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Developing Marine Protected Area Networks in the Philippines: Reconciling Regional-Scale Planning with Community-Based Implementation

Thesis submitted by

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Statement on the contribution of others

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The thesis was supervised primarily by Prof. Garry Russ, who provided guidance and editorial assistance on all chapters, and contributed to the development of ideas explored in the thesis. Prof. Angel Alcala provided guidance and support for research undertaken in the Philippines.

The MPA data complied in Chapter 2 were provided by the Coastal Conservation and Education Foundation (CCEF), SUAKCREM, and the Marine Science Institute (MSI) at the University of the Philippines, Diliman. The reef fish survey data used to create biogeographic classifications in Chapter 3 were collected with assistance from Brian Stockwell, Rich Walker, Chloe Hunt and JJ Harvey. The data used to evaluate the biogeographic classifications in Chapter 3 were collected by Brian Stockwell (SUAKCREM). Abner Bucol (SUAKCREM) collected the data on the spatial distribution of fishing effort around Siquijor, used in Chapters 5 and 6.

Chapters 2, 5 and 6 have been published, or are accepted for publication in peer-reviewed journals. I am the senior author on all papers, and took primary responsibility for the concept, implementation, analysis, and writing of all co-authored papers. Prof. Garry Russ provided guidance, suggestions and editorial advice for all papers. Prof. Angel Alcala and Dr Alan White directed me to sources of data on MPAs in the Philippines, facilitated access to this data, and provided comments on an earlier version of the manuscript for Chapter 2. In addition, Prof. Alcala suggested the study region of Siquijor for research leading to Chapters 5 and 6, and provided comments on these manuscripts. Abner Bucol assisted with the development of the semi-structured interviews used to collect data on the spatial distribution of fishing effort around Siquijor for Chapters 5 and 6, conducted the interviews, and assisted with data interpretation. Matt Watts provided advice on the use of Marxan with Zones to set socioeconomic targets for resource use in Chapter 6. Morena Mills, Dr Natalie Ban, Dr Simon Foale and several anonymous reviewers provided useful comments on earlier versions of these manuscripts.

Declaration on ethics

The research presented and reported in this thesis was conducted within the guidelines for research ethics outlined in the *National Statement on Ethics Conduct in Research Involving Human (*1999*)*, the *Joint NHMRC/AVCC Statement and Guidelines on Research Practice* (1997), the *James Cook University Policy on Experimentation Ethics*. *Standard Practices and Guidelines* (2001), and the *James Cook University Statement and Guidelines on Research Practice* (2001). The proposed research methodology received clearance from the James Cook University Experimentation Ethics Review Committee (approval number H 2814).

REBECCA WEEKS	Date

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List of publications arising from this thesis

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Weeks, R., G. R. Russ, A. A. Bucol, and A. C. Alcala. (2010) Shortcuts for marine conservation planning: the effectiveness of socioeconomic data surrogates. Biological Conservation 143:1236-1244.

Weeks, R., G. R. Russ, A. A. Bucol, and A. C. Alcala. (2010) Incorporating local tenure in the systematic design of marine protected area networks. Conservation Letters, 3:445-453.

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

Publications in preparation

Pressey, R. L., M. Mills and **R. Weeks**. (In prep.) The plan of the day: managing the dynamic transition from region-scale conservation design to local-scale implementation.

Proposed publications

Weeks, R., G. R. Russ, A. C. Alcala, and A. T. White. Community-based marine protected areas achieve local- but not regional-scale objectives for biodiversity conservation.

Weeks, R., R. L. Pressey, G. R. Russ, and A. T. White. Guidelines for marine protected area network development in the Philippines.

Abstract

The aim of this thesis is to identify an approach to developing no-take? marine protected area (MPA) networks in the Philippines that will effectively meet both broad-scale objectives for biodiversity conservation, and the needs of local fishing communities. A national system of comprehensive, ecologically representative, connected and resilient MPA networks is required both to conserve the Philippines' extraordinary marine biodiversity, and to safeguard the livelihoods of coastal communities who are largely dependent upon fisheries for income and subsistence. Although the Philippines is widely regarded as a success story for community-based marine conservation, the current MPA system does not achieve broad-scale biodiversity conservation objectives. In this thesis I explore the need for a more systematic, science-based approach to MPA network design in the Philippines, and identify opportunities to integrate such an approach with proven community-based approaches to implementation.

The structure of the thesis reflects the hierarchical spatial nature of conservation planning. I start with broad analyses conducted at the national scale, which assess the effectiveness of existing MPAs in the Philippines in terms of broad-scale objectives for biodiversity conservation. As I then focus on progressively smaller spatial scales, I gradually increase the complexity of the analyses, incorporating socioeconomic constraints, and addressing complex trade-offs between competing objectives.

Quantifying the extent to which existing reserves meet conservation objectives and identifying gaps in coverage are vital stages in the development of systematic protected area networks. Yet, despite widespread recognition of the Philippines as a global priority for marine conservation, limited work has been undertaken to evaluate the conservation effectiveness of Philippine MPAs. Targets for MPA coverage in the Philippines have been specified in the 1998 Fisheries Code legislation, which calls for 15% of coastal municipal waters (within 15 km of the coastline) to be protected within no-take MPAs, and the Philippine Marine Sanctuary Strategy (2004), which aims to protect 10% of coral reef area in no-take MPAs by 2020. In Chapter 2, I use a newly compiled database of nearly 1000 MPAs to measure progress toward these targets and assess MPA effectiveness for biodiversity conservation. I first determined the degree to which marine bioregions and conservation priority areas are represented within existing MPAs. I then assessed the size and spacing patterns of reserves in terms of best-practice recommendations. I found that the current extent and distribution of MPAs does not adequately represent biodiversity. At present just 0.5% of municipal waters and 2.7–3.4% of coral reef area in the Philippines are protected in no-take MPAs. Moreover, 85% of no-take area is in just two protected

areas, with 90% of MPAs smaller than 1 km^2. Nevertheless, distances between existing MPAs should ensure larval connectivity between them, providing opportunities to develop regional-scale MPA networks. This analysis represents the most rigorous and comprehensive assessment of the conservation effectiveness of Philippine MPAs to date, and provides a clear indication of gaps that need to be addressed.

In Chapter 4, I seek to determine whether community-based efforts can produce ecologically functional MPA networks in the Philippines, or whether a systematic approach to MPA network design is required. My study region for this analysis is the Western Bohol Sea. This region was identified in Chapter 2 as having the highest concentration of existing MPAs in the Philippines, and thus represents an optimistic scenario for community-based MPA establishment. I first reviewed existing literature to assess the extent to which these MPAs achieve 15 local-scale objectives for fisheries management, conservation and socioeconomic development. I then conducted a series of new spatial analyses, using biogeographic classifications developed from reef fish survey data (Chapter 3), to assess whether the MPA system as a whole possesses the characteristics of an ecologically connected and functional MPA network. I found that whilst the 90 MPAs in this region fulfil a range of local-scale objectives for fisheries management, biodiversity conservation and socioeconomic development, they do not form a comprehensive, adequate and representative MPA network. This work demonstrates that even in regions where they have been applied extensively, community-based approaches to MPA establishment have not been sufficient to develop ecologically functional MPA networks.

MPA networks designed without consideration of the interests of local fishing communities are likely to fail. However, in many regions where conservation action is needed most urgently, socioeconomic data are not available at spatial scales relevant to conservation planning. In Chapter 5, I investigated the effects of using different surrogates for small-scale fishing effort at the local scale in the systematic design of an MPA network for Siquijor Province, in the Bohol Sea. Using the conservation planning software Marxan, I compared a reserve selection scenario in which socioeconomic data were not considered with scenarios that included four different surrogates for fishing effort and with empirical data on the spatial distribution of fishing effort collected through interviews. Surrogates modelled on the number of fishers or boats in each community consistently outperformed those based on population census data. However, none of the surrogates I tested were able to accurately predict fine-scale resource use patterns. Whilst socioeconomic surrogates may be able to assist conservation planners to identify regional-scale opportunities where conservation objectives may be met more easily, they cannot act as a shortcut for comprehensive consultation with fishing communities, which will be required to identify actual sites for MPA implementation.

Whilst the importance of incorporating socioeconomic factors into conservation planning is increasingly recognised, there are few examples demonstrating how such factors can be practically included in the design of protected area networks. In Chapter 6, I demonstrate for the first time how spatial zoning software (Marxan with Zones) can be used to consider local marine tenure explicitly in the design of MPA networks. By setting 'socioeconomic targets' for each local fishing community that stipulate the minimum area of fishing grounds that must remain open, I was able to design MPA networks that impact local resource users more equitably, and are therefore more likely to be socioeconomically viable. MPA networks that considered local tenure boundaries had a greater overall area and cost than those that sought to minimise costs to small-scale fishers as a single stakeholder group. Although it appears counter-intuitive that larger MPA networks may be more socially acceptable, in this context, minimising costs to each fishing community individually is likely to be more important than overall spatial 'efficiency'.

Results from this thesis indicate that neither regional-scale systematic conservation planning nor community-based MPA establishment provide an effective approach to develop MPA networks in the Philippines. Whilst community-based approaches have been successful in establishing MPAs to achieve local-scale objectives, they lack the scientific basis and spatial context required to develop ecologically functional MPA networks. Systematic conservation planning approaches provide this perspective, but are likely to be hindered by a lack of appropriate data, and may ultimately fail if they are not accepted by local stakeholders. In Chapter 7, I propose a new planning framework that incorporates elements of both regional-scale conservation planning and community-based implementation. Although yet to be empirically tested, this approach is most likely to achieve a satisfactory trade-off between achieving regional-scale objectives for biodiversity conservation and addressing the concerns of local fishing communities. The proposed framework provides an insight to local practitioners and international NGOs as to how to move towards developing systems of comprehensive, ecologically representative, connected and resilient MPA networks in regions with a similar social, economic, political and ecological context to the Philippines.

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1. Introduction

1.1. Context for the research

1.1.1. Rationale for marine protected areas and marine protected area networks

Coastal and marine ecosystems worldwide are facing significant and increasing threats, both locally, from overfishing, unsustainable fishing practices, land-based pollution and coastal habitat degradation, and globally, from the impacts of climate change. These threats have resulted in the loss of biodiversity, alteration of ecosystem structure and unprecedented fisheries declines (Pauly et al. 1998; Myers & Worm 2003; Pandolfi et al. 2003; Sala & Knowlton 2006; Halpern et al. 2008). Traditional fisheries management and single-species conservation approaches have failed to slow this decline in marine ecosystems (Pauly et al. 2002; Halpern et al. 2008). As a result, attempts to mitigate threats to the marine environment have increasingly focused on ecosystem-based approaches to conservation, including the designation of marine protected areas.

The most commonly adopted definition of a marine protected area (MPA) is: "any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment" (Kelleher & Kenchington 1992). No-take MPAs (also referred to as 'marine reserves') are recognised as a subset of MPAs in which all forms of extraction are prohibited. In reality, the level of protection afforded by MPAs exists on a spectrum, from no-take areas to those in which there are few restrictions on human activity (Kelleher & Kenchington 1992; Agardy 1994; Agardy et al. 2003).

A growing body of empirical evidence suggests that no-take MPAs can achieve a range of objectives for biodiversity conservation and fisheries management (Alcala 1988; Agardy 1999; Halpern & Warner 2003; Lubchenco et al. 2003; Lundquist et al. 2005). MPAs contribute towards marine conservation objectives by protecting species and habitats within their boundaries (Hastings & Botsford 2003; Roberts et al. 2005; Claudet et al. 2008); maintaining ecosystem functions and processes (Babcock et al. 2010); and restoring community structure (Micheli et al. 2004; Babcock et al. 2010). Numerous studies have documented increases in the density and biomass of fishery target species within no-take MPAs, demonstrating their ability to protect critical spawning stock biomass from depletion (Halpern 2003; Russ & Alcala 2003; Claudet et al. 2008; Garia-Charton et al. 2008; Russ et al. 2008; Lester et al. 2009; Molloy et al. 2009). Increasing evidence also suggests that MPAs are capable of enhancing fisheries in adjacent and surrounding areas through spillover and larval export (Roberts et al. 2001a; Gell & Roberts 2003; Russ et al. 2004; Abesamis & Russ 2005; Jones et al. 2009; Stobart et al. 2009; Halpern et al. 2010; Russ & Alcala 2010). In addition to sustaining local fisheries, MPAs provide other economic and social benefits: maintaining livelihoods and food security for coastal communities dependent on fishing for subsistence (Russ et al. 2004); strengthening community ownership of marine resources (Alcala & Russ 2006); and providing areas for education, science, recreation and tourism (Lubchenco et al. 2003). Nevertheless, MPAs are not a panacea for biodiversity conservation or fisheries management; crucially, they are unable to reduce threats that occur outside their boundaries, such as pollution, sedimentation from terrestrial run-off, and climate change-induced coral bleaching (Jones et al. 2004).

