat maps are now created almost routinely, and then used to select new priority areas, assuming they will adequately represent coral-reef biodiversity (rationales in Dalleau et al. (2010) and Wabnitz et al. (2010)). But habitat data used by conservation planners are habitat maps that correspond to a particular classification itself built from a particular image with inherent parameters; the maps are supposed to represent real habitats, which are themselves believed to be potential proxies of biodiversity. There are thus a number of uncertainties and unanswered questions to be addressed before claiming that habitat maps are, or are not, a valid cost-effective tool for conservation planning to protect coral-reef biodiversity.

To support the use of coral-reef habitat maps as surrogates of the biodiversity they support in conservation plans and avoid misleading recommendations for real-world applications, future research needs to build confidence that these plans will benefit the environment, help lower opportunity costs to local communities, and lower data acquisition costs. I identified the following needs. A clear framework should be developed to help decisions on coral reef habitat classification and mapping, taking account of the various parameters involved (e.g. purpose of the study, approach to habitat classification, extent of the study area, resolution and accuracy of maps). Research is needed to identify the costs of common ways of mapping coral reef habitats and understand the effectiveness of these maps in representing biodiversity. Research is also needed to identify what drives the effectiveness of coral reef habitats as selection-based surrogates for biodiversity.

How are perceptions on ecosystem services influenced by reserve design?

Ecosystem services valuation is a growing field which receives increasing attention from economists, as well as conservation scientists and practitioners (Lele et al., 2013). However, including non-economic values of areas and conservation features in decision-making is a current frontier of the field. My attempt (in Chapters 4 and 5) was the first of its kind within the field of systematic conservation planning. Such approaches are important to develop because understanding the spatial patterns of social values and biodiversity values of areas can provide critical insight into identifying areas of conflict or synergy between value sets (Whitehead et al., 2014). My approach should be seen as a stepping stone towards a more comprehensive one, which could be reached by investigating the following research gaps. In my approach, I assumed no spatiotemporal variations in the perceptions of the importance of areas for ecosystem services and benefits they provide (or social values of benefits provided by these areas), and a homogeneous group of individuals. However, variations in perceptions, how and where ecosystem services are delivered, as well as who benefits from these services are likely (Daw et al., 2011a). A second limit of my approach is that I assumed a negative impact of reserves on access to and perceptions of ecosystem benefits. Further research should investigate the range of possible effects of reserve implementation on ecosystem services and benefits. Impact can be positive or negative, reserves can create, remove or displace benefits, which will ultimately affect how people perceive the importance of areas for these benefits. Finally, I quantified subjective values and weighted each benefit by the number of people valuing it. Different approaches to quantify subjective values and to measure the relative perceived importance of ecosystem services in relation to each other could be investigated, for instance by reviewing relevant social sciences or ecosystem service valuation literature.

Concluding remarks

My thesis contributes to the theory and practice of systematic conservation planning by verifying common assumptions about biodiversity and socioeconomic proxies, as well as widely-used objectives in many academic and applied conservation exercises. In particular, my research filled critical gaps for local conservation planning in coral-reef regions, highlighting the inadequacy of common approaches used in other contexts to be replicated in these regions. My work also demonstrates the essential role of objectives and data in driving important conservation decisions, accentuating the critical need for conservation planners (scientists and practitioners) to carefully consider their relevance to the social, ecological, and political context of their planning region. By systematically considering the limitations of their data, engaging with and including a broader range of stakeholders in the planning process, and tailoring objectives, for example using some of the new approaches developed throughout my research, planners will improve the cost-effectiveness of protected areas designed with systematic conservation planning approaches.

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Appendices

Appendix 1. Habitat classifications for the Madang Lagoon.

	Geomorphology		Substrate	Benthos		
G2	G4	G5	S	B	G5SB[1]	Extent (ha)
continental fringing	diffuse fringing	764	soft substrate	bare substrate	1	0.91
	enclosed lagoon or basin	757	mixed substrate	bare substrate	2	2.42
			mixed substrate	colonised substrate	3	1.69
			mixed substrate	coral	4	0.92
			soft substrate	bare substrate	5	31.84
			soft substrate	colonised substrate	6	0.02
	forereef	758	hard substrate	bare substrate	7	0.04
			hard substrate	coral	8	16.30
			hard substrate	seagrass	9	0.03
			mixed substrate	bare substrate	10	0.73
			mixed substrate	colonised substrate	11	46.92
			mixed substrate	coral	12	62.14
			mixed substrate	cyano?	13	0.29
			mixed substrate	seagrass	14	0.36
			soft substrate	bare substrate	15	3.18
			soft substrate	colonised substrate	16	0.22
			soft substrate	coral	17	4.05
			soft substrate	seagrass	18	0.16
	reef flat	760	hard substrate	algae	19	0.04
			hard substrate	colonised substrate	21	0.46

[1] G5SB is a unique class which combines all geomorphology classes (corresponding to G5), the substrate class (S), and the benthos class (B).

Appendix 1. Habitat classifications for the Madang Lagoon.

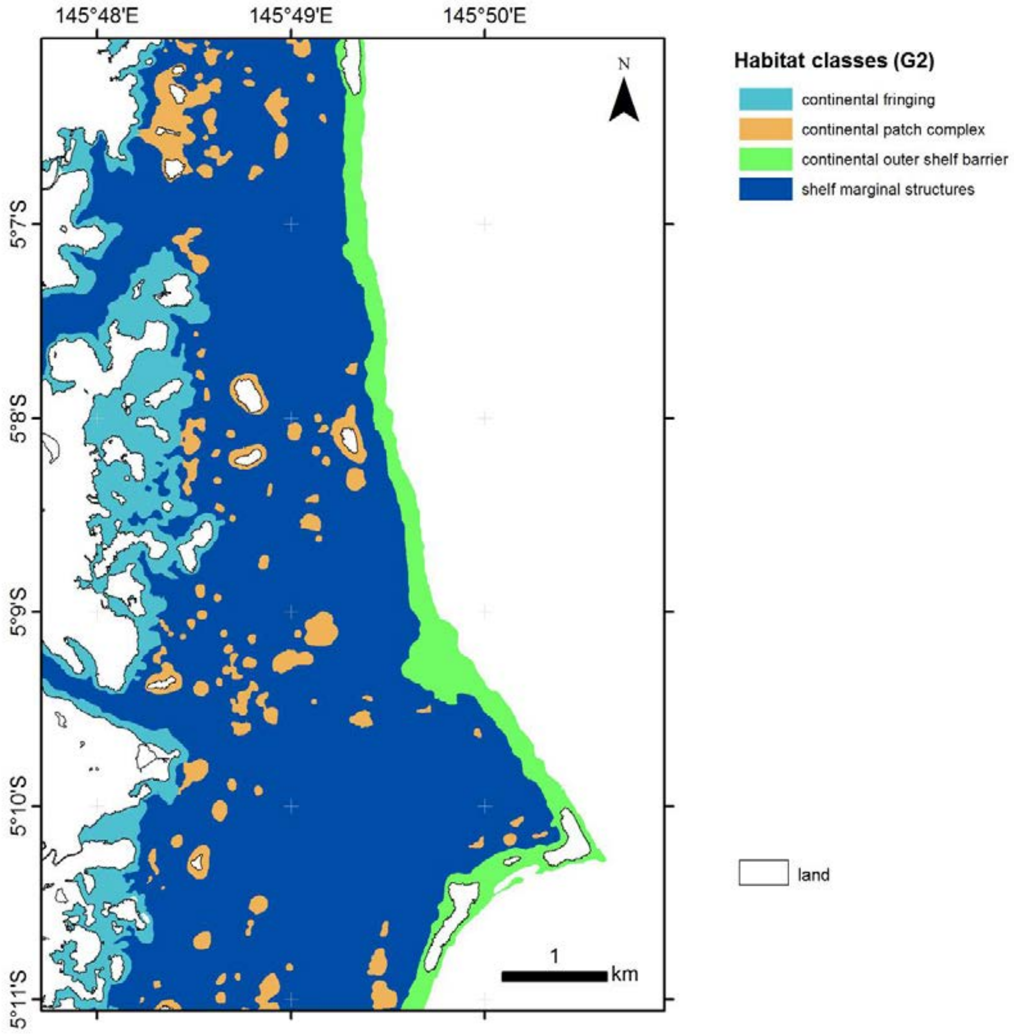
Geomorphology			Substrate	Benthos		
G2	G4	G5	S	B	G5SB[1]	Extent (ha)
continental fringing	reef flat	760	hard substrate	coral	22	39.93
			hard substrate	seagrass	23	0.03
			mixed substrate	algae	24	1.19
			mixed substrate	bare substrate	25	2.39
			mixed substrate	colonised substrate	26	8.39
			mixed substrate	coral	27	8.39
			mixed substrate	cyano?	28	0.22
			mixed substrate	seagrass	29	0.33
			soft substrate	algae	30	0.37
			soft substrate	bare substrate	31	14.15
			soft substrate	colonised substrate	32	1.90
			soft substrate	coral	33	0.94
			soft substrate	seagrass	34	9.80
	river mouth	1002	soft substrate	bare substrate	35	11.38
			soft substrate	seagrass	36	0.19
	shallow terrace	762	mixed substrate	colonised substrate	37	0.60
			mixed substrate	coral	38	0.08
			soft substrate	bare substrate	39	8.22
			soft substrate	colonised substrate	40	2.77
			soft substrate	seagrass	42	0.05
	shallow terrace with constructions	900	mixed substrate	bare substrate	44	0.17
			mixed substrate	colonised substrate	45	3.20
continental fringing	shallow terrace with constructions	900	mixed substrate	coral	46	0.56
			soft substrate	bare substrate	47	0.31
			soft substrate	colonised substrate	48	0.39
			mixed substrate	coral	49	0.60

Geomorphology			Substrate	Benthos		
G2	G4	G5	S	B	G5SB[1]	Extent (ha)
continental outer shelf barrier	deep terrace	661	mixed substrate	bare substrate	66	1.59
			mixed substrate	colonised substrate	67	0.18
			soft substrate	bare substrate	68	9.82
	forereef	666	hard substrate	bare substrate	74	0.08
			hard substrate	colonised substrate	75	7.94
			hard substrate	coral	76	33.53
			mixed substrate	bare substrate	77	1.25
			mixed substrate	colonised substrate	78	23.47
			mixed substrate	coral	79	9.87
			soft substrate	bare substrate	80	6.36
			soft substrate	colonised substrate	81	0.32
			soft substrate	coral	82	0.39
	pass	668	mixed substrate	colonised substrate	83	18.53
	reef flat	670	hard substrate	bare substrate	86	0.01
			hard substrate	colonised substrate	87	0.51
			hard substrate	coral	88	7.09
			mixed substrate	bare substrate	89	0.61
			mixed substrate	colonised substrate	90	2.34
continental outer shelf barrier	reef flat	670	mixed substrate	coral	91	0.55
			soft substrate	bare substrate	92	1.31
			soft substrate	colonised substrate	93	0.22
	shallow terrace	671	mixed substrate	bare substrate	96	2.73
			soft substrate	bare substrate	97	6.18
			soft substrate	colonised substrate	98	0.31
	shallow terrace with constructions	672	mixed substrate	bare substrate	99	0.20
	subtidal reef flat	673	hard substrate	colonised substrate	102	0.99