Recently, in recognition that the small scale at which individual MPAs are implemented is insufficient to counter the global scale at which threats to marine ecosystems are occurring (Agardy 2005; Osenberg et al. 2006), focus has moved towards establishing networks of MPAs (Roberts et al. 2001b; McCook et al. 2009; Gaines et al. 2010). An MPA network can be defined as "a collection of individual MPAs operating cooperatively and synergistically… to fulfil ecological aims more effectively and comprehensively than individual sites could alone" (WCPA / IUCN 2007). Whilst this definition does not require that networks be designed as such, it does imply that the constituent MPAs are ecologically connected and complementary in the biological features and processes that they encompass (McCook et al. 2009). Scientific consensus is that carefully designed networks of MPAs are better able to ensure the persistence of marine populations (Lubchenco et al. 2003; Almany et al. 2009; Botsford et al. 2009; Planes et al. 2009; Gaines et al. 2010) and insure against catastrophic disturbances (e.g. hurricanes, oil spills, coral bleaching events) (Allison et al. 2003; Jones et al. 2004; Game et al. 2008) than individual, unconnected MPAs; and that MPA networks will have greater resilience to the effects of global climate change (McLeod et al. 2009).

MPAs may also be connected through 'social networks', which aim to facilitate learning and improve management effectiveness by providing opportunities for people and organisations involved in MPA management to share experiences and pool resources (Lowry et al 2009). The existence of social links between MPA managers does not require that MPAs be connected ecologically; social networks may be developed as a catalyst for scaling up existing MPAs to form an ecologically connected MPA network (Leslie & McLeod 2007; Lowry et al. 2009), or may exist at greater (e.g. national or international) scales than ecological connections.

There is currently a global momentum towards establishing MPAs and MPA networks. Internationally, the World Summit on Sustainable Development in 2002, the Fifth World Parks Congress in 2003 and the 8th Ordinary Conference of the Parties to the Convention on Biological Diversity in 2006 have all called upon the international community to increase the amount of the world's oceans and coastal areas that are protected within MPAs. Many countries have responded to these calls by establishing national MPA network strategies, for example; Australia, New Zealand and the UK have all made policy commitments to develop comprehensive and representative MPA networks by 2012. Alongside (and indeed prior to) these national and international initiatives, coastal communities in many regions of the world have recognised the need to manage their marine resources sustainably (Johannes 2002; Cinner & Aswani 2007). For example, throughout the Pacific Islands, 'customary' or 'community-based' management practices including closed seasons, closed areas, limited entry areas, size limits and gear restrictions have been employed (Johannes 2002; Cinner et al. 2005; Aswani 2007; Cinner & Aswani 2007). Nevertheless, the total marine area protected globally remains small, with just 0.08% of the world's oceans and 0.2% of the total marine area within Exclusive Economic Zones within no-take areas (Roberts 2007; Wood 2008).

1.1.2. MPAs and MPA networks in the Philippines

The Philippines has been widely recognised as a global priority for marine conservation (Roberts et al. 2002; Carpenter & Springer 2005). Located within the 'Coral Triangle' region (Fig 1.1a), the Philippine archipelago is home to the highest diversity of marine life on Earth, with more than 1,700 species of reef fish (Allen 2008) and an estimated coral reef area of 25,060 km^2, equivalent to almost 9% of the total area of coral reefs worldwide (Spalding et al. 2001). In addition to their biodiversity value, Philippine coral reefs provide food security and livelihoods for millions of people: it is estimated that more than 60% of the population of c.94 million lives on the coast, and fisheries provide more than half of the animal protein needs of the country (Alcala & Russ 2002; World Bank 2005; White et al. 2007a).

Threats to Philippine reefs, and the sustainability of fisheries dependent upon them, come from overfishing, illegal and destructive fishing practices (e.g. dynamite and cyanide fishing), unregulated coastal development and, more recently, the expanding live reef fish trade (Gomez et al. 1994; Fabinyi 2010). These threats are all exacerbated by high population growth. Sustainable and effective management actions are urgently required to reduce these threats and safeguard the Philippines' marine resources for future generations (Alcala & Russ 2006). No-take MPAs are the primary management tool employed for marine conservation and fisheries management (White & Vogt 2000; Alcala & Russ 2006; White et al. 2006a). Alternative management approaches, such as restrictions on catch, effort, gear usage or size of fish landed, are difficult, if not impossible to enforce in multi-species, small-scale and subsistence fisheries (Russ 2002).

No-take MPAs in the Philippines have been established primarily from the bottom up, through community-based initiatives. In the 1970's and '80's, biologists and social scientists from Silliman University worked closely with local fishing communities to establish small no-take MPAs at Sumilon and Apo Islands (Russ & Alcala 1999). The success of these MPAs led them to serve as a template for the expansion of a 'community-based approach' to MPA establishment nationwide (Alcala & Russ 2006). Although biodiversity conservation is a desired outcome, MPA design is primarily driven by local community objectives: sites that fishers are willing to give up or that might create opportunities for tourism-related income (Walmsley & White 2003; Alcala & Russ 2006). Community groups assume many of the responsibilities for implementation, monitoring and enforcement, typically with assistance from local governments and non-governmental organisations (NGOs) (Christie et al. 2002; Alcala & Russ 2006; White et al. 2006a).

Community-based MPAs, established as part of wider coastal resource management initiatives, facilitated by funding provided by international donors (e.g. the United States Agency for International Development , the Global Environment Fund) to the Philippine national government, and implemented by local governments in partnership with NGOs (White et al. 2002; White et al. 2006a). Following the Philippine Local Government Code of 1991 (Republic Act 7160) and Philippine Fisheries Code of 1998 (Republic Act 8550), municipal governments have jurisdiction over their coastal and marine areas to 15 km offshore, including the implementation and management of MPAs. The devolution of responsibility for management of marine resources from national to local government is considered to have played an important role in the proliferation of community-based MPAs throughout the Philippines (Christie & White 1997; White et al. 2002; Alcala & Russ 2006; White et al. 2006a), and has since occurred in the neighbouring countries of Malaysia and Indonesia (Siry 2006).

MPA systems in the Philippines have not been planned as networks, but have developed opportunistically, largely through success stories spread via word-of-mouth (Alcala & Russ 2006). Nevertheless, attempts to develop ecological and social MPA networks around existing MPAs have begun retroactively (Alino et al. 2006; Armada et al. 2009; Eisma-Osorio et al. 2009; Lowry et al. 2009; Pietri et al. 2009). The starting point for this development has been the establishment of social, institutional and learning networks (Lowry et al. 2009). These include: PAMANA Ka Sa Pilipinas (an acronym for the Bisayan: National Alliance of Small Fishers and Communities Managing the Coast and Marine Sanctuaries of the Philippines), an alliance of representatives from more than 100 community-based MPAs that advocates for the rights of small-scale fishers and MPA wardens (Lavides et al. 2004); the National MPA Support Network (MSN), a multi-sectoral group of governmental and non-governmental organisations, people's organisations and academic institutions that aims to facilitate the exchange of information to improve MPA management effectiveness (Campos & Alino 2008); and the Philippine Coral Reef Information Network (PhilReefs), which facilitates information sharing amongst academic institutions working on coral reefs in the Philippines (PhilReefs 2008). Site-based MPA networks are being developed by the local NGOs Coastal Conservation and Education Foundation (CCEF) and Fisheries for Improved Sustainable Harvest (FISH Project), in Southern Cebu (Eisma-Osorio et al. 2009) and Danajon Bank, Bohol (Armada et al. 2009). A social networking approach that has proven to be particularly effective is the use of cross-site visits, which provide community representatives with an opportunity to observe first-hand the potential benefits of an established, functioning MPA (Alcala & Russ 2006; Pietri et al. 2009).

In 2008, then Philippine President Gloria Macapagal-Arroyo, together with the heads of state of Indonesia, Malaysia, Papua New Guinea, Timor Leste and the Solomon Islands, initiated the Coral Triangle Initiative on Coral Reefs, Fisheries and Food Security (CTI). Supported by a host of international NGOs and funding bodies, with as much as US $500 million already committed, the CTI represents the largest and most important marine conservation initiative in the region to date. A key objective of the CTI Regional Plan of Action is the development of a comprehensive, ecologically representative and well-managed system of connected and resilient MPAs and MPA networks (Coral Triangle Initiative 2008). The six member countries of the CTI are currently determining the best approach to move towards achieving this ambitious goal (Coral Triangle Initiative 2008). The Philippines leads the way amongst the Coral Triangle countries (and other tropical developing countries worldwide) in terms of both the number of existing MPAs and capacity for MPA network development (Christie & White 1997; White & Vogt 2000). The country therefore provides an excellent case study to assess the relative strengths and weaknesses of community-based approaches to MPA establishment, and their potential to achieve effective conservation of biodiversity at spatial scales larger than that of an individual MPA.

1.1.3. Systematic conservation planning

Systematic conservation planning provides a science-based framework to guide the process of designing, implementing, and managing protected area networks to achieve explicit objectives for biodiversity conservation (Margules & Pressey 2000). Applied at regional-scales, this approach allows for the consideration of complementarity (Vane-Wright et al. 1991) and connectivity (Almany et al. 2009; McCook et al. 2009; Beger et al. 2010) in the design of protected area networks, and is thus more likely to result in networks that adequately represent biodiversity than purely opportunistic approaches (Pressey et al. 1993; Margules & Pressey 2000). Emphasis is placed on planning approaches that are transparent and accountable to stakeholders, and that identify the most efficient use of limited resources available for conservation (Pressey et al. 1993). Although initially proposed for planning terrestrial protected areas (e.g. Cowling & Pressey 2003), this approach is increasingly applied in freshwater (Nel et al. 2009) and marine (Leslie et al. 2003; Leslie 2005; Stewart & Possingham 2005; Klein et al. 2008b; Fraschetti et al. 2009; Smith et al. 2009) environments.

The design of MPA networks involves decisions concerning the placement, size and shape of individual MPAs, levels of representation and replication of features within them, and levels of connectivity between them (Halpern & Warner 2003). Systematic conservation planning prescribes that these decisions should be guided by explicit objectives, preferably translated into quantitative, operational targets (Margules & Pressey 2000). This allows objective comparison of alternative options, and robust assessment of effectiveness post-implementation, facilitating adaptive management (McCook et al. 2010). Numerous scientific recommendations have been made to guide the placement, size, shape and spacing of MPAs (e.g. Airame et al. 2003; Botsford et al. 2003; Halpern 2003; Roberts et al. 2003; Shanks 2003). These have been synthesised into general principles for MPA network design, developed to ensure that MPA networks are comprehensive, representative, and adequate (ANZECC TFMPA 1998). More recently, these guidelines have been supplemented to include rules of thumb for resilience (McLeod et al. 2009) and connectivity (McCook et al. 2009).

A key feature of many systematic conservation planning initiatives is the use of decision-support tools to assist in the identification of new sites for inclusion in protected area networks (Sarkar et al. 2006). These tools are capable of incorporating multiple datasets and objectives, and provide an explicit, transparent and repeatable method to perform an inherently complex and potentially subjective process. Several software packages have been developed to perform this role, including C-Plan (Pressey et al. 2008), Marxan (Ball &

Possingham 2000; Possingham 2000), and Zoneation (Moilanen et al. 2005). The conservation planning software most frequently applied in MPA network design is Marxan (e.g. Sala et al. 2002; Airame et al. 2003; Leslie et al. 2003; Fernandes et al. 2005; Klein et al. 2008b; Ban et al. 2009b; Fraschetti et al. 2009; Klein et al. 2009; Smith et al. 2009). Marxan uses a simulated annealing algorithm (Ball & Possingham 2000; Possingham 2000) to identify sets of sites that achieve quantitative targets for biodiversity representation, such as conserving 20% of the area of each habitat type, for a minimum cost. In this context, 'cost' may be the economic cost of acquiring, designating and managing protected areas, or foregone opportunity costs to resource users (Naidoo et al. 2006; Ban & Klein 2009). In addition to identifying different options to achieve specified biodiversity objectives, Marxan can be used to set priorities for conservation action by identifying sites that occur in a large number of solutions, and to explore trade-offs between competing objectives. A new extension of this software, Marxan with Zones (Watts et al. 2009) allows users to allocate sites to a range of different zones that offer different levels of protection (e.g. no-take, habitat protection, open access), providing greater flexibility to achieve objectives. It is important to note however that decision-support tools cannot produce a final MPA network solution, and results will always need to be interpreted and refined prior to implementation.

The best-known application of systematic conservation planning to develop an MPA network is the rezoning of the Great Barrier Reef Marine Park (GBRMP), Australia, in 2004 (Fernandes et al. 2005). This process increased the area of the GBRMP within no-take areas from 4.5% to 33%, and included many stages characteristic of systematic planning: collecting spatial data on biodiversity and socioeconomic uses; assessing the adequacy of the existing no-take areas; defining explicit and quantitative principles to guide the development of a network of no-take areas (Day et al. 2002); and applying decision-support software to identify potential networks of no-take areas that achieve these objectives, which were then refined through extensive stakeholder and expert consultation (Lewis et al. 2003). The final zoning scheme adheres to scientific recommendations for MPA network design well, for example, including at least 20% of 70 'bioregions' in no-take areas, and ensuring minimum sizes for no-take areas (Fernandes et al. 2005). Similar planning processes have been successfully applied in California, under the Marine Life Protection Act (Gleason et al. 2010), and elsewhere.

1.1.4. Challenges facing the application of systematic marine conservation planning in the Philippines

Despite increasing interest in systematic approaches to MPA network design (in particular from the NGO sector), academic studies still outnumber practical applications. Examples of MPA networks that have been implemented following a systematic approach are largely restricted to regions where expertise, resources and data are relatively abundant (e.g. Australia, the US) (Osmond et al. 2010). Thus, the context in which systematic conservation planning frameworks and tools have been developed is very different to that in which we now need to apply them.

Historically, systematic conservation planning has managed the technical aspects of conservation assessment better than the complexities and uncertainties of implementation (Knight et al. 2006a; Knight & Cowling 2007; Pressey & Bottrill 2008). The design of MPA networks for example, has frequently prioritised ecological criteria, considering socioeconomic factors only at a later stage in the planning process (Roberts et al. 2003; Stewart & Possingham 2005). Yet empirical studies have demonstrated that socioeconomic, cultural and political factors are critically important to the effectiveness of MPAs (Lundquist et al. 2005). In countries such as the Philippines, if they are to be successful for biodiversity conservation, MPAs must also generate livelihoods and food security for coastal communities (Cinner et al. 2005; Alcala & Russ 2006; McClanahan et al. 2006). In the last decade, some progress has been made to better integrate socioeconomic considerations into the systematic design of MPA networks (Knight et al. 2006b; Klein et al. 2008; Pressey & Bottrill 2008; Ban et al. 2009a; Knight et al. 2010). However, there have been few attempts to undertake systematic marine conservation planning in tropical developing countries (TNC et al. 2008; but see Green et al. 2009), and it remains unclear whether this approach will be appropriate or effective in this context. The fine spatial scale of governance, existence of formal or informal local tenure and limited spatial mobility of small-scale fishers all present new challenges to the application of regional-scale conservation planning. Furthermore, the empirically proven, 'bottom-up' community-based approach to MPA implementation in the Philippines appears to be fundamentally at odds with the concept of regional-scale planning, which is inherently a 'top-down' management approach (Rodriguez et al. 2007; Abrams et al. 2009). The objectives of local fishing communities, typically concerned with livelihood security, may differ considerably from those of conservation organisations driving the planning process, resulting in potential conflict that is detrimental to effective resource management (Agardy 2005; Christie & White 2007; Cinner & Aswani 2007). An important challenge currently facing conservation planners is how to design MPA networks that will both achieve regional objectives for biodiversity conservation and earn the support of local coastal communities.

1.2. Purpose and structure of the thesis

1.2.1. Aims and Objectives

This thesis aims to investigate the potential for systematic conservation planning to support the development of MPA networks in the Philippines. To achieve this goal, I address the following specific objectives:

1. To quantify progress made towards targets for MPA establishment, and identify key gaps in the existing MPA system in the Philippines.
2. To assess the potential for community-based approaches to MPA implementation to achieve regional-scale objectives for biodiversity conservation.
3. To identify appropriate socioeconomic datasets to design MPA networks that minimise the cost of implementation to small-scale fishers.
4. To determine whether systematic approaches can design MPA networks that achieve conservation objectives within local-scale socioeconomic constraints.
5. To identify a framework for MPA network development in the Philippines that will achieve regional objectives for biodiversity conservation and be supported by local coastal communities.

MPAs in the Philippines have, to date, exclusively been established to protect coral reef-associated habitats. Thus, the focus of this thesis is the development of MPA networks in coastal regions; the design and implementation of 'high seas' MPAs, whilst undoubtedly important, lies beyond the scope of this research. Similarly, discussion of non-spatial approaches to fisheries and coastal resource management is beyond the scope of this thesis.

The level of protection afforded by MPAs exists on a spectrum, from no-take areas to those in which there are few restrictions on human activity, and MPA networks may comprise areas, or zones, with varying levels of protection. Nevertheless, to effectively achieve their objectives, MPA networks must be adequate to ensure the persistence of the features that they are designed to protect. In the Philippines, no-take MPAs are widely recognised as the most effective tool for marine conservation and fisheries management. Thus, no-take MPAs, and networks thereof, are the primary focus of this thesis.

1.2.2. Structure of the thesis

The structure of this thesis reflects the hierarchical spatial nature of conservation planning. I start with broad analyses conducted at the national scale (Fig. 1.1b), which assess the effectiveness of existing MPAs in the Philippines in terms of broad-scale objectives for biodiversity conservation. As I then focus on progressively smaller spatial scales (Fig. 1.1c and d), I gradually increase the complexity of the analyses, incorporating socioeconomic constraints, and addressing complex trade-offs between competing objectives.

In the first section of the thesis, I explore the comparative strengths and limitations of current approaches to MPA network design and implementation in the Philippines, to determine whether there is scope for systematic conservation planning to improve upon them. In Chapter 2, I assess the effectiveness of MPAs for biodiversity conservation at the national scale (Fig. 1.1b). I apply a gap analysis approach using a newly developed GIS database of existing Philippine MPAs to identify where the current MPA system falls short.

In Chapters 3 and 4, I focus in on the Western Bohol Sea region (Fig. 1.1c). This region has the highest concentration of existing MPAs in the Philippines, and thus represents an optimistic scenario for community-based MPA establishment, and an ideal system to assess the potential of such approaches to achieve both local- and regional-scale objectives. In Chapter 3, I develop a biogeographic classification for the Western Bohol Sea, applying multivariate analyses to reef fish survey data that I collected at 42 sites across the region. This classification forms the basis of the spatial analyses in Chapter 4, in which I assess the ability of an *ad hoc* system of community-based MPAs to achieve different local- and regional-scale objectives for fisheries management, biodiversity conservation and socioeconomic development.

In the second section of the thesis, I explore how to incorporate local-scale socioeconomic factors into the systematic design of MPA networks, focusing on the island of Siquijor (Fig. 1.1d). In Chapter 5, I assess the effectiveness of different surrogates for the distribution of small-scale fishing effort, and in Chapter 6, I demonstrate, for the first time, how local marine tenure boundaries might be incorporated into the design of MPA networks using the conservation planning software Marxan with Zones.

In Chapter 7 I bring together the results from previous chapters, to make recommendations for a hierarchical framework for MPA network development in the Philippines, which merges elements of regional-scale systematic conservation planning with a community-based approach to implementation. I conclude with a general

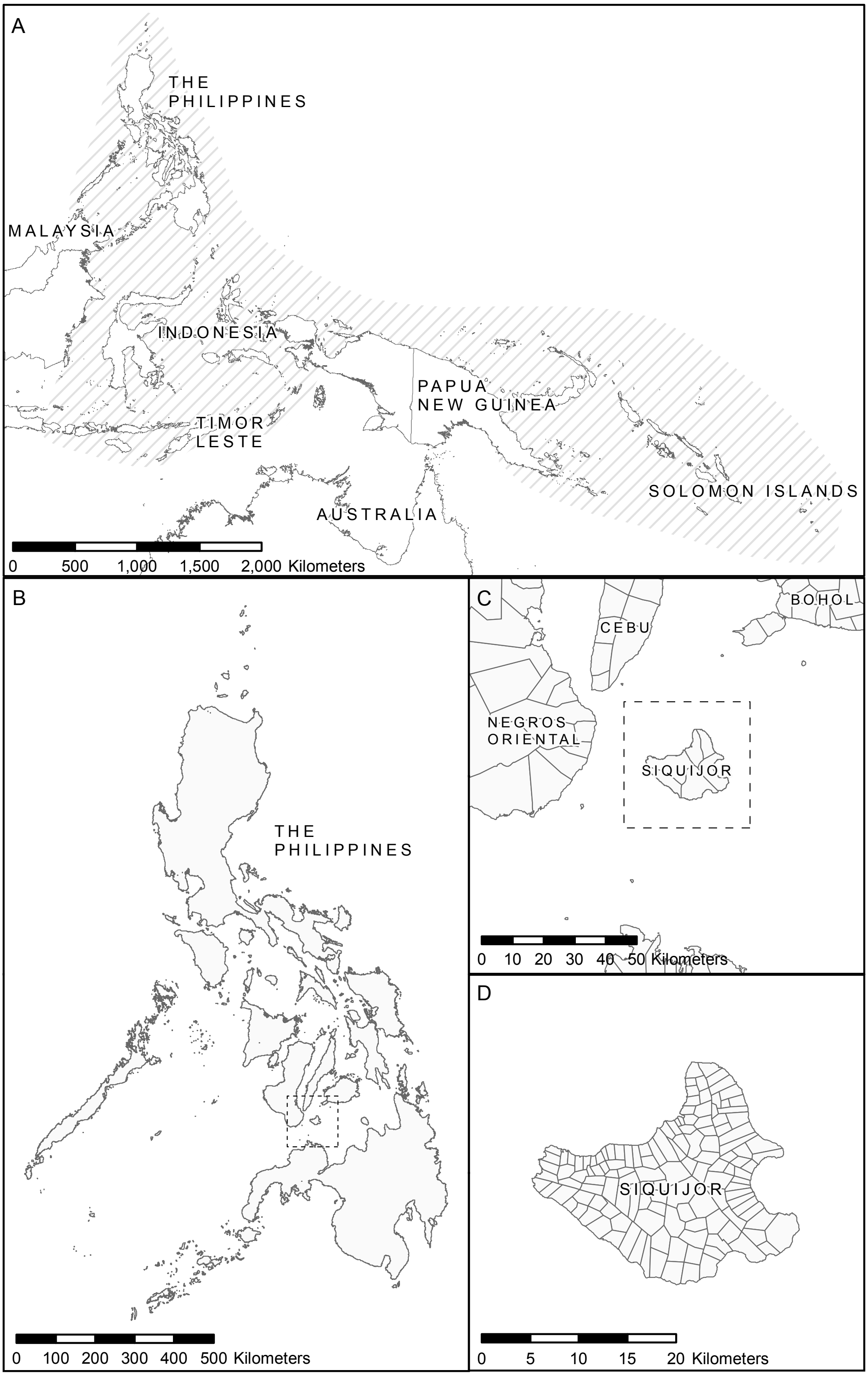


Figure 1.1. Spatial scales of analyses undertaken in the thesis; (A) the location of the Philippines, within the Coral Triangle region; (B) national scale analyses, Chapter 2; (C) regional-scale analyses in the Western Bohol Sea, Chapters 3 and 4; (D) local-scale analyses in Siquijor province, Chapters 5 and 6. Dashed boxes in B and C indicate the locations of study regions in C and D respectively.

discussion of the findings presented in this thesis, their limitations, and some recommendations for future work (Chapter 8).

2. The effectiveness of marine protected areas in the Philippines for biodiversity conservation

A version of this chapter has been published as: Weeks, R., G. R. Russ, A. C. Alcala, and A. T. White. 2010. The Effectiveness of Marine Protected Areas in the Philippines for Biodiversity Conservation. Conservation Biology 24:531-540.

2.1. Introduction

Recent attempts to mitigate threats to marine ecosystems have focused on the implementation of networks of marine protected areas (MPAs) (Sala et al. 2002; Lubchenco et al. 2003; Lundquist & Granek 2005; Lundquist et al. 2005; Roberts et al. 2005; Mora et al. 2006). A growing body of theoretical and empirical evidence suggests that no-take MPAs can simultaneously meet conservation and fisheries management objectives, protecting species and habitats within their boundaries and enhancing surrounding fisheries via spillover and larval export (Russ 2002; Halpern & Warner 2003; Hastings & Botsford 2003; Palumbi et al. 2003; Lundquist et al. 2005). The difficulty of applying catch, effort or gear restrictions makes MPA networks particularly appealing as a fisheries management tool in developing countries (Russ 2002).

The Philippines is widely recognised as a global priority for marine conservation (Roberts et al. 2002; Carpenter & Springer 2005). Located within the highly diverse 'coral triangle' region, the Philippine archipelago is home to more than 1700 reef fish species (Allen 2008) and has an estimated coral reef area of 25,060 km^2, equivalent to almost 9% of the global total (Spalding et al. 2001). Despite overwhelming pressures on marine resources (White et al. 2000; Burke et al. 2002; Roberts et al. 2002; Uychiaoco et al. 2002), the Philippines has emerged as a well-documented success story for MPA implementation due to the success of a community-based approach to management (Christie et al. 2002; Alcala & Russ 2006).

A defining characteristic of community-based MPAs is the involvement of local communities throughout the processes of planning, implementation and enforcement (White et al. 2002). In the Philippines, these MPAs are typically supported through the local government system (White et al. 2006a). Examples of community-based coastal resource management occur throughout Southeast Asia (Crawford 2004), the Pacific Islands (Johannes 2002; Aswani 2007), and South America (Rodríguez-Martínez 2008). As a leading proponent of community-based MPAs, the Philippines provides a conceptual framework for fisheries management and biodiversity conservation throughout these regions (Christie et al. 2002; Christie & White 2007).

Quantifying the extent to which existing reserves meet conservation objectives, and identifying gaps in protected area coverage is a vital stage in developing a systematic protected area network (Margules & Pressey 2000). Although small community-based MPAs have been widely proposed as a fisheries management solution that is compatible with biodiversity conservation objectives (Halpern 2003; Alcala & Russ 2006; Christie & White 2007), there has been no quantitative assessment of their effectiveness at achieving conservation goals, distinct from those of fisheries management. Whilst the fisheries objectives of MPAs are concerned with sustaining populations of target species and enhancing surrounding fisheries (Halpern 2003; Sale et al. 2005), conservation objectives typically emphasise the representation of biodiversity features (Margules & Pressey 2000; Day et al. 2002).

Previous attempts to assess the effectiveness of MPAs in the Philippines have focused on the fisheries benefits of individual reserves. Empirical studies have demonstrated positive effects on the density and biomass of target and non-target species within MPA boundaries, and spillover to adjacent fished areas (Christie et al. 2002; Russ et al. 2004; Alcala et al. 2005; Abesamis et al. 2006a; Samoilys et al. 2007). Frameworks have also been developed to evaluate the management effectiveness of individual MPAs (White et al. 2006b). There has been no systematic assessment of the effectiveness of MPAs at larger spatial scales, or specifically focusing on conservation objectives.