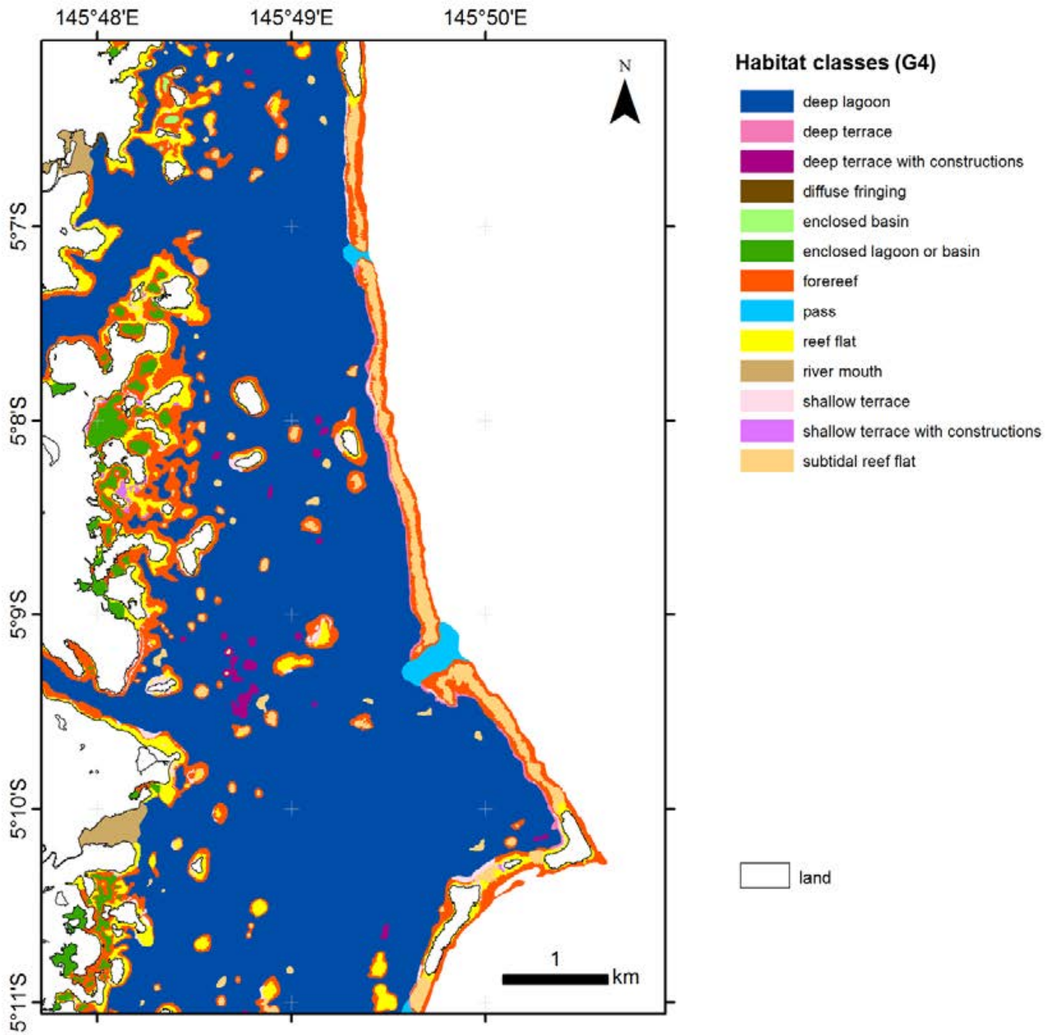
Geomorphology			Substrate	Benthos		
G2	G4	G5	S	B	G5SB[1]	Extent (ha)
			hard substrate	coral	103	10.76
			mixed substrate	bare substrate	104	0.34
			mixed substrate	colonised substrate	105	48.09
			mixed substrate	coral	106	4.06
			soft substrate	bare substrate	107	0.15
continental patch complex	deep terrace with constructions	538	mixed substrate	coral	108	13.16
	enclosed basin	539	soft substrate	bare substrate	109	1.61
	forereef	545	hard substrate	colonised substrate	111	0.19
			hard substrate	coral	112	16.09
			mixed substrate	bare substrate	113	5.57
			mixed substrate	colonised substrate	114	15.22
			mixed substrate	coral	115	39.95
			soft substrate	bare substrate	116	3.28
			soft substrate	colonised substrate	117	0.75
continental patch complex	forereef	545	soft substrate	coral	118	0.03
	reef flat	550	hard substrate	bare substrate	119	0.20
			hard substrate	colonised substrate	120	0.24
			hard substrate	coral	121	16.90
			mixed substrate	bare substrate	122	0.28
			mixed substrate	colonised substrate	123	2.94
			mixed substrate	coral	124	4.28
			soft substrate	bare substrate	125	1.08
			soft substrate	colonised substrate	126	0.12
	shallow terrace	551	mixed substrate	colonised substrate	130	0.34
			soft substrate	bare substrate	131	7.11
			soft substrate	colonised substrate	132	2.00

Geomorphology			Substrate	Benthos		
G2	**G4**	**G5**	**S**	**B**	**G5SB[1]**	**Extent (ha)**
			soft substrate	coral	133	0.22
	subtidal reef flat	553	hard substrate	colonised substrate	134	0.07
			hard substrate	coral	135	8.31
			mixed substrate	colonised substrate	136	5.20
			mixed substrate	coral	137	14.10
			soft substrate	bare substrate	138	0.01
			soft substrate	colonised substrate	139	0.01
			soft substrate	coral	140	0.01
shelf marginal structures	deep lagoon	788	soft substrate	bare substrate	142	1,655.12

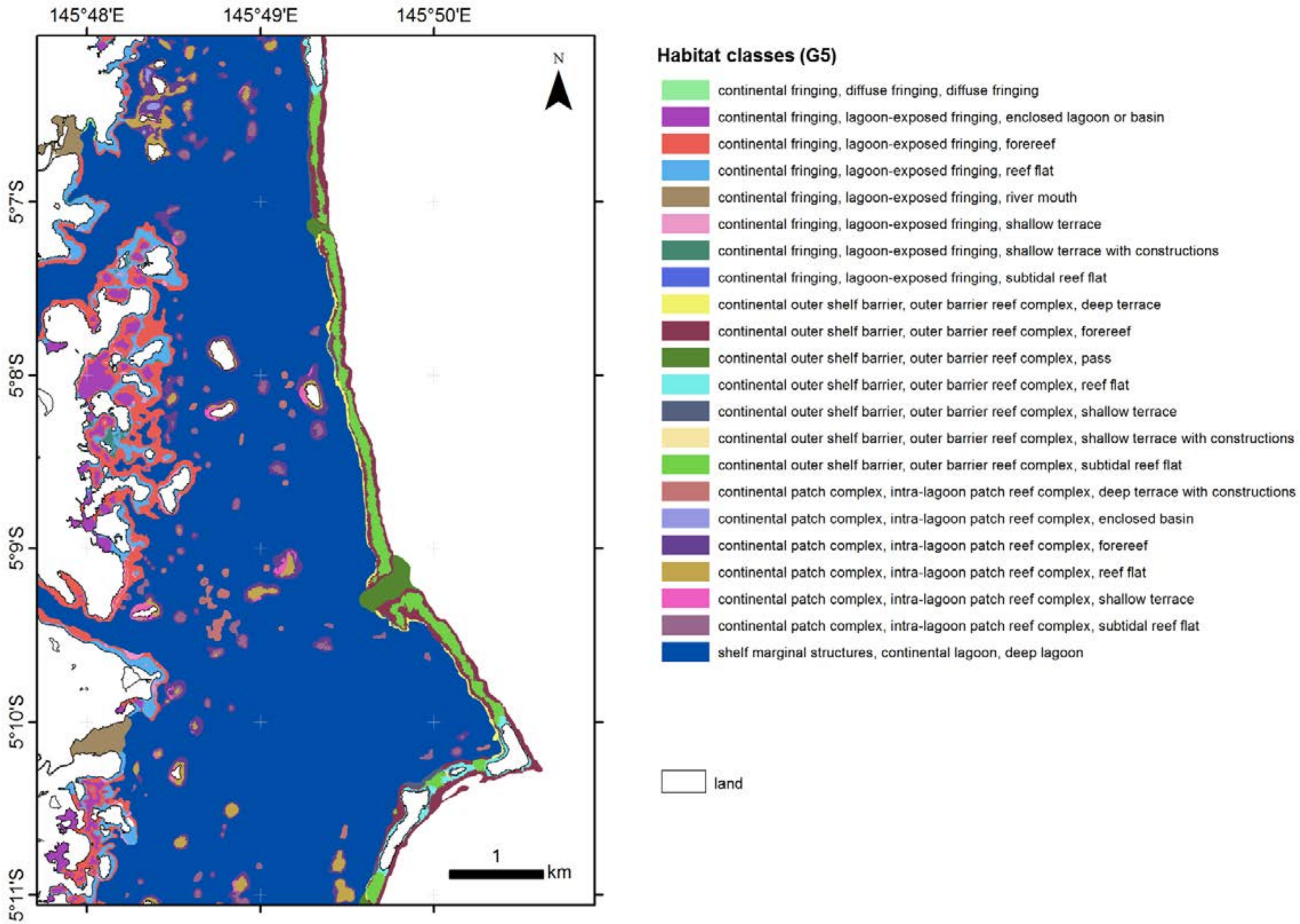
Appendix 2. Shallow marine habitat maps of the Madang Lagoon.



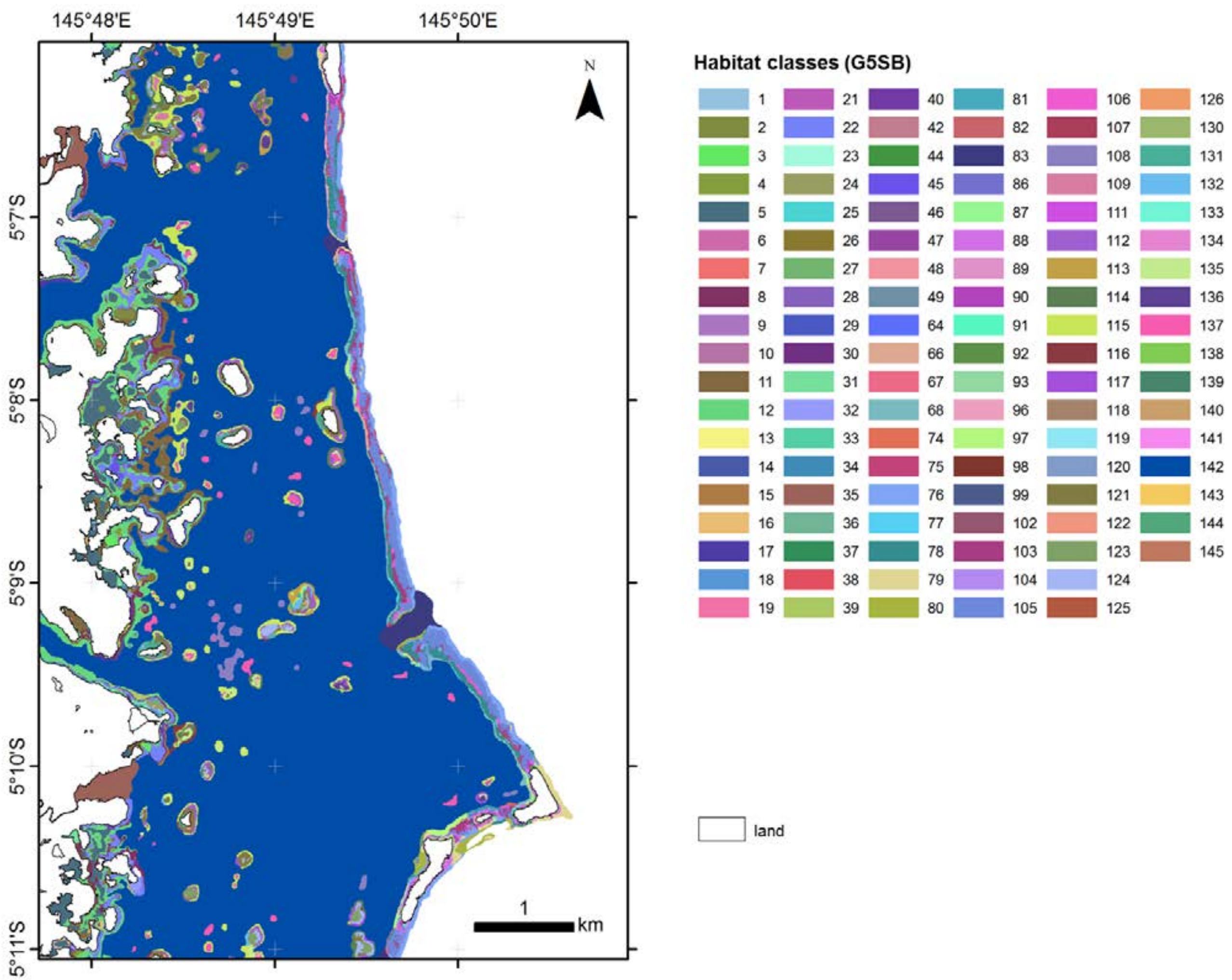
Map of geomorphologic marine habitats G2 found within the Riwo waters (indicated with a grid of planning units).