Here I provide an assessment of the current extent, conservation effectiveness and gaps in coverage of the national portfolio of MPAs in the Philippines, focusing on the contribution made by community-based MPAs. I use two approaches to evaluate the conservation effectiveness of MPAs. First, I determine the degree to which broad-scale measures of biodiversity are represented within existing MPAs. Whilst the representation of these features is not a conservation objective *per se*, assessing the percentage coverage of bioregions within MPAs is a convenient and commonly applied measure of conservation efficacy at broad spatial scales, where comprehensive information on the distribution of species and habitats is unavailable. Second, I assess size and spacing patterns of individual reserves in terms of recommendations based on current knowledge about larval dispersal distances. I then assess the feasibility of meeting conservation targets under a 'best-case scenario' for MPA implementation.

2.2. Methods

2.2.1. MPA Database

I sought spatial and descriptive data for marine protected areas from NGOs and academic institutions in the Philippines (Coastal Conservation and Education Foundation, Inc. (CCEF), Cebu; Silliman University Angelo King Center for Research and Environmental Management, Dumaguete; University of the Philippines Marine Science Institute, Manila), global protected area databases (World Database on Protected Areas (UNEP WCMC 2007), MPA Global (Wood 2007) and peer-reviewed and grey literature (e.g. Crawford et al. 2000). These data were compiled to form a new geodatabase of Philippine MPAs using ArcGIS v. 9.2 (ESRI, Redlands, USA), with information on the location, year of creation, size, level of protection, governance and legislation of MPAs. Sites included in the database conformed to the IUCN (1988) definition of a marine protected area: 'any area of intertidal or subtidal terrain, together with its overlying water and associated flora, fauna, historical and cultural features, which has been reserved by law or other effective means to protect part or all of the enclosed environment'. I included primarily terrestrial reserves (e.g. Watershed Forest Reserves) in the database only where information on the area of the marine component of the site was available. Protected areas in freshwater environments, e.g. Lake Taal, were not included.

After removing duplicate records, and combining records that were found to be distinct zones within a single protected area (e.g. core and buffer zones), a total of 985 MPAs were used for analysis. I split MPAs into two categories based on their governance. 'Community-based' MPAs were identified as those designated under local or municipal level ordinances, and governed by community organizations with or without assistance from local government units. 'Nationally-designated' sites were those designated under the National Integrated Protected Areas (NIPAS) act, Ramsar Convention or declared as World Heritage Sites under the United Nations Educational, Scientific and Cultural Organization (UNESCO) legislation, and governed primarily by a national government agency (e.g. the Department of Environment and Natural Resources).

Many database records were incomplete, with missing data values for one or more data fields. Rather than exclude sites without all spatial information (n=385), which would result in an underestimate of MPA extent, I substituted incomplete records of MPA size with median values for that MPA type (nationally designated = 64.57 km^2, community-based = 0.12 km^2). Geographical coordinates were available for 43% of sites; the location of other sites was only available to municipality (56%) or provincial (1%) level. I estimated locations for these sites using a gazetteer. Spatial polygons were only available for very few sites, and were not considered to be sufficiently accurate to be used in analyses.

Inconsistent terminology made it difficult to assess the level of protection afforded by individual MPAs without reading each specific ordinance. For simplicity, community-based MPAs were assumed to be entirely no-take (this was the case for 86% of MPAs with available data). Nationally-designated MPAs were not assumed to have a no-take component, unless specific information on zoning was available. Where reserves were stated to have a core zone and buffer zone, but the area of each was not provided, the core zone was assumed to be the median size of no-take MPAs.

I used nearest neighbour analysis in ArcGIS to examine the spatial distribution of MPAs. I generated 100 random MPA distributions within the same spatial constraint (MPAs can only occur within 15km of the coastline), and compared the nearest neighbour statistic for the real distribution to bootstrapped values from the random distributions. This analysis indicated whether the spatial distribution of MPAs differs from random.

2.2.2. Biodiversity Data

In my assessment of biodiversity feature representation, I used marine biogeographic regions (hereafter referred to as bioregions), conservation priority areas and marine corridors identified by the Philippine Biodiversity Conservation Priority-setting Program (Ong et al. 2002). Six marine bioregions, thirty-five priority areas and nine marine biodiversity corridors were identified through a series of expert workshops. The bioregionalisation process was informed by the evolutionary geology of the Philippine archipelago and available data for reef fish assemblages. Priority areas were identified on the basis of biogeographic representativeness, ecosystem diversity, ecosystem function, threat status, species richness and endemism, and were classified into 'extremely high' (n=13), 'very high' (n=12) and 'high' (n=10) priorities. Marine biodiversity corridors were identified as areas of importance for connectivity between bioregions. Further information on the priority-setting process can be found in Ong et al. (2002).

Biodiversity data were available in shapefile format and processed in ArcGIS. I clipped polygon areas of bioregions and priority areas to Philippine municipal water boundaries (within 15 km of the coastline), and a 20 km buffer was applied either side of each polyline corridor to create 40 km wide corridors.

2.2.3. Assessment of conservation effectiveness

Gap analysis provides a framework for assessing the extent to which biodiversity features are represented within a network of protected areas (Jennings 2000). This approach has been used extensively in terrestrial environments, at spatial scales ranging from local to global (Rodrigues et al. 2004; Araujo 2007; Maiorano et al. 2007), and has recently been applied to marine systems (e.g. Gleason et al. 2006; Wells et al. 2007; Tognelli et al. 2009). I used a gap analysis approach to determine how well existing MPAs in the Philippines represent broad-scale measures of marine biodiversity. I overlaid polygon feature layers for bioregions, priority areas and corridors with a point feature shapefile of the locations of MPAs. To quantify the degree to which each feature was represented by the current MPA portfolio, I identified unique IDs of MPAs occurring within each feature polygon and summed their area and no-take area from the MPA attribute table. I determined levels of representation as the percentage of the area of each biodiversity feature that is within MPAs.

In addition to quantifying current levels of representation, I estimated whether the existing MPA portfolio represents biodiversity better than would be expected by chance. I compared the observed number and area of MPAs occurring in each bioregion to the expected number, derived from the null hypothesis that the distribution of MPAs is proportional to the area of municipal waters in each bioregion. I compared the number of MPAs within priority regions to the mean number occurring within 100 random distributions generated using the Hawth's Analysis Generate Random Points tool (Beyer 2004).

The size and spacing of MPAs should be informed by an understanding of larval dispersal distances (Botsford et al. 2001; Gaines et al. 2003; Halpern & Warner 2003; Shanks 2003; Almany et al. 2007). I used a size-frequency distribution to identify the proportion of MPAs (for which spatial data were available) that meet minimum size requirements based on recommendations in the literature. I then performed a simple analysis of connectivity between all MPAs by calculating the Euclidean distance to the nearest MPA for each site, using the Hawth's Analysis Distance Between Points tool (Beyer 2004) in ArcGIS. The observed inter-MPA distances were compared to recommendations for MPA network design taken from Shanks (2003) and Jones et al. (2008).

2.2.4. Feasibility of attaining targets

Targets for marine protected area coverage in the Philippines have been specified in the 1998 Fisheries Code legislation, which calls for 15% of coastal municipal waters (within 15 km of the coastline) to be protected within no-take MPAs, and the Philippine Marine

Sanctuary Strategy (Arceo et al. 2004), which aims to protect 10% of coral reef area in no-take MPAs by 2020. I assessed the feasibility of meeting these targets under a 'best-case scenario' for community-based MPA implementation of one no-take MPA designated for every coastal *barangay* (the smallest political unit within a city or municipal government) nationwide. Although admittedly optimistic, I deemed this scenario to be theoretically viable, given that at least four municipalities have achieved this target (Calape, Bohol; Dauin, Negros Oriental; Inopacan, Leyte; and Magsaysay, Misamis Oriental).

Annual growth rates of MPA implementation used to predict target attainment dates were assumed to be the maximum historical rate (97 MPAs per year, in 2002). For targets that aim to protect a specified percentage of coral reef area, I assumed that all current and future MPAs are sited on reefs.

2.3. Results

To date, at least 985 MPAs have been established in the Philippines (Fig. 2.1), covering an estimated area of 14,943 km^2. Of these, 942 MPAs have a no-take component, which together total 1459 km^2. This represents 4.9% of coastal municipal waters (within 15km of the coastline), with 0.5% within no-take areas. Community-based MPAs comprised 95% of records in my database, with a combined estimated area of 628 km^2 and no-take area of 206 km^2.

Following a gradual increase in the number of MPAs between 1974 and 1990, the total number of MPAs increased almost exponentially between 1992 and 2006 (Fig. 2.2). During this time period the total area within MPAs increased ten-fold. However, the area within no-take MPAs has remained relatively constant, with only small increases in most years. Notable increases in the extent of no-take MPAs coincide with the designation of Tubbataha Reef National Park (331 km^2) in 1988, an increase in size of Tubbataha Reef National Park (from 331 km^2 to 968 km^2) in 2006 and the designation of the existing Apo Reef Natural Park (275 km^2) as entirely no-take in 2007. Growth in the number and area of MPAs appears to have slowed since 2006. However, I believe that this is due to a time lag between MPA implementation and appearance in databases, rather than a real decline in the number of sites being designated. Such a time lag is likely, as most sites are designated under local legislation, and given that there is no 'official' MPA database for the Philippines, data are only recorded for new protected areas as and when the NGOs collating MPA data become aware of them.

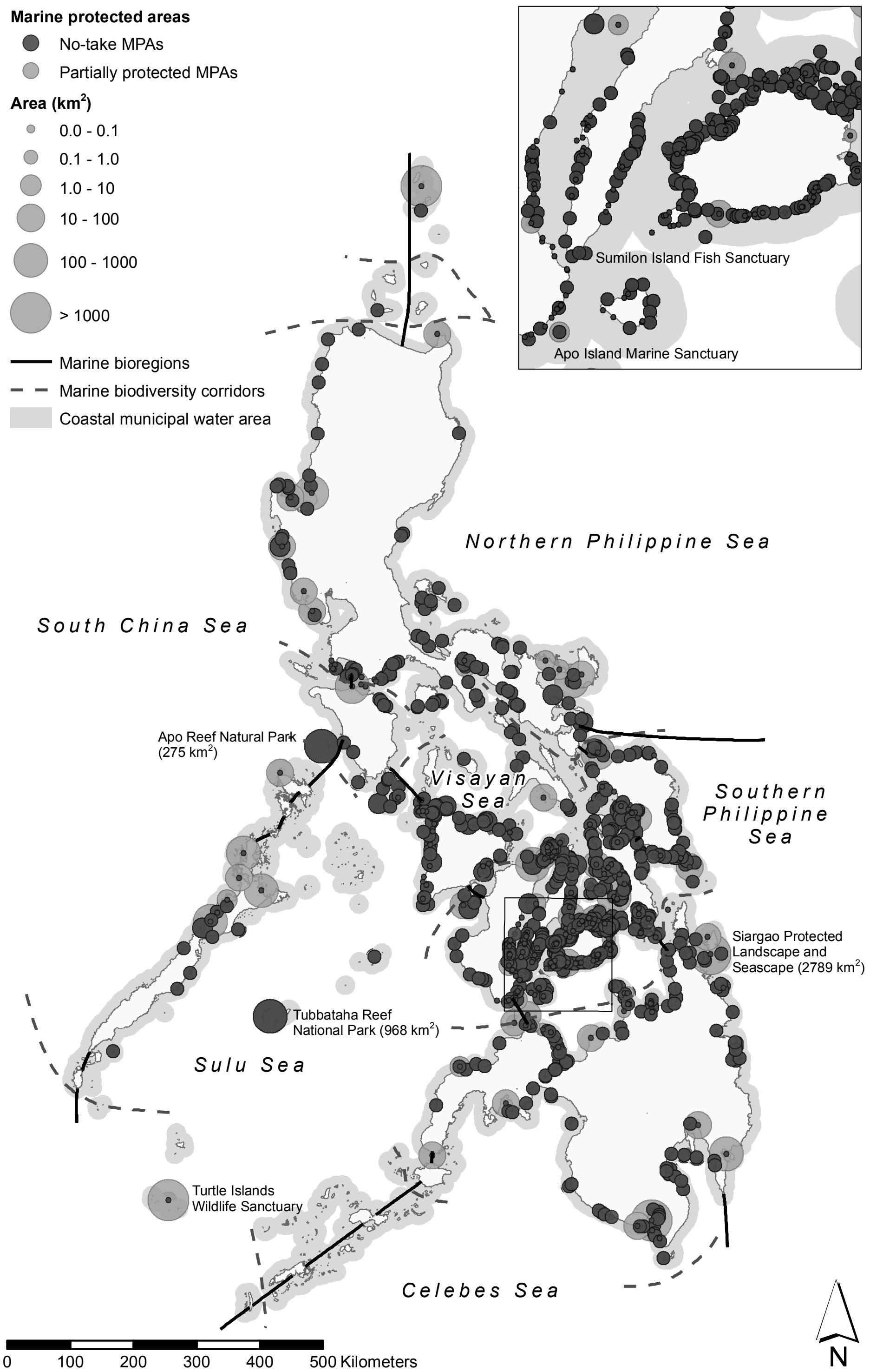


Figure 2.1. Distribution of marine protected areas throughout the Philippines, with respect to marine bioregions and corridors identified by Ong et al. (2002). The area enlarged shows the highest concentration of community-based MPAs, in the Visayan Sea bioregion. The MPAs with the largest no-take areas are Tubbataha Reef National Park and Apo Reef Natural Park; together they comprise 85% of the total no-take area in the Philippines. Siargao Protected Landscape and Seascape is the largest individual MPA, although some forms of fishing are allowed in much of its area. The Turtle Islands Wildlife Sanctuary is the most isolated MPA.