Map of geomorphologic marine habitats G4 found within the Riwo waters (indicated with a grid of planning units).



Map of geomorphologic marine habitats G5 (a combination of G2, G3 and G4) found within the Riwo waters (indicated with a grid of planning units).



Map of marine habitats G5SB (a combination of geomorphology G5, substrate stability S and benthos B) found within the Riwo waters (indicated with a grid of planning units).

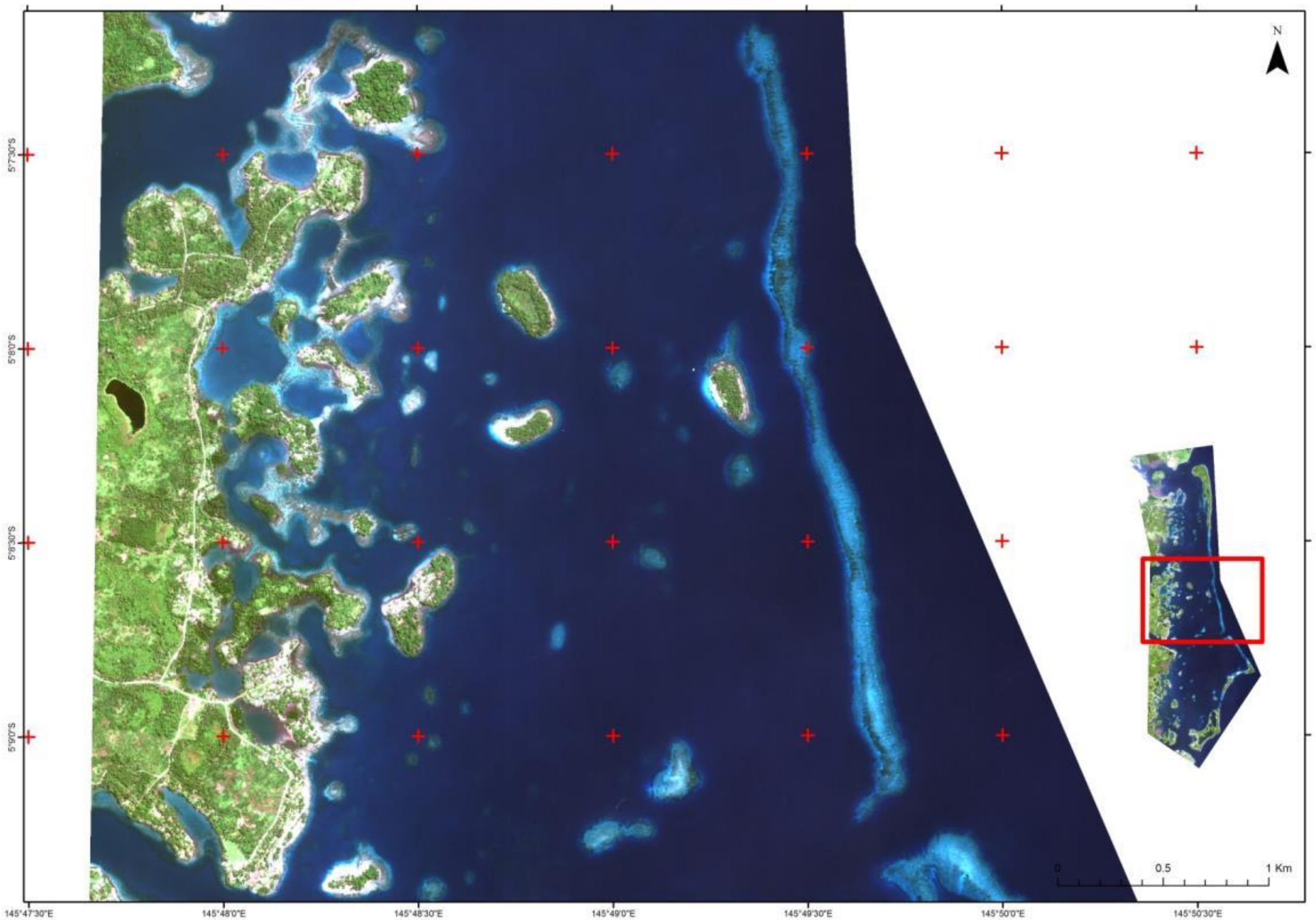
Appendix 3. Fisher survey material

Current fishing activity questionnaire

1. Date and time of landing
2. Weather (e.g. sunny, windy, dull)
3. Landing site name
4. Fishing ground identification number [to be reported on the map]
5. Fisher's name
6. Fisher's gender (male or female)
7. Transport type used for this trip (e.g. canoe, kayak, motorboat, swim, walk)
8. Primary gear used for this trip (e.g. handline, speargun, gillnet, iron rod)
9. Trip duration (hours)
10. Crew size (number of people)
11. Total catch weight (kg) [Weigh total catch.]
12. Location, shape and size of fishing spot for this trip [Show a satellite view of the region and ask fisher to draw fishing spot on the map.]

Example of satellite image used during the fisher surveys.

Seven overlapping sections of the Madang Lagoon such as the image below were printed on A3 sheets for the fisher surveysImages were overlaid with tracing paper for participatory mapping.



Appendix 4. Household survey material

Part I of the household survey: General questions on household and respondent

1. Household identification number
2. Date
3. Village
4. Name of respondent
5. Age of respondent
6. Gender of respondent
7. Name of head of household
8. Where are you originally from? (this community, other)
9. How long have you lived in this village?
10. [Ask only if not born here.] Why did you move to this village?
11. How many people live in your household, including yourself? (number of adults, children, males and females) [Define "household" if necessary.]
12. What are the activities that bring cash into your household? (e.g. fishing, gleaning, gardening, tourism, salary, selling goods/handicrafts at the markets) Please rank these activities from the one that brings most income to the one that brings least. In your household, how many people bring cash with each of these activities?

Part II of the household survey: General questions on fishing habits

1. How long have you been fishing/gleaning for?
2. When is the best fishing season/time for you to go fishing/gleaning? Why?
3. When you go fishing, what gear and transport do you use? (Gears: handline, speargun, gillnet, iron rod, gleaning, others. Transports: canoe, kayak, motorboat, swim, walk) How often do you use each gear and transport? (number of trips per week in good and bad fishing week)
4. Some days you catch a lot of fish, other days you may not catch many fish. What is your catch on a good day? (units: number of fish or octopus, big/small plates, esky). How long do you usually go out on good days? What gear do you use (primary)? How many people usually contribute to the catch on good days? How much is that worth? (monetary value) [Same questions for a poor day, then a normal day.]
5. Consider your catch on normal days [Recall previous answers if necessary.]. How much of this catch do you sell (which Market?), give away, and bring home on such days? (units: same as above or percentages)
6. Consider your normal day fishing gear and transport [Recall previous answers if necessary.].What are the 3 species you catch most often? (local name and common name)

Part III of the household survey: Fish and seafood consumption

1. How much fish or other seafood does your household eat in a normal week? (units: number of items per meal/plate/time, and number of meals/plates/times per week)
2. Where does your household usually get fish or other seafood from? (Caught myself/member of this household, given away by somebody in the family/village, bought at/from …)
3. In your household, what are the fish and other seafood you most frequently have on your table? (common name and local name. units: same as above)

Part IV of the household survey: Ecosystem benefits and services survey questionnaire

The "card game": benefits and services in the region of interest

[Show a satellite view of the region and discuss location of main landmarks with respondent to ensure common understanding of the spatial environment.]

1. How does your household benefit from the lagoon? (e.g. fishing, recreation, traditional medicine…) [Open-ended question, then probe with "benefit cards" showing/explaining each use/service if necessary ("what about this?")]
2. How important are the following benefits for your household? Please rank these benefits from the one that is the most important to your household to the one that is the least important [Use "benefit cards" for ranking]

The "drawing game" and the "tokens game": mapping places of importance for specific benefits and services for the community

Fishing benefit:

3. Where do you (or main fisher in the household) usually go fishing? What gear do you use in these fishing zones? What do you catch most often in these fishing zones? [Ask respondent to draw fishing zones, and annotate map with gear and main species caught in each fishing zone]
4. How important is each of these fishing zones for your household? [Give a fixed number of tokens and ask to distribute the tokens between fishing zones. Report number of tokens on the map next to corresponding fishing zone]
5. Which one of these fishing zones is your favourite? [Report favourite fishing zone on map.] Why is it your favourite?

Other benefits:
[Show "benefit cards" one by one, by order of decreasing importance as per Q2. Ask Q6 and Q7 and perform mapping exercise for each card at a time.]

6. Can you show me on the map the places that are accessed for this benefit by your household? [Ask respondent to draw boundaries of these places. One colour per benefit category. Report legend on paper questionnaire.]

7. How important is each of these places for your household? [Give a fixed number of tokens for each benefit of interest, and ask to distribute the tokens within all places corresponding to the benefit of interest. Report number of tokens on the map next to corresponding place]

Part V of the household survey: Current protected areas, and wanted / unwanted no-go areas.

1. Are there areas where you are not allowed to go, or only allowed under specific conditions? [Ask respondent to draw no-go areas, and number them. Report details on paper questionnaire.]

[The following two questions are extreme scenarios. Explain thoroughly the fictitious nature of these questions.]

2. Now imagine that you are given the power to choose one area that will become a no-go area. Which area (one only, any shape or size) would you like to see in a permanent no-go area within the tenure of your village? Why would you like to restrict access to this area in particular? [Ask respondent to draw wanted no-go area and number it. Report details on paper questionnaire.]
3. Now imagine that the whole region is a no-go area. This time you are given the power to choose which area will remain open to your community (one only, any shape or size). Which area would you NOT like to see in a permanent no-go area? Why would you like to keep this area in particular open? [Ask respondent to draw no-go area to be kept open, and number it. Report details on paper questionnaire.]

Benefit cards used in the household survey.

Each of the 10 cards, showing only images, was printed on an A5 sheet.

Fishing (FI)	**Perceived biological richness (RI)**	**Aesthetic enjoyment (AE)**
		
Recreation (RE)	**Traditional medicine (TM)**	**Education and knowledge sharing (ED)**
		

Lime material to chew with betel nuts (LI)

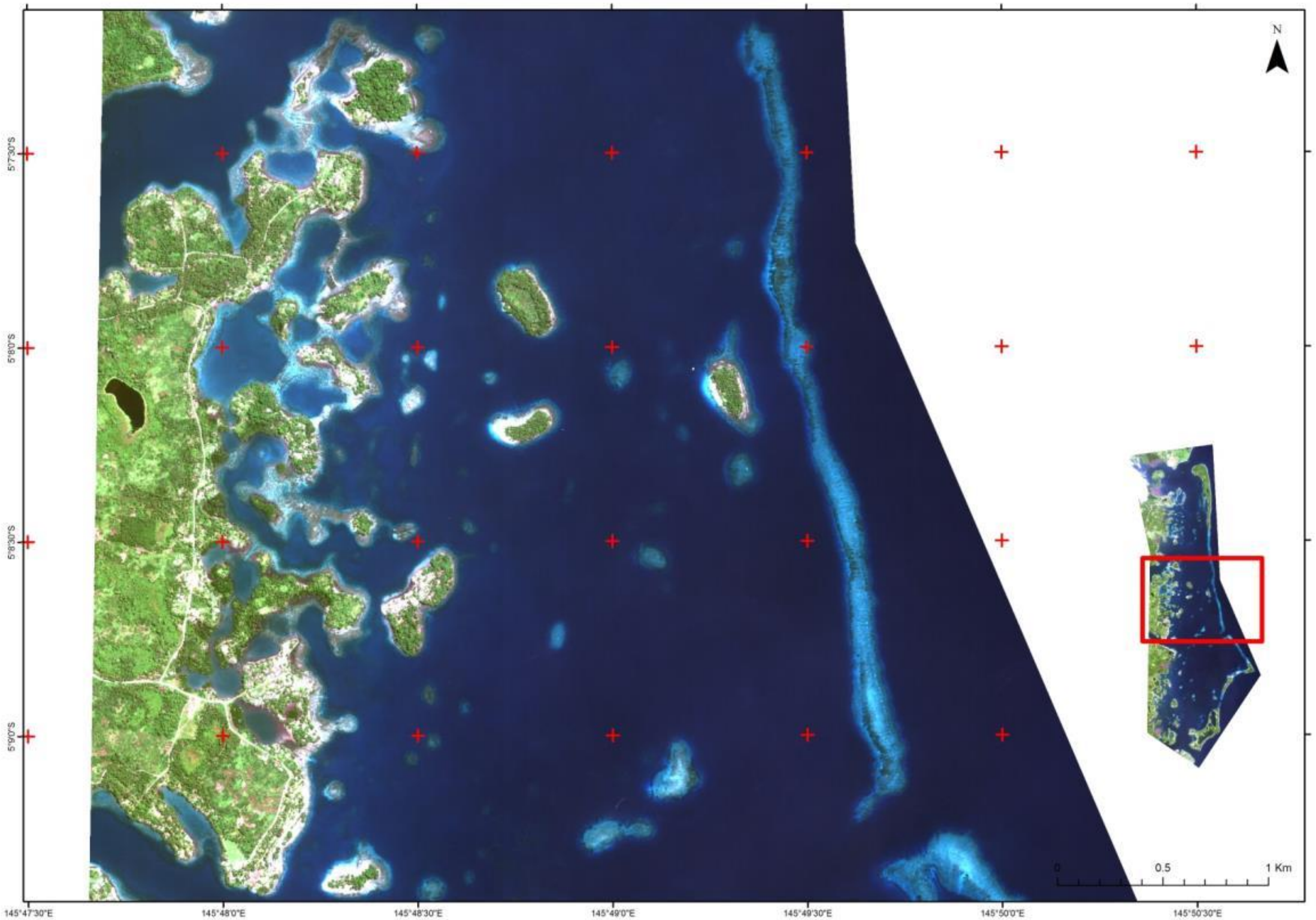
Spiritual value (SP)

Wrecks (WR)

Tourism (TO)

Example of satellite image used during the household surveys.

Four (non-overlapping) sections of the Madang Lagoon such as the image below were printed on A3 sheets for the households suverys. Images were overlaid with tracing paper for participatory mapping.



Appendix 5. Characterisation of the Riwo community's small-scale fishery in the Madang Lagoon.

From the catch surveys, the average duration of a fishing trip was 4h30 (±19min) and the average fishing crew size (number of people contributing to a catch) was 1.88 (±0.21). Average total catch per trip was 4.29 kg (±0.57) with a maximum weight at 30 kg in one trip caught by 5 fishers in 5h. Women fished mainly reef fish with handlines (79%), but also octopus and shellfish (total females surveyed n =28). Males fished mainly reef fish with speargun (48%) and handlines (33%) (total males surveyed n =40).

From the value surveys, the average age of respondents was 40 years. Most respondents were men (66%). The main activity that brought cash into households was fishing: 83% of households ranked fishing as their first income-generating occupation. Dependence on reef resources as a source of protein was high. All respondents (100%) reported eating fish between twice per day and once per week, and 80% ate octopus between several times per week and once per month. The main fishing gears were small hand lines, spearguns, gillnets, and iron rods. Although not approved by most respondents, poison roots and dynamite were still used occasionally. People used on average two gears (1.98±0.13). Reef fishing was non-selective: all fish that could be caught were taken. One man fished squid only. Women were often specialised in octopus hunting or shellfish gleaning, but many also used handlines. People fished from local outrigger canoes, fiberglass kayaks, motorboats (typically fiberglass dinghies or "banana boats" with 40 horsepower outboard engines), or while swimming or walking (3.42±0.20 means of transport per fisher on average). Catch was either eaten, sold, given away, or exchanged for garden food. Catch was sold either at the main market in Madang (51.5%) or in local markets (48.5%).

Analysis of individual fishers' information collected through catch and value surveys (from the socioeconomic questionnaire) showed similar percentages of fishers using different gears and transport, (**Figure 1** and **Table 1** below). Any discrepancies between percentages found with the catch survey and those found with the value survey were due to small sample sizes (<5 people) for the category of interest.

Table 1. Number of fishers surveyed through the fisher surveys and the household surveys, for different gears, transports, and genders in my samples. In the household surveys, respondents mentioned the use of several transport types for each gear and vice versa. Hence the sum of gears/transport users is larger than the total sample size n.

	User groups	Fisher surveys (n=68)	Household surveys (n=52)
Gears	Handline users	35	47
	Speargun	19	23
	Gillnet	8	16
	Rod	6	10
	Gleaning	0	2
	Trolling line	0	1
Transports	Canoe	36	40
	Kayak	19	17
	Motorboat	1	9
	Walk	1	2
	Swim	11	3
Genders	Men	40	35
	Women	28	18

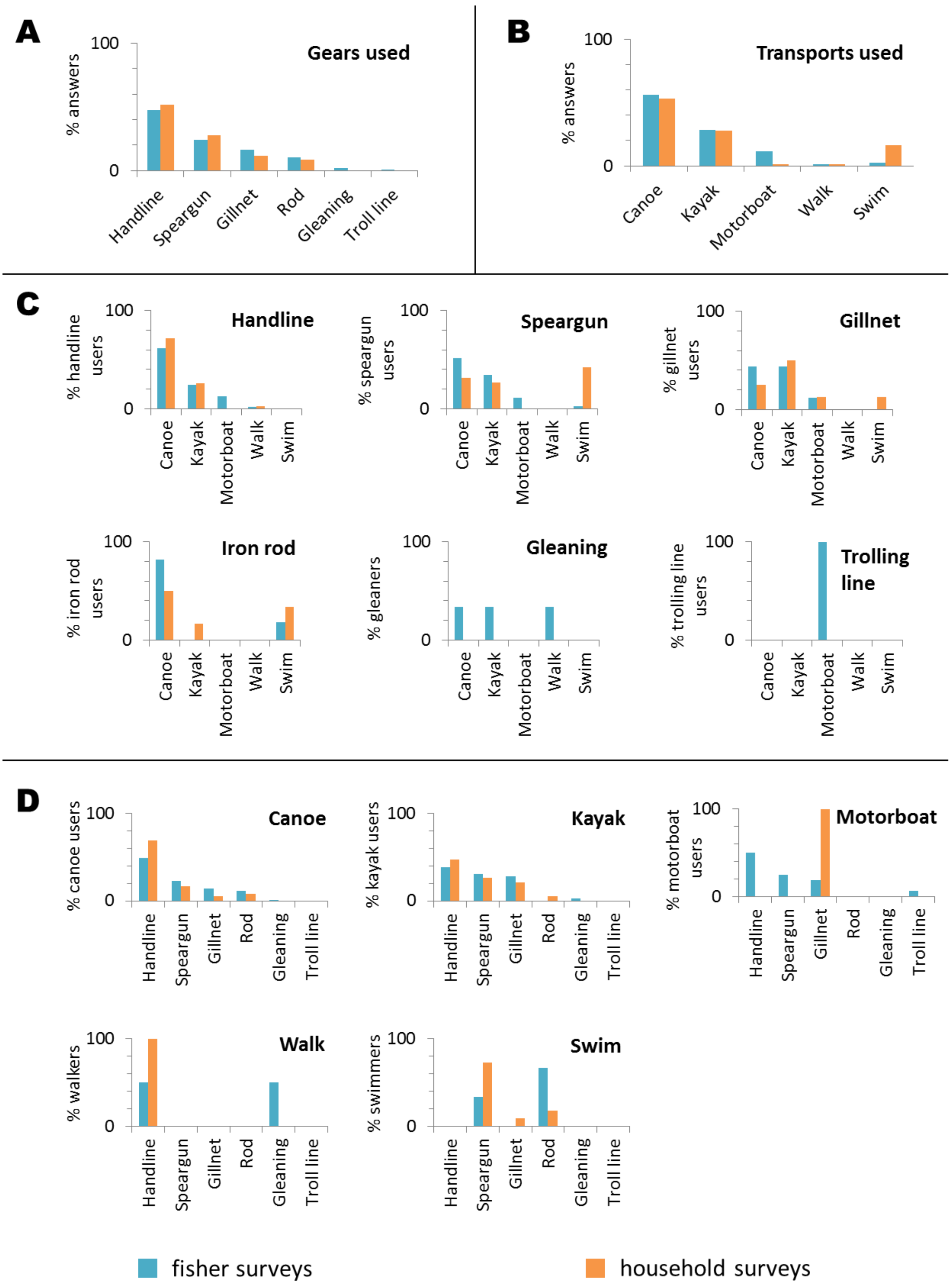


Figure 1. Comparison between fishing data collected through the fisher surveys (n = 68 fishers) and fishing data on collected through household surveys (n=52 households). a) Gears and b) transports used by the Riwo community in the Madang Lagoon. The distribution of transport types per gear type is shown in c) and the distribution of gear types per transport type is shown in d).