2.3.1. Representation of marine bioregions

There was variation in both the number and extent of MPAs across different marine bioregions (Number of MPAs: Chi square, χ = 368.90, d.f. = 5, P<0.001, total MPA area: Chi square, χ = 2641.46, d.f. = 5, P<0.001). The Visayan Sea bioregion contains 67% of all MPAs, 2.5 times more than would be expected based on its municipal water area (Fig. 2.1, Table 2.1). The Sulu Sea bioregion has nearly three times fewer MPAs than would be expected. However, in terms of the total area of MPAs, the Sulu Sea and South China Sea bioregions are best protected, with 1.25% and 0.66% of their municipal water area within no-take MPAs (Table 2.1). These bioregions benefit from the large no-take areas ofTubbataha Reef National Park and Apo Reef Natural Park respectively (Fig. 2.1). The high number of MPAs in the Visayan Sea does not translate into a large area protected; only 1.50% of municipal waters are within MPAs and 0.17% within no-take MPAs (Table 2.1). I found no significant effect of bioregion identity on the no-take area of individual MPAs (Kruskal Wallis, χ = 3.47, d.f. = 5, P = 0.6285).

The nearest neighbour ratio of MPAs was significantly smaller than those from random distributions (NNR = 0.314, P <0.001, bootstrapped p-value based on 10000 iterations), confirming that the spatial distribution of MPAs is non-random. Visual examination of the distribution of MPAs throughout the Philippines indicates a cluster of MPAs in the southern area of the Visayan Sea bioregion (Fig. 2.1).

2.3.2. Representation of priority areas and marine corridors

Representation of marine conservation priority areas within no-take MPAs varied between 0% - 48% (Table 2.1). The Tubbataha Reef priority area is best represented with 48% coverage within a single large no-take MPA. Two 'very high' priority regions (Balabac Island and Cuyo Islands) do not contain any MPAs.

MPAs protect priority areas for marine conservation better than would be expected if sites were selected randomly (t-test, t = -5.2797, d.f. =99, P <0.001). More than 50% of total MPA area and 70% of total no-take area is within 'extremely high priority' regions. However, this result is driven by the effect of the Tubbataha Reef MPA, which has the largest no-take area of any MPA in the Philippines and occurs within an 'extremely high priority' region.

Marine corridors are not well represented by the current MPA portfolio (Table 2.2). Four of the nine corridors do not contain any MPAs. At best, the Mindoro Calavite Tablas Triangle has just 1.24% of its area inside no-take MPAs.

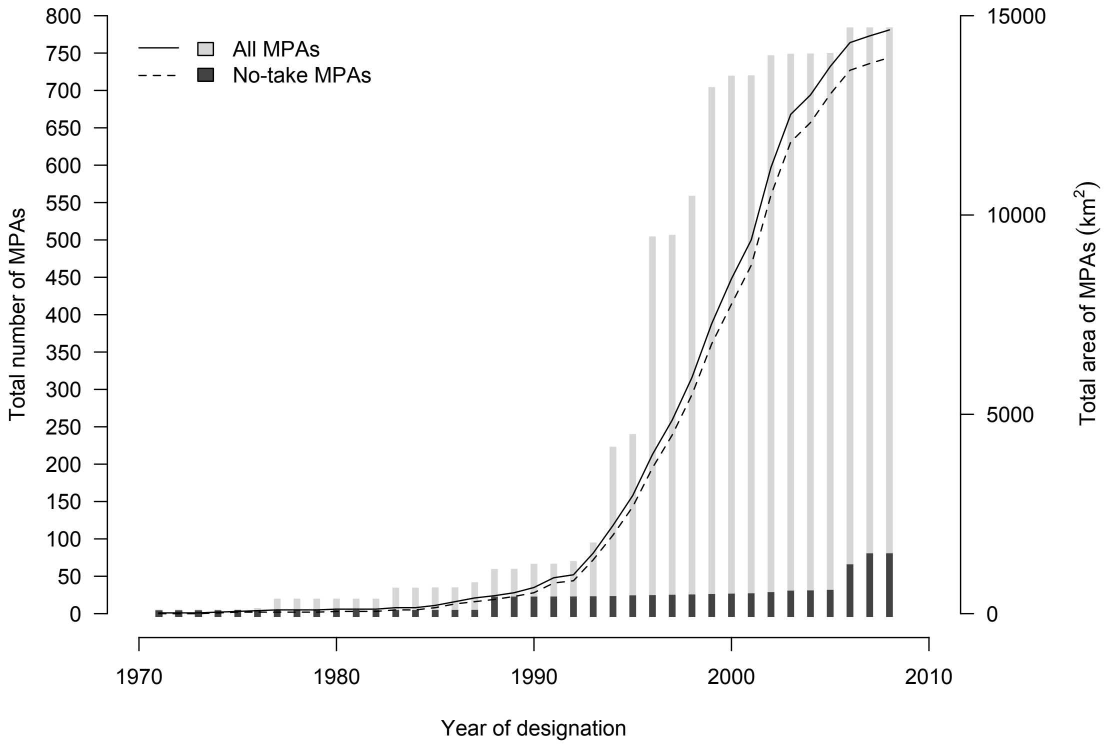


Figure 2.2. Cumulative growth in the number (lines, left axis) and area (bars, right axis) of MPAs in the Philippines between 1970 and 2008. Only MPAs with data available for year of designation are plotted (*n*=775).

Table 2.1. Representation of marine bioregions within Philippine MPAs

Marine Bioregion [a]	No. of MPAs	MPA coverage (km^2)		Municipal waters [b] protected (%)	
		All	No-take	All	No-take
Celebes Sea	50	2345.13	7.37	6.77	0.02
Northern Philippine Sea	35	2469.60	7.54	6.52	0.02
South China Sea	51	1836.93	283.71	4.26	0.66
Southern Philippine Sea	102	3500.02	33.16	12.67	0.12
Sulu Sea	90	3573.35	991.29	4.52	1.25
Visayan Sea	663	1219.50	136.50	1.50	0.17

[a] *Marine bioregions taken from Ong et. al. (2002, see Figure 2.1)*

[b] *Municipal water boundaries (15km offshore) were calculated following DENR (2001)*

2.3.3. *Size and spacing of individual MPAs*

The size of individual MPAs (for which spatial data were available) ranged from 0.01 km^2 to 2789.14 km^2 (Siargao Protected Landscape and Seascape, Fig. 2.1) with a mean of 23.60 km^2 and a median of 0.15 km^2 (n =604) (Fig. 2.3). The no-take area of MPAs ranged from 0.01 km^2 to 968.28 km^2, with a mean of 2.48 km^2 and a median of 0.12 km^2 (n = 571). The substantial difference between mean and median no-take area is due to the disproportionate contribution of the two largest no-take MPAs, Tubbataha Reef National Park (968.28 km^2) and Apo Reef Natural Park (274.69 km^2) (Fig. 2.1). Together these two sites constitute over 85% of the total extent of no-take area in the Philippines. Ninety percent of all MPAs for which spatial data were available had a total area of less than 1 km^2, with the most common size class between 0.1-0.5 km^2 (Fig. 2.3). No fully protected MPAs fall within the 'intermediate' size class of 10-100 km^2 recommended by Halpern and Warner (2003), and only two no-take MPAs exceed this size.

In general, MPAs were found to be well spaced and thus potentially well connected, with 94% of MPAs within 1-20 km of at least one other reserve, and more than 70% of MPAs within 5km of another reserve (Fig. 2.4). This corresponds to recommended inter-protected area distances for MPA networks (Shanks 2003; Jones et al. 2008). The most isolated MPA, the Turtle Islands Wildlife Sanctuary, is 250 km from its nearest MPA neighbour (Fig. 2.1).

2.3.4. *Feasibility of attaining targets*

Under a best-case scenario of one MPA designated per coastal *barangay* (in addition to existing MPAs), at the current median size of community-based MPAs (0.12 km^2), only 0.3% of coastal municipal waters, or 7.28% of coral reef area would be protected in no-take MPAs (Table 2.3). This falls short of the targets laid out in the 1998 Fisheries Code legislation and the Philippine Marine Sanctuary Strategy, which call for 15% of municipal waters and 10% of reef area to be protected within no-take MPAs respectively.

To achieve representation of 10% of coral reef area in no-take MPAs, an additional 2030 km^2 of coral reef no-take area needs to be designated. Under a scenario of one community-based MPA per coastal *barangay*, this would require an increase in the size of no-take MPAs to 0.31 km^2 (Table 2.3). Assuming the maximum rate of MPA implementation (97 MPAs per year, in 2002), this target would not be reached until 2076. To meet the target by 2020 would require the designation of an additional 545 MPAs every year.

Table 2.2. Representation of marine conservation priority areas and marine corridors within Philippine MPAs

Biodiversity Feature [a]	Bioregion [a]	No. MPAs	MPA coverage (%)	
			All	No-take
Priority areas				
Extremely high priority				
Babuyan Islands	Northern Philippine Sea / South China Sea	1	< 0.01	< 0.01
Bohol Triangle	Visayan Sea	77	0.15	0.07
Cagayan de Tawitawi Turtle Islands	Sulu Sea	1	23.47	0.00
Calamianes	Sulu Sea / South China Sea	1	0.26	0.00
Danajon Reef	Visayan Sea	65	12.89	2.14
El Nido to Ulugan Bay	South China Sea	6	17.03	0.01
Siargao Dinagat	Southern Philippine Sea	12	36.69	0.01
Surigao	Visayan Sea	28	0.11	0.11
Tañon Strait	Visayan Sea	48	0.62	0.43
Ticao San Bernardino Strait Lagonoy Gulf	Northern Philippine Sea / Southern Philippine Sea / Visayan Sea	22	3.03	0.06
Tubbataha Reefs	Sulu Sea	1	48.02	48.02
Visayan Sea	Visayan Sea	52	3.61	0.22
West Samar	Visayan Sea	29	6.66	0.45
Very high priority				
Balabac Island	South China Sea / Sulu Sea	0	0.00	0.00
Batanes	Northern Philippine Sea / South China Sea	2	26.27	0.00
Cuyo Islands	Sulu Sea	0	0.00	0.00
Honda Bay	Sulu Sea	1	0.03	0.03
Malita, Davao del Sur	Celebes Sea	2	0.12	0.12
Moro Gulf	Celebes Sea	8	0.25	0.01
Polillo Island	Northern Philippine Sea	7	0.02	0.02
South Negros	Visayan Sea / Sulu Sea	24	0.13	0.05
Sulu Archipelago	Sulu Sea / Celebes Sea	1	0.04	0.00
Tablas Strait	Visayan Sea / Sulu Sea	14	0.06	0.06
Tapal Santa Ana Valley Point	Northern Philippine Sea	1	5.03	0.00
Taytay Dumaran Bay	Sulu Sea / South China Sea	2	1.59	0.01

Table 2.2. (***continued***)

High priority				
Bolinao	South China Sea	6	2.01	0.08
Calauag Bay	Northern Philippine Sea	1	0.02	0.02
Lianga Bay	Southern Philippine Sea	1	0.07	0.07
Palanan Divilacan Bay Area	Northern Philippine Sea	1	< 0.01	< 0.01
Panay Gulf Guimaras Strait	Visayan Sea / Sulu Sea	18	0.27	0.12
Sarangani Bay	Celebes Sea	21	93.55	0.09
South Leyte	Southern Philippine Sea	5	0.08	0.08
Verde Island Passage Batangas	Visayan Sea / South China Sea	16	14.16	0.08
Zambales Coast	South China Sea	10	3.43	0.03
Zamboanga del Norte	Visayan Sea / Sulu Sea	10	0.77	0.02
Marine Corridors				
Babuyan Corridor	Northern Philippine Sea / South China Sea	3	0.30	0.00
Balabac Strait Corridor	South China Sea / Sulu Sea	0	0.00	0.00
Bohol Sea Corridor Surigao Strait	Southern Philippine Sea / Visayan Sea / Sulu Sea	28	0.19	0.01
Mindoro Calavite Tablas Triangle	South China Sea / Sulu Sea / Visayan Sea	36	2.28	1.24
Panay Gulf Guimaras Strait Corridor	Visayan Sea / Sulu Sea	11	0.17	0.03
Philippine Sea Corridor	Southern Philippine Sea / Celebes Sea	0	0.00	0.00
Sibutu Passage Sulu Archipelago Corridor	Sulu Sea / Celebes Sea	0	0.00	0.00
Tapiantana Corridor	Sulu Sea / Celebes Sea	0	0.00	0.00
Ticao Pass San Bernardino Strait Samar Sea Corridor	Northern Philippine Sea / Southern Philippine Sea / Visayan Sea	24	0.04	0.04