Appendix 6. Supplementary material for Chapter 4

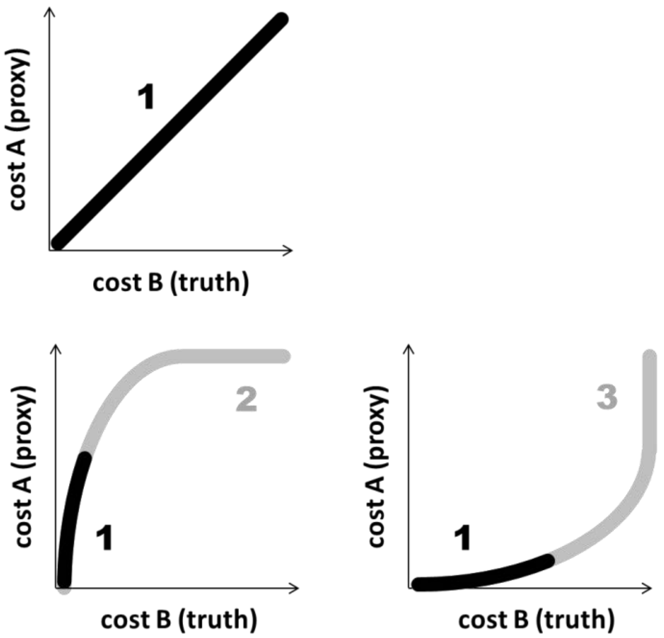


Figure 1. Graphic illustration of the information content of non-linear relationships between socioeconomic cost layers for conservation planning. Consider reserves designed to achieve representation of a conservation feature while minimising socioeconomic cost A, a proxy of socioeconomic cost B. In the case where cost A has a strong linear relationship with cost B (top left), avoiding the reservation of planning units with a cost A will always efficiently avoid planning units with cost B (1). In other words, the incidental cost B of minimising A will always be unpredictable. In a case where cost A has a strong non-linear relationship with cost B (bottom panels), the information content of the relationship is only predictable for some values of cost A (1). For example, in the bottom left graph, planning units with a high cost A can have any value for cost B (2). Therefore, minimising the reservation of planning units with a high cost A will often incur unpredictable incidental costs B. Similarly, in the bottom right graph, after reaching a certain threshold value, cost B will always be high regardless of cost A in a planning unit (3). Therefore, for systematic conservation planning, investigating the non-linear relationships between a proxy of socioeconomic cost and a cost of interest have little value since they will not allow current prioritisation tools to consider them. However, this could be the subject of further research for algorithm developers.

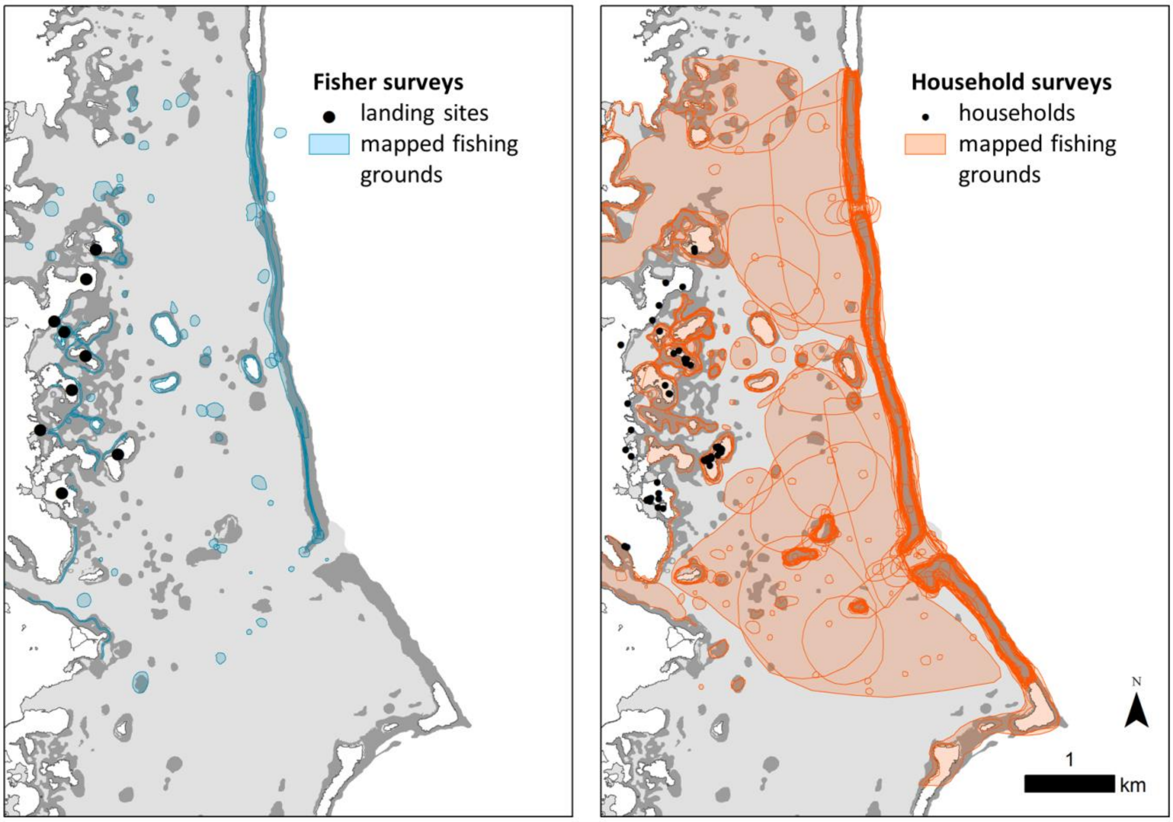


Figure 2. Spatial distribution of catch and fishing value datasets (raw data). a) Visual representation of the raw catch dataset showing the location of surveyed landing sites and the distribution of recorded fishing activities within the study area. b) Visual representation of the raw fishing value dataset showing the location of surveyed households and the distribution of places of value for fishing within the study area.

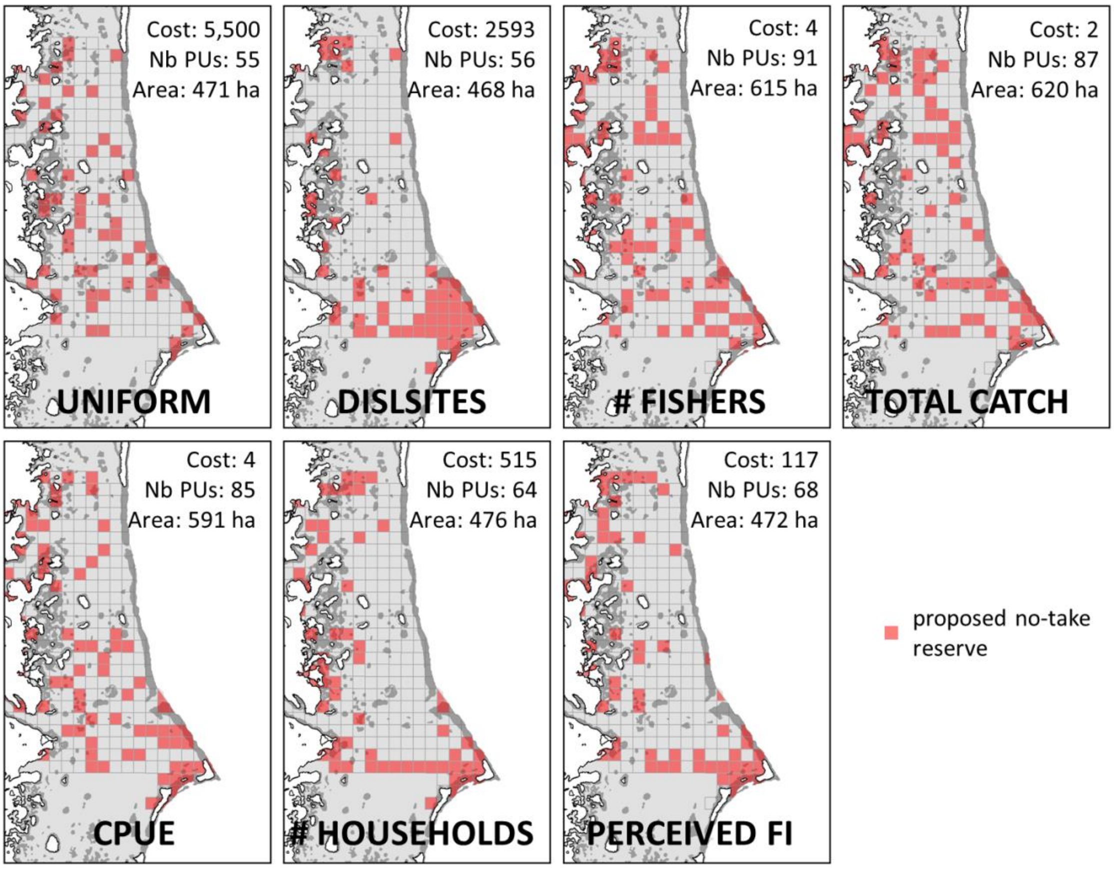


Figure 3. Best solutions (i.e. lowest-cost reserve systems) for all scenarios, across 1000 runs. Minimised costs are indicated in capital letters. All solutions contain at least 20% of each habitat type in the proposed reserve system (i.e. conservation objectives are always achieved). Selected planning units are indicated by red squares.

Appendix 7. Supplementary material for Chapter 5

Methods: measuring the difference between TOKENS, and HOUSEHOLDS, two indicators measuring the importance of areas for each benefit.

For each benefit, I created two indicators (ranging from 0 to 100) by scaling the number of households and of tokens as a percentage of the maximum value assigned to an area (hereafter "HOUSEHOLDS" and "TOKENS", respectively). TOKENS accounts for the relative importance of each of the areas identified by each household for a given benefit, while HOUSEHOLDS measures the number of households valuing each area for this benefit. To investigate differences between these two indicators, I created difference maps. For each benefit, I converted my HOUSEHOLDS and TOKENS vector datasets (polygons) into raster format (pixels). Then I subtracted the HOUSEHOLDS raster from the TOKENS raster. The resulting difference raster contained pixels with values potentially ranging from -100 to 100. Negative values (shown in green or blue tones in **Figure 1**) indicated that HOUSEHOLDS gave lower values than TOKENS to pixels for the benefit of interest, while positive ones (shown in orange or red tones in **Figure 1**) indicated that HOUSEHOLDS gave higher values than TOKENS. Pixels close to 0 (shown in yellow in **Figure 1**) indicated no notable difference between the two indices.