[a]*Taken from Ong et. al. (2002)*

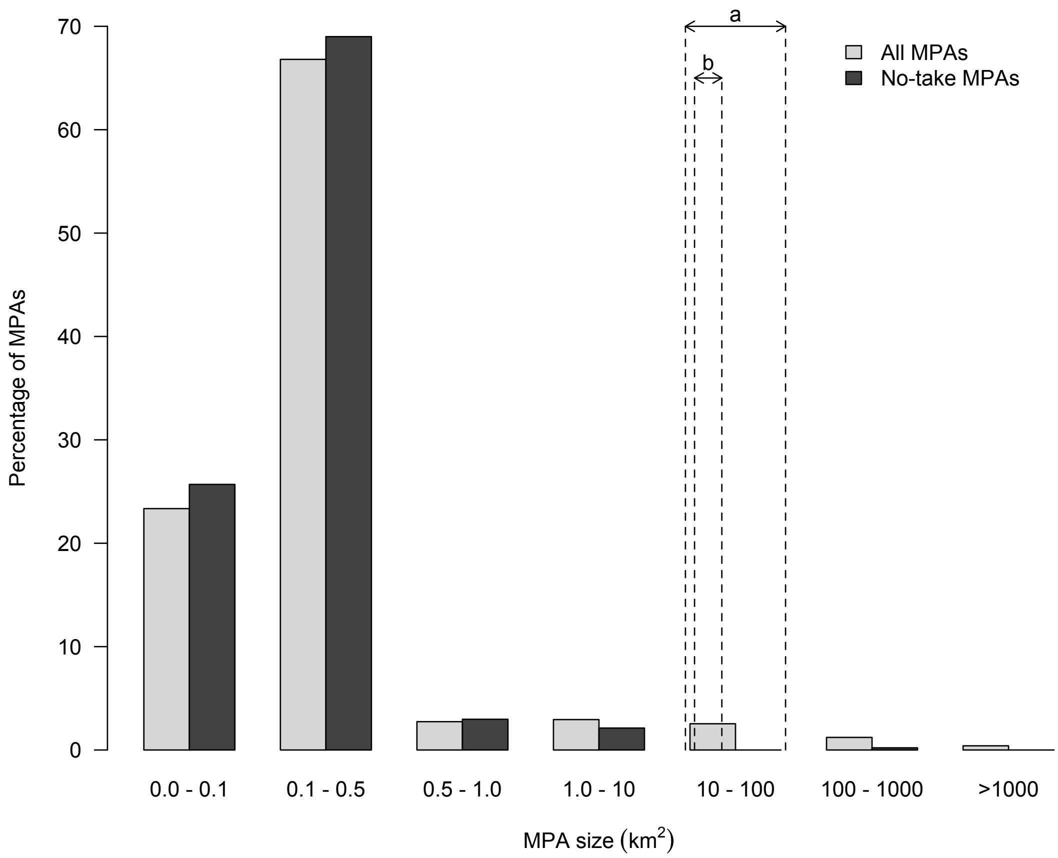


Figure 2.3. Size frequency distribution of all MPAs (light bars, *n*=600) and no-take MPAs (dark bars, *n*=567) for which spatial data were available, showing recommended MPA size based on predicted larval dispersal distances: (a) 10–100km^2 (Halpern & Warner 2003), and (b) 12.5-28.5km^2 (Shanks 2003).

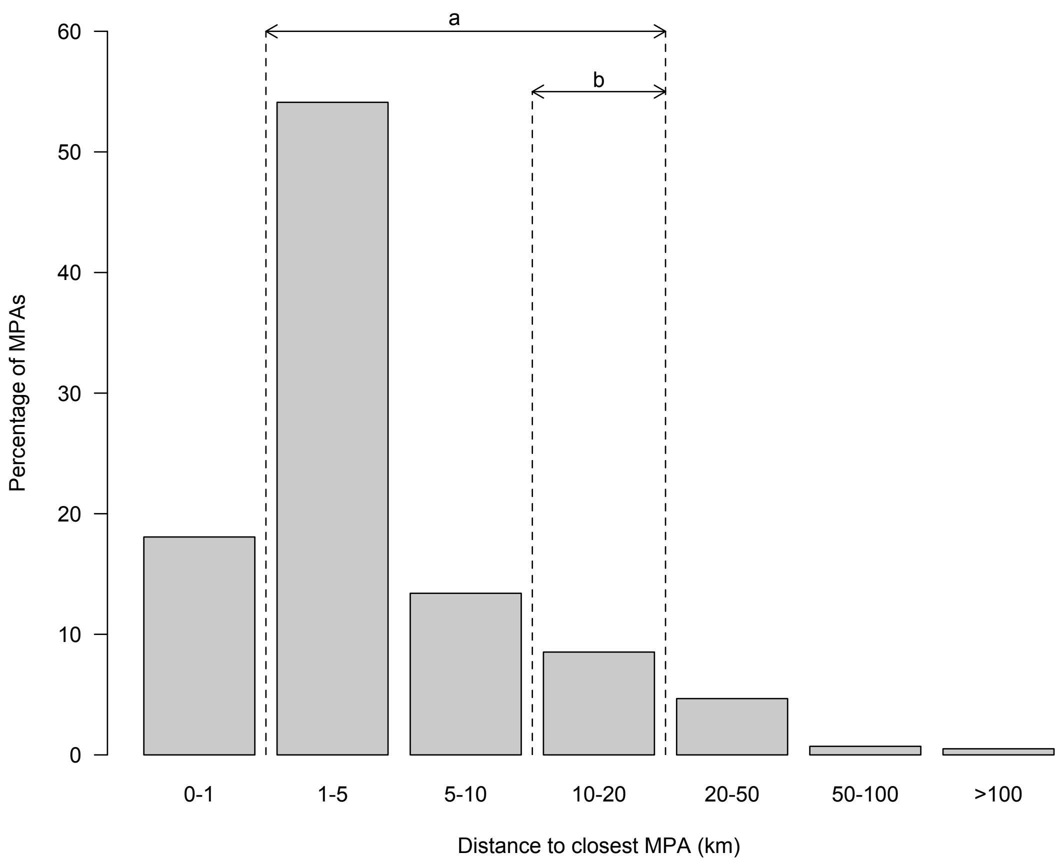


Figure 2.4. Frequency of distance to the nearest MPA, showing recommended distances between MPAs in a network based on predicted larval dispersal distances taken from (a) Jones et al. (2008) and (b) Shanks (2003). All locations were estimated to be accurate to within 5 km. Euclidean distance between MPAs was measured using the Hawth's Analysis Distance Between Points tool (Beyer 2004) in ArcGIS.

2.4. Discussion

My results provide the first comprehensive national-scale assessment of the conservation effectiveness of MPAs in the Philippines. They indicate that the current extent, distribution and size of MPAs is inadequate to fulfil conservation objectives at this scale. Whilst individual MPAs have been demonstrated to meet local-scale fisheries objectives (e.g. Alcala & Russ 2006), they do not constitute a comprehensive national MPA network that is needed to achieve adequate representation of biodiversity.

Estimates of coral reef area in the Philippines vary between 20,000 km^2 (A. C. Alcala, personal communication) and 26,000 km^2 (Burke et al. 2002). Accounting for this potential range, and the uncertainty in the total extent of MPAs arising from missing spatial information, I estimate that between 2.7% and 3.4% of coral reef area in the Philippines is protected within no-take MPAs. This falls short of the targets established by the Philippine Marine Sanctuary Strategy, which aims to protect 10% of coral reef area, and of general recommendations that a minimum of 20% and an optimum of 30 to 50% of area be set aside in reserves (Roberts et al. 2002; Airame et al. 2003; Jones et al. 2008). However, the Philippines appears to be closer to meeting these targets than other countries with similar socioeconomic contexts. McClanahan et al. (2008) estimate that Madagascar, Mauritius and Tanzania have less than 2% of their coral reef area within no-take MPAs, and Mora et al. (2006) estimate that 0.6% of coral reefs throughout Southeast Asia are protected within no-take MPAs, implying that the Philippines is performing considerably better than its geographical neighbours.

At present, the distribution of MPAs in the Philippines is not representative of marine biodiversity. The Sulu Sea bioregion and marine biodiversity corridors are particularly poorly represented. The pronounced bias in the distribution of MPAs towards the Visayan Sea likely reflects the history of community-based management efforts, which originated at Sumilon and Apo Islands (Alcala & Russ 2006), and the locations of NGOs and academic institutions that continue to facilitate the establishment of MPAs in that region (e.g. Silliman University in Dumaguete, and CCEF in Cebu) (Pollnac et al. 2001; White et al. 2002; White et al. 2006b). In addition, the Coastal Resource Management Project undertaken in the Central Visayas between 1996 and 2004 built the capacity of local municipal and city governments for integrated coastal management, focusing on the establishment of MPAs (White et al 2006b; CRMP 2004).

The rapid increase in the number of designated no-take MPAs from 1992 onwards highlights the importance of the 1991 Local Government Code (LGC). This piece of legislation devolved responsibility for managing coastal resources to municipal governments, facilitating the legal establishment of locally-managed MPAs (White et al. 2002; White et al. 2006a).

Recommendations based on predicted larval dispersal distances suggest that the size and spacing of MPAs should be varied to take account of natural variation in dispersal distances within and among species (Halpern & Warner 2003; Roberts et al. 2003; Jones et al. 2008). Despite an impressive increase in the number of community-based MPAs, the size of individual reserves remains small. The median no-take area of MPAs in the Philippines is just 0.12 km^2, compared to 6 km^2 in Latin America and the Caribbean

(Guarderas et. al. 2008), and 4.6 km^2 globally (Wood 2008). Recent evidence suggests that local retention of larvae is more common than previously thought (Jones 2005; Almany et al. 2007), indicating that even small MPAs may provide some recruitment benefits within and close to their boundaries (Jones 2005). This is supported by empirical studies that have demonstrated that MPAs smaller than 1 km^2 can fulfil fisheries management objectives, conditional on good levels of enforcement and compliance (Russ et al. 2004; Abesamis et al. 2006a). However, such small MPAs are unlikely to provide protection for larger, more mobile species (Halpern 2003), and contribute little towards achieving large-scale conservation objectives, which are typically stated as percentage area targets. Networks comprised exclusively of small MPAs will be inadequate to protect the full range of biodiversity or the processes underpinning it (Roberts et al. 2003).

Results from my analysis of potential connectivity between MPAs are more encouraging. MPAs in the Philippines are typically spaced at distances between 1 – 5 km, compared to distances of 100 km or more between MPAs in Latin America and the Caribbean (Guarderas et al. 2008). Globally, 56% of MPAs are connected within 20 km (Wood 2008), compared to 94% in the Philippines. These results indicate that Philippine MPAs are spaced at distances that are likely to promote larval connectivity. However, connectivity is not determined by distance alone, and larval dispersal models that account for the direction and strength of water movement and species' larval characteristics would be required to better predict connectivity between MPAs (Gaines et al. 2003; McCook et al. 2009).

The result that MPAs in the Philippines are failing to achieve national-scale conservation targets is not surprising, but is important to quantify nonetheless. It is perhaps more useful to evaluate progress towards these targets, by assessing how well the current MPA portfolio represents biodiversity compared to a random set of MPAs.

Table 2.3. Percentage of target area protected under different community-based MPA scenarios

Scenario	Municipal waters (%)	1km offshore (%)	Reef area[b] (%)
Current[a]	0.48	5.02	1.89[c]
Current, plus 1 MPA per coastal *barangay* ($0.12km^2$)	0.27	2.78	7.28
Current, plus 1 MPA per coastal *barangay* ($0.31km^2$)	1.19	12.2	10.24

[a] *Includes no-take area in nationally-designated MPAs*

[b] *Reef area estimate taken from Spalding et al. (2001)*

[c] *Accounts for the reef area only in Tubbataha Reef and Apo Reef MPAs*

Areas identified as the highest priorities for marine conservation contained more MPAs than would be expected by chance, indicating that the distribution of MPAs may be targeted towards areas of conservation importance. However, caution should be taken when interpreting this result. Prioritisations undertaken by expert working groups are liable to spatial and taxonomic bias towards the participants' areas of expertise. It is possible that the delineation of priority areas was influenced by the availability of biodiversity data, which is typically greatest in regions with higher concentrations of existing MPAs, and the knowledge of scientists involved in the decision-making process, many of whom are involved in MPA implementation. As such, it may be the case that the location of priority areas reflects the distribution of MPAs, rather than the other way around.

The number and area of MPAs can be misleading indicators of conservation effectiveness if they are not well managed and enforced (McClanahan 1999; Chape et al. 2005; Mora et al. 2006). Globally, MPA management effectiveness is low (Kelleher et al. 1995), and many MPAs fail to meet their management objectives (Jameson 2002). Illegal fishing within no-take MPAs is thought to be especially prevalent in Southeast Asia (White et al. 2005; Mora et al. 2006). Fisheries benefits of MPAs in the Philippines are conditional on good levels of enforcement and compliance (Alcala & Russ 2006; Samoilys et al. 2007). The same is likely to be true for conservation benefits. Estimates of the proportion of existing MPAs that are well managed and enforced were made using data from the Marine Protected Area Rating System (White et al. 2006b). Of 251 sites surveyed, just 12% were rated as 'sustained', with a fully operational enforcement system, management plan and biophysical monitoring program. A further 35% were rated as 'enforced'. If these figures are an accurate depiction of MPAs throughout the Philippines, the results presented here are likely to be optimistic assessments of conservation effectiveness.

Community-based MPAs constitute most of the number, but not area of the current MPA portfolio in the Philippines. Future projections based on a best-case scenario indicate that even if adopted Philippine-wide, community-based MPAs will not fulfil conservation targets without a significant increase in their size. In the context of the Philippines, widespread increases in the no-take area of community-based MPAs are unlikely due to socioeconomic constraints. However, where larger MPAs can be designated without unduly inhibiting local fishers, efforts should be made to do so.

The inability to meet conservation targets using community-based MPAs alone highlights the importance of large no-take areas such as Tubbataha Reef National Park and Apo Reef

Natural Park. Together, these two sites comprise more than 85% of the total no-take area of MPAs in the Philippines (it should however be noted that much of this area is a 'buffer zone' in open sea, and as such does not contain coral reef). Ensuring their continued protection should be a high priority. Larger, more remote sites present a greater enforcement challenge (White & Palaganas 1991), but if managed well, funds generated through tourism can contribute towards costs (Tongson & Dygico 2004).

Most of the existing large MPAs in the Philippines allow extractive activities within their boundaries and consequently afford little protection to marine biodiversity. However, they do offer an existing commitment to the management of marine resources. Opportunities may exist to increase the conservation value of these sites, either through designating their entire area as no-take (Apo Reef Natural Park was established in 1996, but only became fully no-take in 2007) or by establishing core no-take zones within them. This approach may be more expedient than identifying new sites for large no-take MPAs.

A major challenge to quantifying the extent of any protected area system is the dynamic nature of the network itself and of the data available about it (Mora et al. 2006). The MPA database compiled for this paper is the most comprehensive to date. However, I acknowledge that there are probably many more MPAs in the Philippines for which data were not available. Additionally, the degree of spatial bias in the reporting rate of MPAs is unknown, resulting in a potential source of inaccuracy in my analyses. I estimate that there are currently around 1100 MPAs in the Philippines. Ongoing efforts to collect and verify data for existing and newly designated MPAs will likely result in revised estimates of coverage and extent, and will allow more comprehensive assessments of their effectiveness.

The contribution of community-based MPAs towards achieving conservation targets has often been overlooked, due to their small size and poor levels of documentation. Previous attempts to assess the effectiveness of protected area networks have excluded sites that did not meet criteria based on minimum size or data availability. For example, Mora et al. (2006) excluded many MPAs due to their small size or lack of recognition by national governments. Rodrigues et al. (2004) only considered protected areas greater than 1 km^2: applying this criterion would have excluded 93% of the MPAs in my database. Such criteria clearly underestimate the number and extent of MPAs in regions where community-based management prevails. The MPA Global database (Wood 2007) currently lists 202 MPAs in the Philippines, just 20% of the sites considered here. Although community-based MPAs are typically much smaller than nationally designated areas, their no-take status and typically higher levels of compliance implies that they might be more effective in terms of achieving conservation objectives.

Small, community-based MPAs are vital to ensuring fisheries sustainability and will continue to play an important role in marine conservation efforts in regions such as the Philippines, where opportunities to designate large no-take areas are limited. For this reason, it is important that the role they play is recognised and understood. This will only happen if adequate systems for recording and monitoring these sites are put in place.

2.5. Conclusions

My results highlight a need to expand the area within no-take MPAs in the Philippines. This should be done strategically, to address biases in the representation of bioregions and target priority areas for marine conservation. The development of systematic, regional-scale networks of MPAs that address both fisheries sustainability and biodiversity conservation objectives should be a priority.

To better satisfy conservation objectives, I recommend that the Philippines work on three tracks to improve MPA implementation and effectiveness. First, continue efforts to increase the number and, where possible, size of community-based MPAs. Second, designate additional large no-take areas specifically to address conservation goals while also achieving benefits to fisheries. Finally, continue to build capacity of both local government supported MPAs and national agency managed MPAs to improve overall management effectiveness and governance of individual sites.

3. Biogeographic Classifications for the Western Bohol Sea and Siquijor

3.1. Introduction

A primary objective of systematic conservation planning is the development of ecologically representative protected area networks (e.g. Margules & Pressey 2000). To achieve this goal requires knowledge of spatial biodiversity patterns. However, the paucity of distribution data for marine species, and the logistical difficulty of collecting such data means that conservation planners frequently rely on the use of biodiversity surrogates (Rodrigues & Brooks 2007). In many cases, benthic habitat types are the primary features targeted for representation in MPA networks (e.g. Beck & Odaya 2001; Sala et al. 2002; Airame et al. 2003; Leslie 2005; Klein et al. 2008; Lindsay et al. 2008; Fraschetti et al. 2009). Habitat data can be obtained relatively quickly and inexpensively (Ward et al. 1999; Mumby et al. 2008) through interpretation of satellite or aerial images (Mumby & Edwards 2002; Dalleau et al. 2010; Wabnitz et al. 2010), and it is assumed that an MPA network that adequately represents the full range of habitat types within a planning region will also represent the species that inhabit them (Ward et al. 1999; Beck & Odaya 2001; Mumby et al. 2008). However, the effectiveness of habitat types as a surrogate for marine biodiversity varies with the spatial scale of observation and the resolution of habitat classification (Banks & Skilleter 2007; Lindsay et al. 2008). When classified broadly, representation of habitat types alone is unlikely to result in fully representative MPA networks: the species assemblage at one 'fringing coral reef ' site may differ greatly to that on another fringing reef 30 km away. Thus, guidelines for MPA network design recommend that a minimum percentage of each habitat be protected within each biogeographic region (Roberts et al. 2003).

3.1.1. Biogeographic classifications

Biogeographic regions, or 'bioregions', are geographically distinct areas that are characterised by similar biological composition and environmental conditions (Spalding et al. 2007; Green & Mous 2008; McCook et al. 2009). Biogeographic classifications provide a fundamental framework for the development of representative MPA networks (Day et al. 2002; Edgar et al. 2004a; Lourie & Vincent 2004; Spalding et al. 2007) and form the basis of conservation planning and management strategies of many governments (e.g. Day & Roff 2000; Commonwealth of Australia 2005) and NGOs (Olson & Dinerstein 1998; Beck & Odaya 2001; Spalding et al. 2007). Classifications may be informed by biological (e.g.

species distributions), ecological (e.g. ecosystem or habitat types) or physical (e.g. bathymetry) data, or a combination of these. Bioregions can vary in their spatial extent and resolution (Lourie & Vincent 2004; Spalding et al. 2007; Shears et al. 2008), and the data used to inform their delineation may vary accordingly. Whilst broad-scale classifications often have a basis in physical characteristics such as bathymetry, productivity and oceanographic processes, those undertaken at local scales are more likely to emphasise biotic factors such as species distributions, dominant habitats or substrate types (Day & Roff 2000; Lourie & Vincent 2004).

3.1.2. Application to conservation planning

To successfully inform conservation planning, the spatial scale at which bioregions are delineated needs to reflect the scale at which management actions are applied. Lourie and Vincent (2004) provide examples of marine biogeographic classifications undertaken at scales ranging from global to local site level. Ideally, classification systems will comprise a hierarchical scheme that reflects the variety of spatial scales at which ecological processes occur (Lourie & Vincent 2004; Spalding et al. 2007). Broad-scale bioregions, at spatial scales of hundreds to thousands of kilometres, are sufficiently large to encompass ecological and life history processes and thus form intuitive planning regions (Lourie & Vincent 2004; Shears et al. 2008; Mills et al. 2010). Finer scale subdivisions are required to guide the selection of sites for inclusion in MPA networks.

3.1.3. Marine biogeographic classifications for the Philippines

The most comprehensive global marine biogeographic classification to date is the Marine Ecoregions of the World (MEOW) system (Spalding et al. 2007). MEOW is a hierarchical system comprising 12 realms, 62 provinces, and 232 ecoregions. This classification is advantageous in that it is globally consistent, and has been used by international NGOs to guide global-scale conservation efforts (Spalding et al. 2007). Nevertheless, the spatial extent of ecoregions, the smallest-scale units in the system, is too large to practically inform MPA network design. For example, the Eastern Philippines ecoregion (Fig. 3.1a) comprises an area of approximately 1180,000 km^2, which encompasses almost the whole of the Philippines, excluding only the island of Palawan.

The Philippine Biodiversity Conservation Priority-setting Program (Ong et al. 2002) identified six marine biogeographic regions for the Philippines on the basis of reef fish assemblages, evolutionary geology, and predominant ocean circulation patterns (Fig. 3.1b). I used this classification in Chapter 2, to assess the effectiveness of Philippine MPAs at representing biodiversity at a national scale. Although these bioregions are considerably smaller than those proposed by the MEOW system, the small size of MPAs in

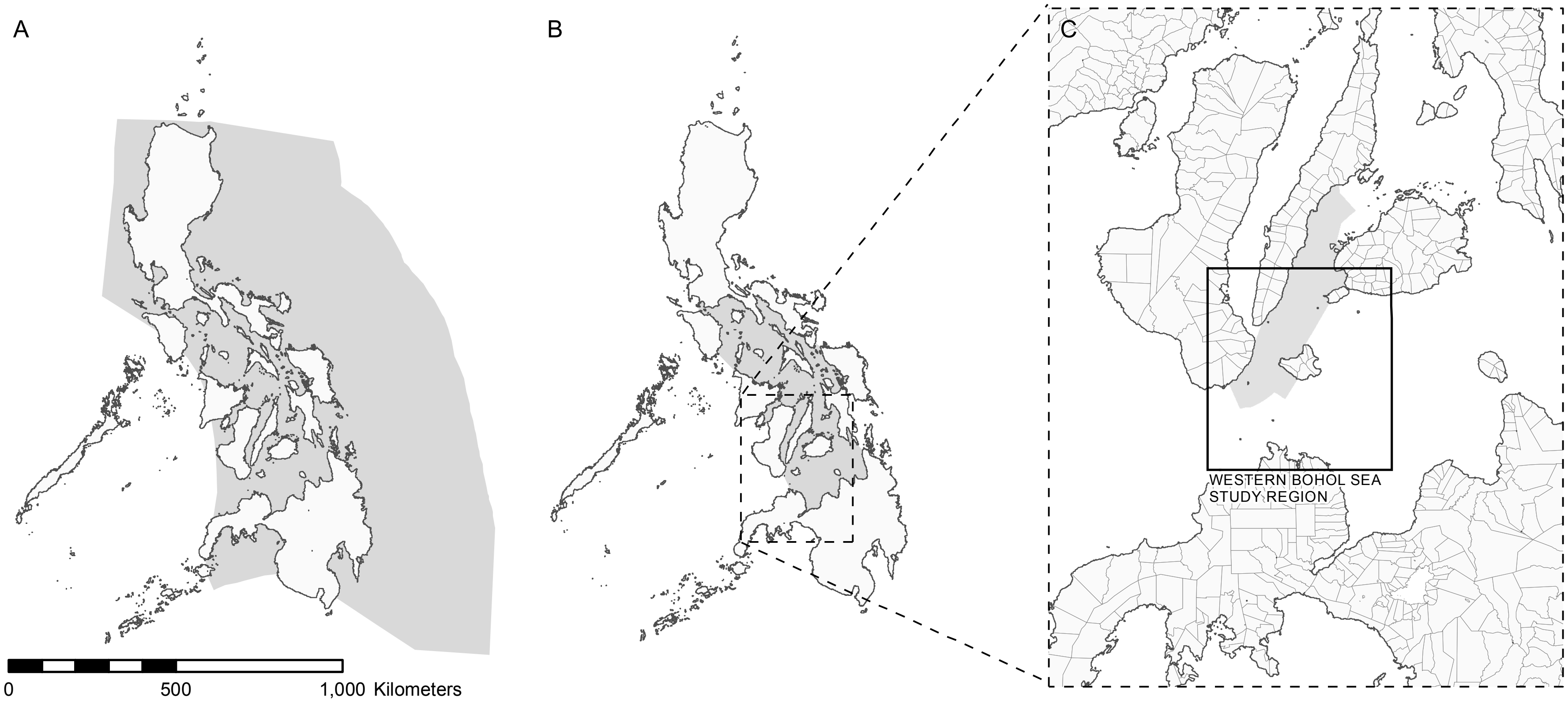


Figure 3.1. Comparison of the spatial scale of biogeographic classifications (shaded areas) for the Philippines: (a) Eastern Philippine Ecoregion, 1,180,000 km^2 (Spalding et al. 2007); (b) Visayan Sea Bioregion, 110,400 km^2 (Ong et al. 2002); and (c) Cebu Strait Fisheries Ecosystem, 3,900 km^2 (Green et al. 2004).

the Philippines (most are <1 km2 ,Weeks et al. 2010a, Chapter 2) demands that classifications be undertaken at an even finer scale if they are to inform MPA network design. Green et al. (2004) identified seven fisheries ecosystems in the Central Visayas region (Fig. 3.1c) ranging between 2,475 km^2 (Danajon Bank) and 285,612 km^2 (Sulu Sea). Although at the smaller end of the spectrum these regions approach an appropriate scale for conservation planning in the Philippines, delineation of fisheries ecosystems was influenced primarily by socioeconomic (e.g. fishing gears utilised, species targeted), rather than ecological factors.

The aim of this chapter is to develop fine-scale bioregional classifications for the Western Bohol Sea and Siquijor study regions, in order to assess how well existing MPAs represent biodiversity at regional scales (Chapter 4), and to explore options for designing ecologically representative MPA networks in the Philippines (Chapters 5 and 6).