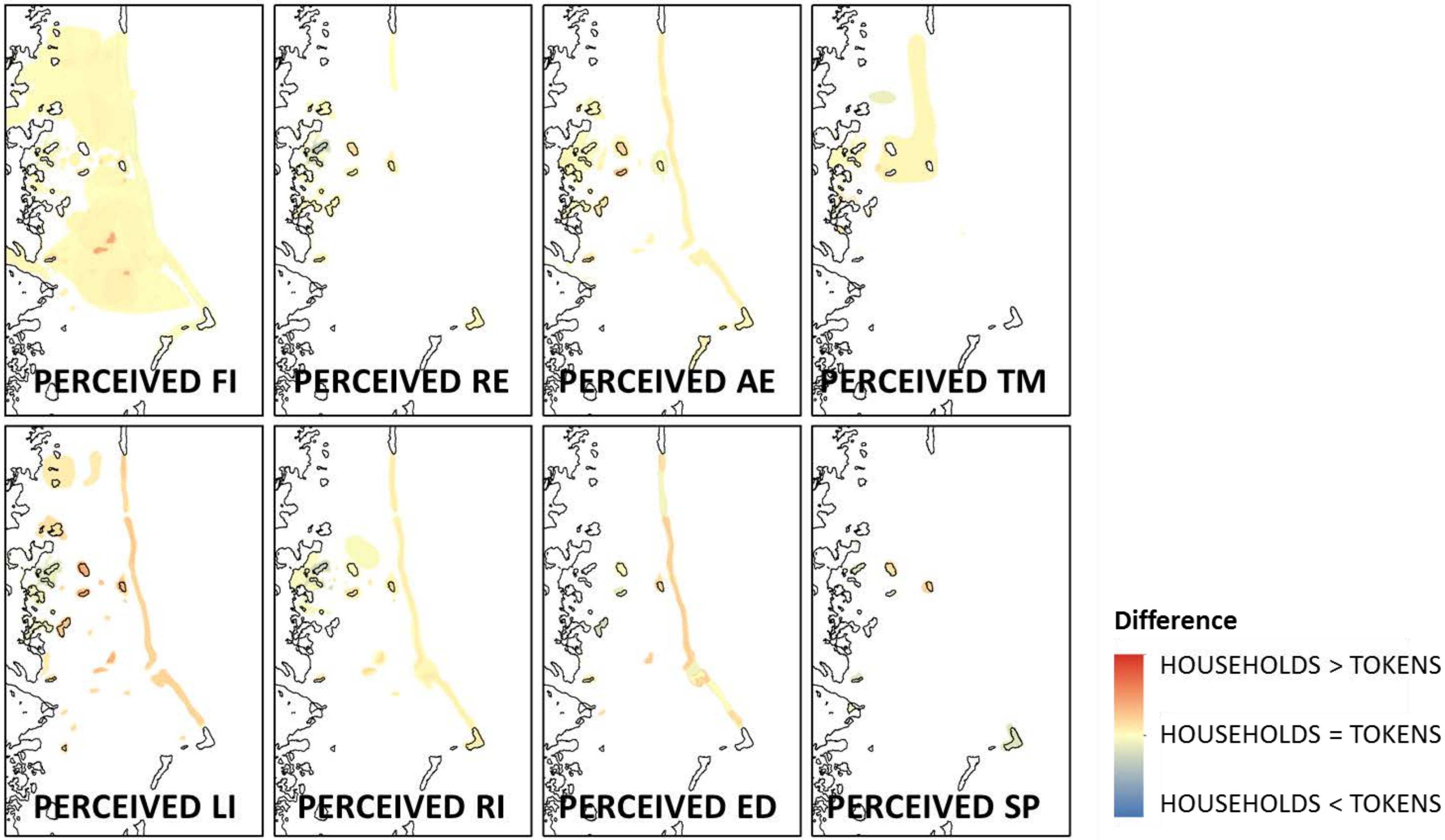


Figure 1. Map of the importance of areas as measured by the number of tokens (TOKENS) and the number of households (HOUSEHOLDS) for individual benefits. Benefits of interest included fishing (FI); recreation (RE); aesthetics (AE); traditional medicine (TM); collecting material to make lime for betel nut chewing (LI); perceived biological richness (RI); education and knowledge sharing (ED); spiritual value (SP). Orange or red areas are more important according to HOUSEHOLDS than TOKENS. Green or blue areas are less important according to HOUSEHOLDS than TOKENS. Yellow areas are areas which importance is similar based on both indicators.

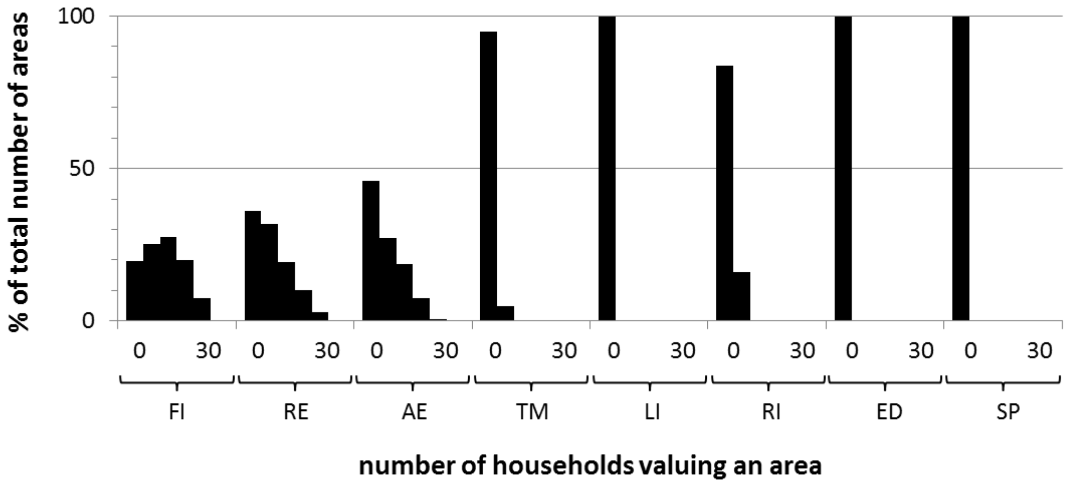


Figure 2. Frequency of areas in each "number of households" class for different ecosystem benefits. Benefits of interest included fishing (FI); recreation (RE); aesthetics (AE); traditional medicine (TM); collecting material to make lime for betel nut chewing (LI); perceived biological richness (RI); education and knowledge sharing (ED); spiritual value (SP). Benefits for which areas were valued by many households (higher classes on the x axis) reflect more consensus among households, whereas benefits for which areas were mostly valued by fewer households (lower classes on the x axis) reflect less consensus.

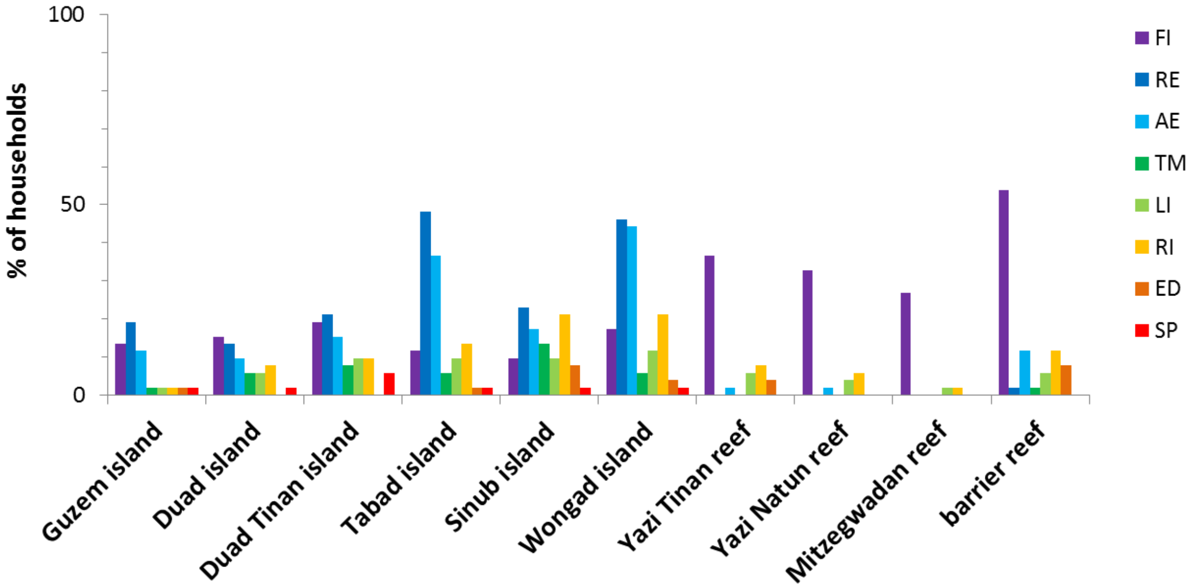


Figure 3. Geomorphologic entities of importance for people of Riwo, according to the number of households valuing them for a range of ecosystem benefits. Islands and reefs were valued for fishing (FI), recreation (RE), aesthetics (AE), traditional medicine (TM), lime material (LI), perceived biological richness (RI), education and knowledge sharing (ED), and spiritual value (SP). Bars show the maximum proportion of households valuing each area (i.e. single polygon or overlap between polygons) covering each of these geomorphologic entities.

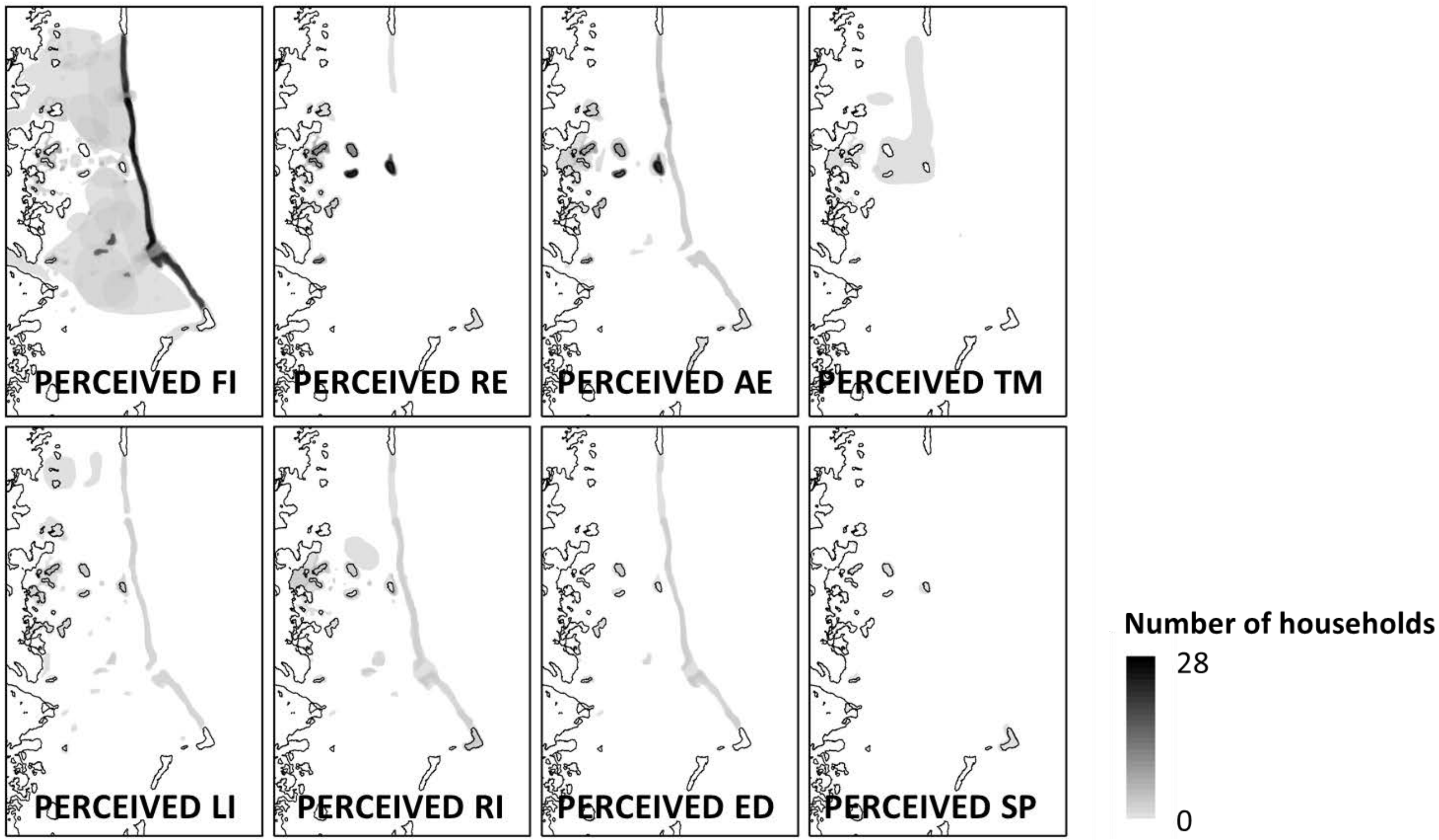


Figure 4. Number of households visiting and valuing different areas of the Madang Lagoon for the ecosystem benefits they provide. Types of benefits are: fishing (FI); recreation (RE); aesthetics (AE); traditional medicine (TM); collecting material to make lime for betel nut chewing (LI); perceived biological richness (RI); education and knowledge sharing (ED); the spiritual value of places (SP). To enhance differences, the maximum possible value of the gradient was set to 28, the maximum number of households (out of 52) valuing a same area across all benefits. This value was measured for the fishing benefit.

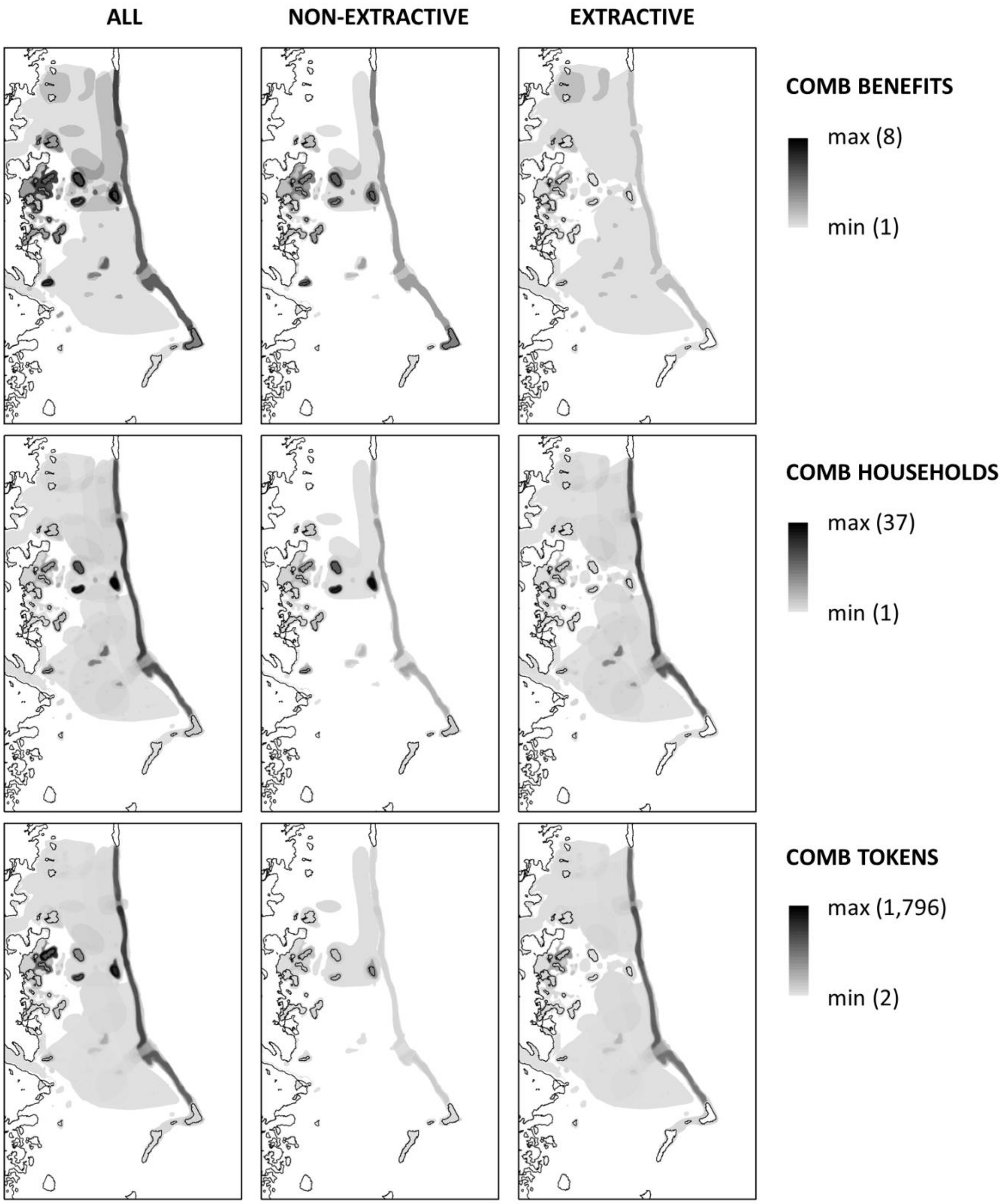


Figure 5. Importance of areas when combining benefits (raw data). All benefits combined (ALL), benefits related to non-extractive uses only (NON-EXTRACTIVE), and benefits related to extractive uses only (EXTRACTIVE). Three indicators were used: the number of benefits assigned to each area (theoretical minimum: 1 ≤ COMB BENEFITS ≤ theoretical maximum: 8), the number of households valuing each area (1 ≤ COMB HOUSEHOLDS ≤ 52) for any benefit, and the weighted sum of all tokens assigned to each area by all households (2 ≤ COMB TOKENS ≤ 1,796). See main text for equations to calculate each indicator. To enhance differences in the maps, the colour scale for each indicator was designed with colour gradients stretched from the minimum recorded value to the maximum recorded value.

Appendix 8. Supplementary material for Chapter 6

Optimum scenarios for biodiversity benefits

An ideal optimal scenario to measure o_{bio} would seek to use the reference biodiversity data to provide maximum biodiversity benefits, while minimising each given socioeconomic cost proxy, for the same number of planning units as the one used in both the proxies and random scenarios. Indeed, because the same number of planning units as the proxies scenario is used for the random scenario, the optimum scenario must be constrained similarly to allow for an unbiased comparison of scenarios. Because of the limitations associated with using Marxan, this ideal could not be attained. Therefore, I ran two different alternative optima, each with their on advantages and limitations, and chose the less biased for my analyses.

The first optimum scenario (OPTIBIO 1) aimed for the representation of as many species from the reference dataset as possible in the protected-area system, without socioeconomic cost constraints, and using the same number of planning units as the relevant proxies scenario. This allowed me to use a uniform cost (UNIFORM), and therefore control the maximum number of planning units to reserve. Because in the uniform cost layer, each planning unit had a cost of 100, I set a cost threshold of 100 times the number of planning units recorded for the proxies scenario to control the maximum number of planning units to reserve. The limitation of this optimum is that it does not consider socioeconomic costs, and therefore can never be attained by the corresponding surrogate scenario, based on the representation of a biodiversity proxy, while minimising a socioeconomic cost proxy. However, it is "locked" with the same number of planning units used in both the corresponding surrogate and random scenario.

The second optimum scenario (OPTIBIO 2) aimed for the representation of as many species from the reference dataset as possible in the protected-area system, with the same socioeconomic cost constraint as the corresponding surrogate scenario. However, Marxan solves the "minimum set" reserve design problem, which means it is not designed to allow for limiting selections to a given number of planning units with heterogeneous costs. The limitation of this optimum is that it allows for an unlimited number of planning units to be reserved to achieve maximum representation.

Optimum scenarios for socioeconomic costs

An ideal optimum scenario to measure o_{sec} would seek to achieve conservation objectives for the biodiversity proxies, while minimising the reference cost, for the same number of planning units as the one used in both the surrogate and random scenarios. However, for similar reasons as the optimum scenario for biodiversity benefits, this ideal could not be attained. Therefore, I ran two different alternative optima, each with their on advantages and limitations, and chose the less biased for my analyses.

The first optimum scenario (OPTISEC 1) aimed for the protected areas with the lowest possible socioeconomic costs (i.e. lost access to important areas for ecosystem services and benefits as perceived by the Riwo community, as per the reference socioeconomic dataset), without considering any conservation objectives. This allowed me to control the number of planning units to reserve. The limitation of this optimum is that it does not consider biodiversity representation, and therefore can never be attained by the corresponding surrogate scenario, based on the representation of a biodiversity proxy, while minimising a socioeconomic cost proxy. However, it is “locked” with the same number of planning units used in both the corresponding surrogate and random scenario.

The second optimum scenario (OPTISEC 2) aimed for the same representation objective for the biodiversity proxy as the corresponding surrogate scenario, while minimising the reference cost (PERCEIVED ALL). This scenario allows for an unlimited number of planning units to be reserved to achieve maximum representation.

Comparison of the two approaches

I calculated PEI_{bio} and PEI_{sec} using both optima (OPTIBIO 1 and OPTISEC 1, OPTIBIO 2 and OPTISEC 2, respectively), for all combinations of proxies, and all sets of planning units.

As seen in Chapter 5, PEI is calculated as follow:

$$PEI = \frac{s - r}{o - r}$$

With:

- s : biodiversity benefit (s_{bio}) or socioeconomic cost (s_{sec}) of the best protected-area system designed with a “proxies scenario”. The proxies scenario optimises the representation of a given biodiversity proxy while minimising a given socioeconomic cost proxy. The letter “S” stands for “surrogate”, to use the same terminology as in Ferrier and Watson (1997) ;
- r : biodiversity benefit (r_{bio}) or socioeconomic cost (r_{sec}) of protected areas selected randomly. The letter “R” stands for “random”;
- o : biodiversity benefit (o_{bio}) or socioeconomic cost (o_{sec}) of the best-possible protected-area system for the representation of biodiversity, or for the reduction of socioeconomic costs, respectively. The best-possible system for biodiversity benefits is designed with an “optimum biodiversity scenario”, whereas for socioeconomic costs, it is designed with an “optimum socioeconomic scenario”. The letter “O” stands for “optimum”.

To visualise the potential bias in the two indices related to the number of planning units selected, I plotted elements of the equation used to calculate PEI_{bio} and PEI_{sec}, for each set of optimum scenarios: OPTIBIO 1 and OPTIBIO 2 (**Figures 1 and 2**), and OPTISEC 1 and OPTISEC 2 (**Figure 3**), respectively.

Figure 1 shows that the denominator of PEI_{bio} is biased by the proportion of planning units when the optimum scenario OPTIBIO 2 is used for the calculation: the difference between the optimum scenario and the random scenario decreases with an increasing proportion of planning units. Analysing each element of PEI_{bio} in **Figure 2** shows that the number of species reserved in the random scenario, the surrogate scenario, and the optimum scenario OPTIBIO 1 for all combinations of proxies vary with the proportion of planning units reserved, as expected. However, the number of species reserved in the optimum scenario OPTIBIO 2 appears constant, due to the fact that the number of planning units reserved was unlimited. This explains the bias observed in **Figure 1**: as the number of planning units reserved increase, the difference between the optimum and random decreases, while the difference between the surrogate and random remains constant.

Figure 3 shows that the optimum scenario OPTISEC 1 always produces lower costs compared to the other scenarios, as expected. However, the difference between the optimum and random scenarios appears to be decreasing with an increasing proportion of planning units reserved, indicating a possible bias. On the other hand, the optimum scenario OPTISEC 2 does not always produce lower (i.e. better) costs than the other scenarios, because an unlimited number of planning units is allowed, ultimately increasing the cost of resulting reserves. This results in cases where the random scenario performs better than the optimum scenario, which provides misleading measures of PEI_{sec}.

For the reasons stated above, I chose to use OPTIBIO1 and OPTISEC 1 in Chapter 6. Further research should look into using a combination of algorithms that can solve the “minimum set” problem (which captures a set amount of biodiversity for the least cost like Marxan), and the “maximum coverage” (which captures as much biodiversity as possible beneath a fixed budget, or a fixed number of planning units).

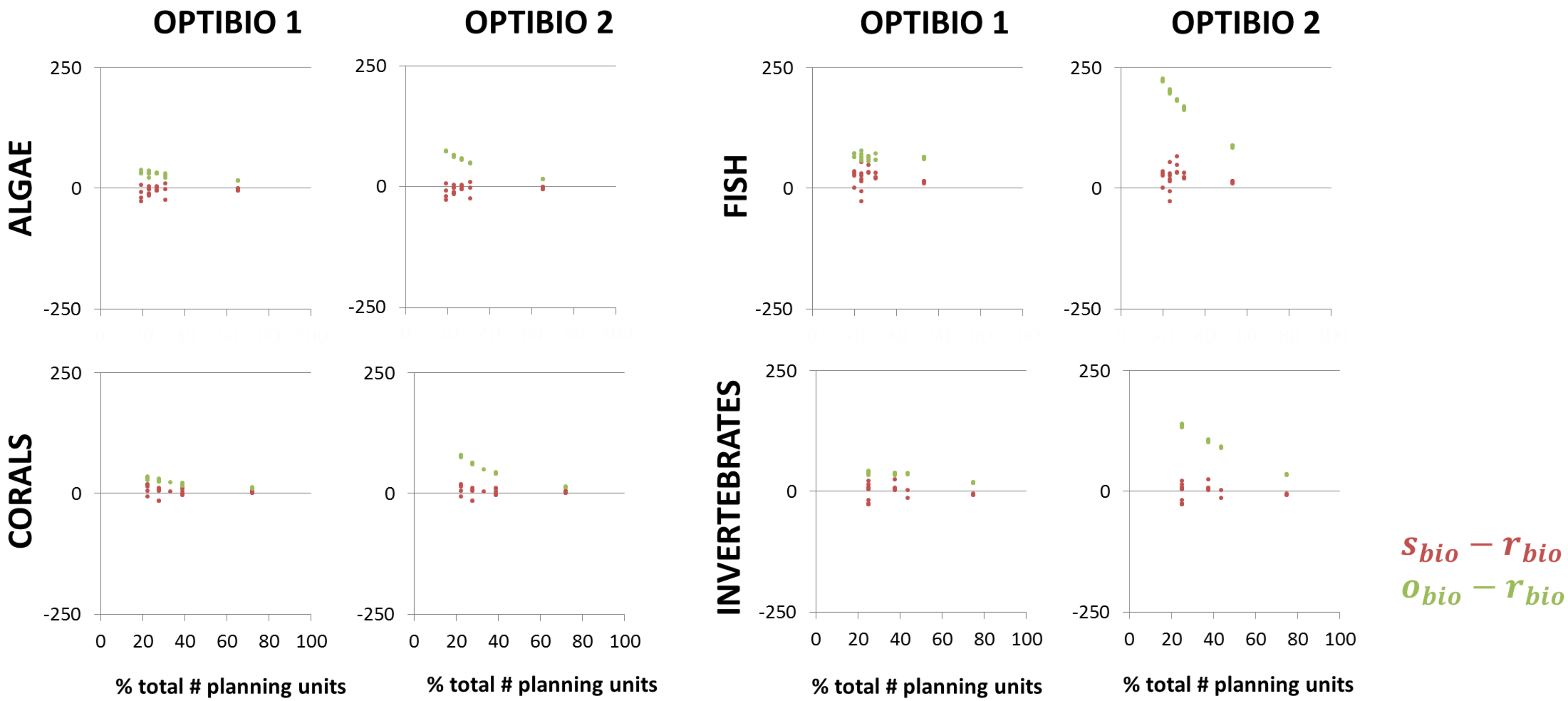


Figure 1. Relationship between numerator or denominator of PEI_{bio} and the proportion of the initial set of planning units reserved in the corresponding scenarios, using the two optimum scenarios (OPTIBIO 1 and OPTIBIO 2). The y axis represents a number of species.

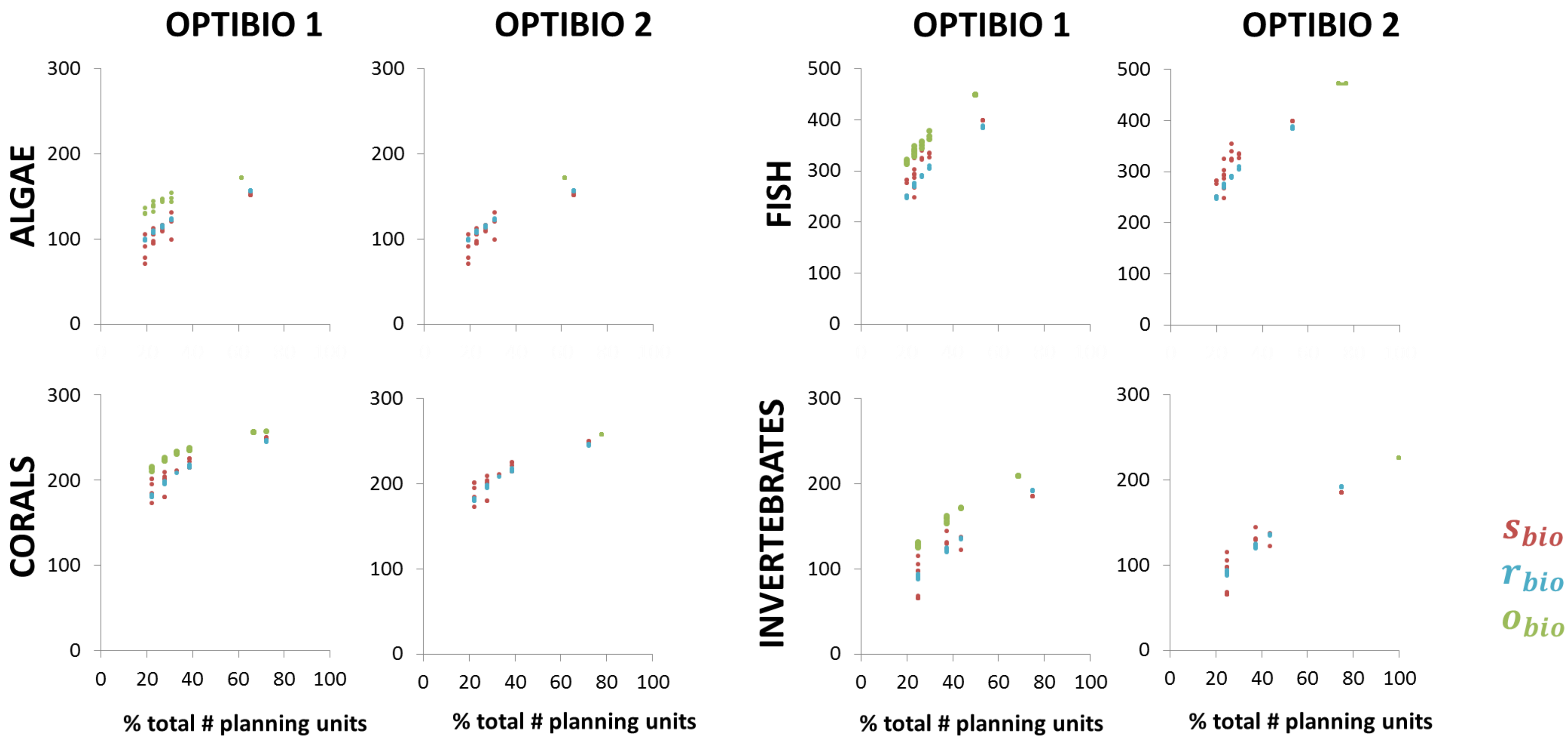


Figure 2. Relationship between variables of PEI_{bio} and the proportion of the initial set of planning units reserved in the corresponding scenarios, for the two optimum scenarios (OPTIBIO 1 and OPTIBIO 2). The three variables are measured in number of species for each taxon (algae, corals, fish, and invertebrates). The y axis represents a number of species.

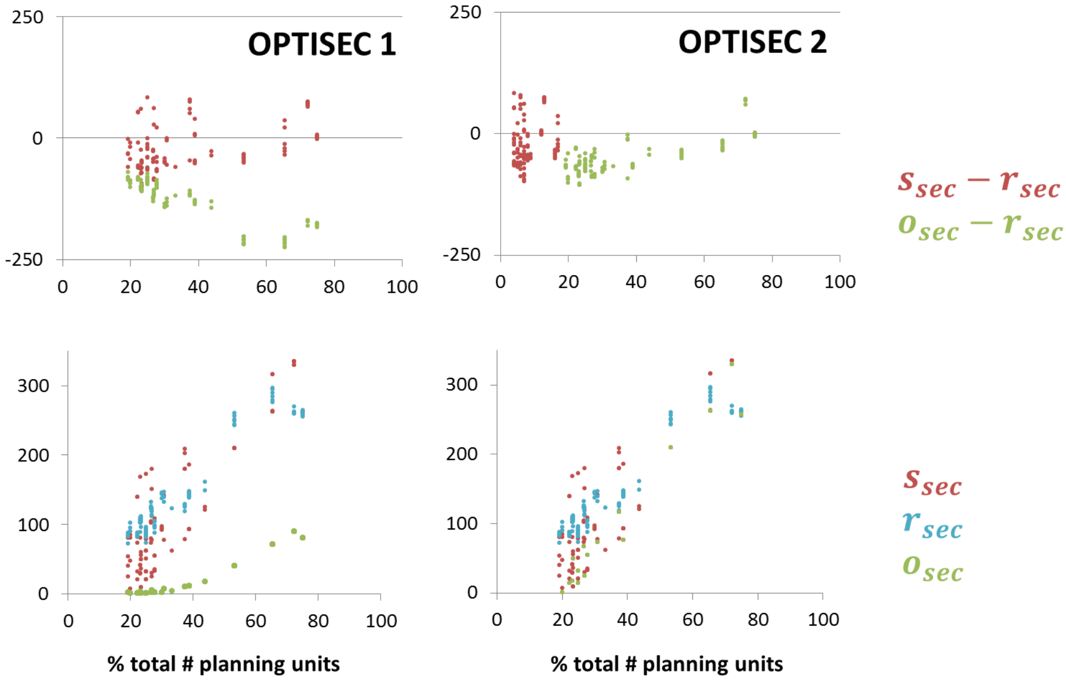


Figure 3. Relationship between numerator and denominator (top row) or variables (bottom row) of PEI_{sec} and the proportion of the initial set of planning units reserved in the corresponding scenarios, for the two optimum scenarios (OPTISEC 1 and OPTISEC 2). The y axis represents socioeconomic costs (arbitrary unit).

Appendix 9. Taxonomic expert questionnaire on costs of data collection.

1. **How many days did you spend in the field to obtain the list you sent me?**

Days in the field:	

2. **How many assistants (including volunteers) did you hire during field work, and what was their daily salary?**

	Estimated daily salary
Field assistant 1:	
Field assistant 2:	
Field assistant 3:	

3. **What essential equipment did you have to bring in the field to do your surveys (no need for a detailed list)? Estimated total price?**

Equipment	Estimated price

4. **How many full days of work (estimate) did it take you to process all your field data and obtain the list you sent me (i.e. identification, specimen preservation, photo sorting, data entry, validation...)?**

Days in the lab/office:	

5. **How many assistants did you hire during data processing, and what was their daily salary (including volunteers)?**

	Estimated daily salary
Processing assistant 1:	
Processing assistant 2:	
Processing assistant 3:	

6. **Finally, what is your job title and an estimate of your daily salary (including in-kind contribution costs)?**

Job title	Estimated daily salary