3.2. Methodology

3.2.1. Reef Fish Data

Underwater visual census surveys were conducted at 42 sites across the region between November 2007 and May 2008 (Fig. 3.2). Survey sites were selected as a random sample of 1 km sections of the coastline, stratified by administrative province (Fig. 3.2). I recorded the abundance of 112 species (Table 3.1) of reef fish from 12 families using a timed swim method. Divers descended directly to 18 m, then swam in a free transect parallel to the reef, with time spent evenly across the depth gradient, reef slope, crest and flat. Each survey was of 60 minutes duration, and covered a mean distance of 550 m. The abundance of fish species observed 5 m either side of the transect was recorded cumulatively on a log4 scale (category 1 = 1 fish; category 2 = 2–4 fish; category 3 = 5–16 fish; category 4 = 17–64 fish; category 5 = 65–256 fish; category 6 = 257–1024 fish; category 7 = 1025–4096 fish)(Russ 1985; English et al. 1997). Three 60 minute surveys were undertaken at each site, with staggered entry points so that the majority of the site was surveyed at more than one depth. Preliminary surveys undertaken at sites known to have relatively high diversity and abundance of fishes (inside well-established MPAs) indicated that this survey effort was sufficient such that all focal species occurring at the site would be recorded. Data were pooled at the site level prior to analysis.

A second reef fish data set was collected independently at 41 sites across the study region (Fig. 3.2) between February 2006 and May 2008. These data were used to evaluate the reliability of the bioregional classification for the Western Bohol Sea region (see section 3.2.4), and to increase the number of survey sites available to identify bioregions for

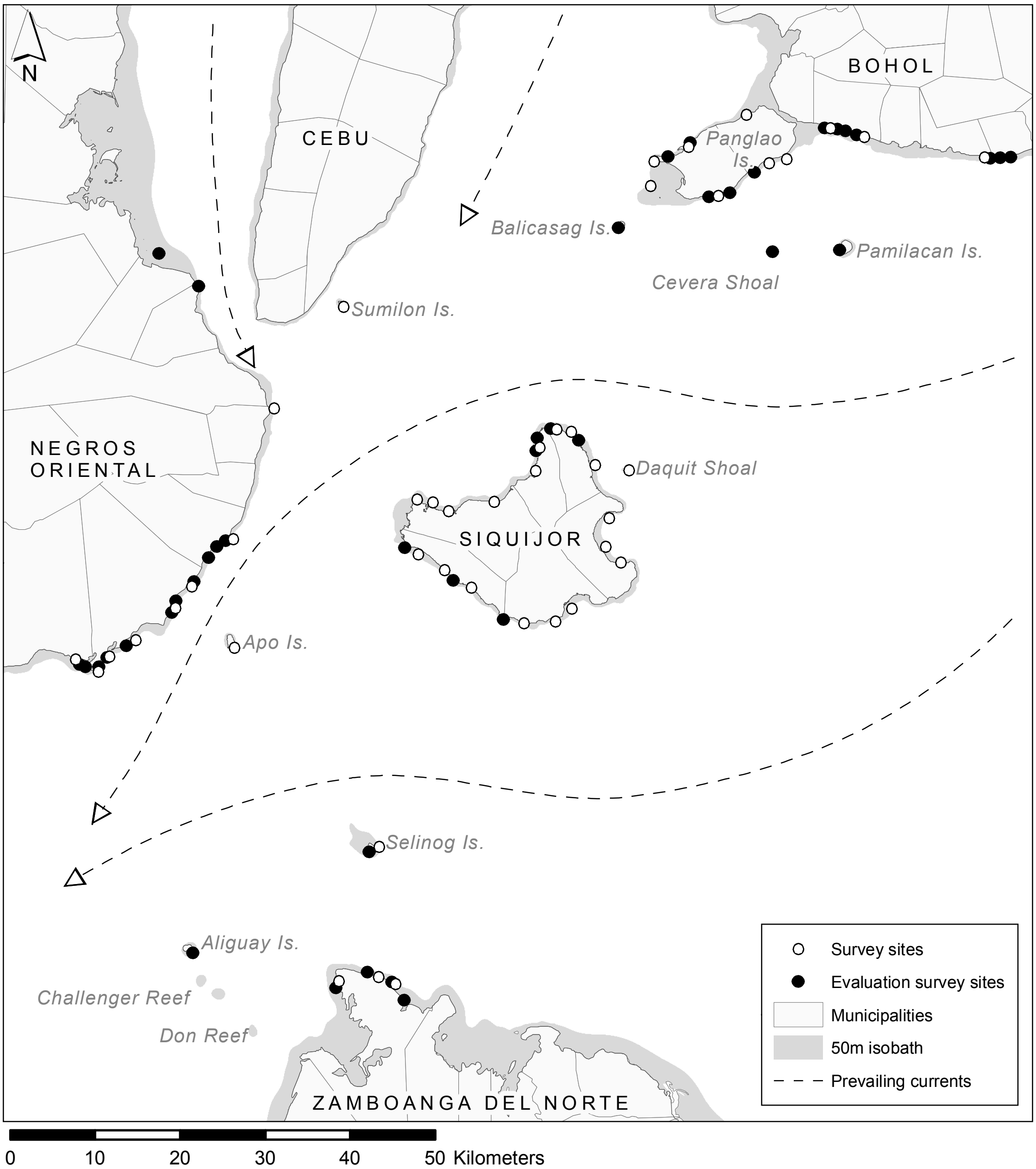


Figure 3.2. The location of reef fish survey sites within the Western Bohol Sea study region. Hollow circles (*n* = 42) indicate the location of survey sites used to create the bioregional classification; filled circles (*n* = 41) indicate the location of survey sites used for evaluation. Administrative provinces are annotated in regular type; labels in italics indicate the locations of small islands and shoals.

Table 3.1. Reef fish species surveyed

Species	Code	Species	Code
Acanthuridae		*Labridae (cont.)*	
Acanthurus japonicus	ACA.JAPO	Hemigymnus melapterus	HEM.MELA
Acanthurus nigricans	ACA.NANS	Labroides bicolor	LAB.BICO
Acanthurus pyroferus	ACA.PYRO	Oxychelinus diagrammus	OXY.DIAG
Acanthurus thomsoni	ACA.THOM	Thalassoma hardwickii	THA.HARD
Ctenochaetus tomiensis	CTE.TOMI	Thalassoma lunare	THA.LUNA
Naso lituratus	NAS.LITU		
Naso unicornis	NAS.UNIC	*Lethrinidae*	
Naso vlamingi	NAS.VLAM	Lethrinus erythracanthus	LET.ERYT
Zebrasoma scopas	ZEB.SCOP	Lethrinus harak	LET.HARA
		Monotaxis grandoculis	MON.GRAN
Chaetodontidae			
Chaetodon adiergastos	CHA.ADIE	*Lutjanidae*	
Chaetodon auriga	CHA.AURI	Lutjanus argentimaculatus	LUT.ARGE
Chaetodon baronessa	CHA.BARO	Lutjanus biguttatus	LUT.BIGU
Chaetodon kleinii	CHA.KLEI	Lutjanus decussatus	LUT.DECU
Chaetodon lineolatus	CHA.LINE	Lutjanus ehrenbergi	LUT.EHRE
Chaetodon lunula	CHA.LUNU	Lutjanus fulvus	LUT.FULV
Chaetodon lunulatus	CHA.LUNS	Lutjanus monostigma	LUT.MONO
Chaetodon melannotus	CHA.MELA	Macolor macularis	MCR.MACU
Chaetodon ocellicaudus	CHA.OCEL		
Chaetodon octofasciatus	CHA.OCTO	*Mullidae*	
Chaetodon ornatissimus	CHA.ORNA	Parupeneus barberinoides	PAR.BNOI
Chaetodon punctatofasciatus	CHA.PUNC	Parupeneus barberinus	PAR.BNUS
Chaetodon rafflesi	CHA.RAFF	Parupeneus bifasciatus	PAR.BIFA
Chaetodon speculum	CHA.SPEC	Parupeneus multifasciatus	PAR.MULT
Chaetodon trifascialis	CHA.TLIS		
Chaetodon unimaculatus	CHA.UNIM	*Nemipteridae*	
Chaetodon vagabundas	CHA.VAGA	Scolopsis bilineatus	SCO.BILI
Labridae		*Pomacanthidae*	
Bodianus axillaris	BOD.AXIL	Apomelichthys trimaculatus	APO.TRIM
Bodianus diana	BOD.DIAN	Centropyge bicolor	CEN.BICO
Bodianus mesothorax	BOD.MESO	Centropyge bispinosus	CEN.BISP
Cirrhilabrus cyanopleura	CIR.CYAN	Centropyge nox	CEN.NOXX
Cirrhilabrus lubbocki	CIR.LUBB	Centropyge tibicen	CEN.TIBI
Epibulus insidiator	EPB.INSI	Centropyge vroliki	CEN.VROL
Gomphosus varius	GOM.VARI	Chaetodontoplus mesoleucus	CPS.MESO
Halichoeres biocellatus	HAL.BIOC	Geniacanthus lamarcki	GEN.LAMA
Halichoeres chrysus	HAL.CHRY	Pomacanthus imperator	PCA.IMPE
Halichoeres hortulans	HAL.HORT	Pomacanthus narvachus	PCA.NARV
Halichoeres melanurus	HAL.MELA	Pomacanthus semicirculatus	PCA.SEMI
Halichoeres richmondi	HAL.RICH	Pygoplites diacanthus	PYG.DIAC
Halichoeres solorensis	HAL.SOLO		

Table 1. ***(continued)*** Reef fish species surveyed

Species	Code	Species	Code
Pomacentridae		*Serranidae*	
Abudefduf lorenzi	ABU.LORE	Cephalopholis argus	CEP.ARGU
Abudefduf vaigiensis	ABU.VAIG	Cephalopholis boenack	CEP.BOEN
Amblyglyphidodon aureus	AMB.AURE	Cephalopholis cyanostigma	CEP.CYAN
Amblyglyphidodon curacao	AMB.CURA	Cephalopholis microprion	CEP.MICR
Amblyglyphidodon leucogaster	AMB.LEUC	Cephalopholis miniata	CEP.MINI
Amphiprion clarkii	AMP.CLAR	Cephalopholis sexmaculatus	CEP.SEXM
Amphiprion frenatus	AMP.FREN	Epinephelus fasciatus	EPI.FASC
Amphiprion perideraion	AMP.PERI	Epinephelus merra	EPI.MERR
Chromis amboinensis	CHR.AMBO	Pseudanthias huchti	PSE.HUTC
Chromis retrofasciata	CHR.RETR	Pseudanthias tuka	PSE.TUKA
Chromis scotochiloptera	CHR.SCOT	Variola louti	VAR.LOUT
Chromis viridis	CHR.VIRI		
Dascyllus aruanus	DAS.ARUA	*Siganidae*	
Dascyllus melanurus	DAS.MELA	Siganus corallinus	SIG.CORA
Dascyllus reticulatus	DAS.RETI	Siganus guttatus	SIG.GUTT
Dascyllus trimaculatus	DAS.TRIM	Siganus puellus	SIG.PUEL
Dischistodus melanotus	DIS.MELA	Siganus unimaculatus	SIG.UNIM
Plectroglyphidodon dickii	PGY.DICK	Siganus virgatus	SIG.VIRG
Plectroglyphidodon lacrymatus	PGY.LACR	Siganus vulpinus	SIG.VULP
Pomacentrus amboinensis	POM.AMBO		
Pomacentrus brachialis	POM.BRAC	*Zanclidae*	
Pomacentrus coelestis	POM.COEL	Zanclus cornutus	ZAN.CORN
Pomacentrus moluccensis	POM.MOLU		

Siquijor. A different observer (B. Stockwell) recorded the abundance of all non-cryptic, diurnally active reef fish species along a 50 m x 10 m transect. Six transects were surveyed at each site, three on the reef slope (10 – 12 m depth) and three on the reef flat (2 – 3 m depth) (Stockwell et al. 2009). Data were pooled at site-level, and fourth-root transformed prior to analysis. These data were collected as part of a project monitoring the effectiveness of no-take MPAs in the region; accordingly, several surveys were comparisons of species' abundance inside and outside MPAs. To control for the effect of no-take protection on fish community composition, data for MPA and control site pairs were pooled prior to analysis.

3.2.2. Multivariate Analyses

Anderson and Willis (2003) suggest that multivariate analysis of ecological data should include four components: a robust unconstrained ordination; a constrained ordination by reference to a specific hypothesis; a rigorous statistical test of the hypothesis; and characterisation of the variables responsible for multivariate patterns. I applied this framework to develop bioregional classifications based on patterns in reef fish community composition among survey sites in the Western Bohol Sea, and separately for the island of Siquijor. Undertaking a separate classification for Siquijor allowed me to identify bioregions at a finer spatial resolution, which was required to explore options for designing ecologically representative MPA networks for the Province (see Chapters 5 and 6).

For the Western Bohol Sea bioregionalisation, multivariate analysis was performed on log4 abundance data for reef fish from a single dataset. For Siquijor, to increase the number of survey sites available and thus improve the resolution of the classification, I used surveys from both datasets (Fig. 3.3). Due to the different survey techniques employed to collect the data (the first observer employed a timed swim method, the second, replicate 50 m transects), abundance data were converted to presence-absence prior to analysis. In all other aspects the statistical methods for the two classifications were the same. Unless otherwise stated, all statistical analyses were performed with the software package '*R*' (R Development Core Team 2008).

Similarities in reef fish community composition between sites were initially examined using complete linkage hierarchical cluster analysis based on Bray-Curtis dissimilarities. Similarity profile permutation (SIMPROF) tests (Clarke et al. 2008) were performed to identify significant clusters of sites at the 99 % significance level. The resulting bioregional classification was evaluated with an unconstrained analysis of similarities (ANOSIM) test.

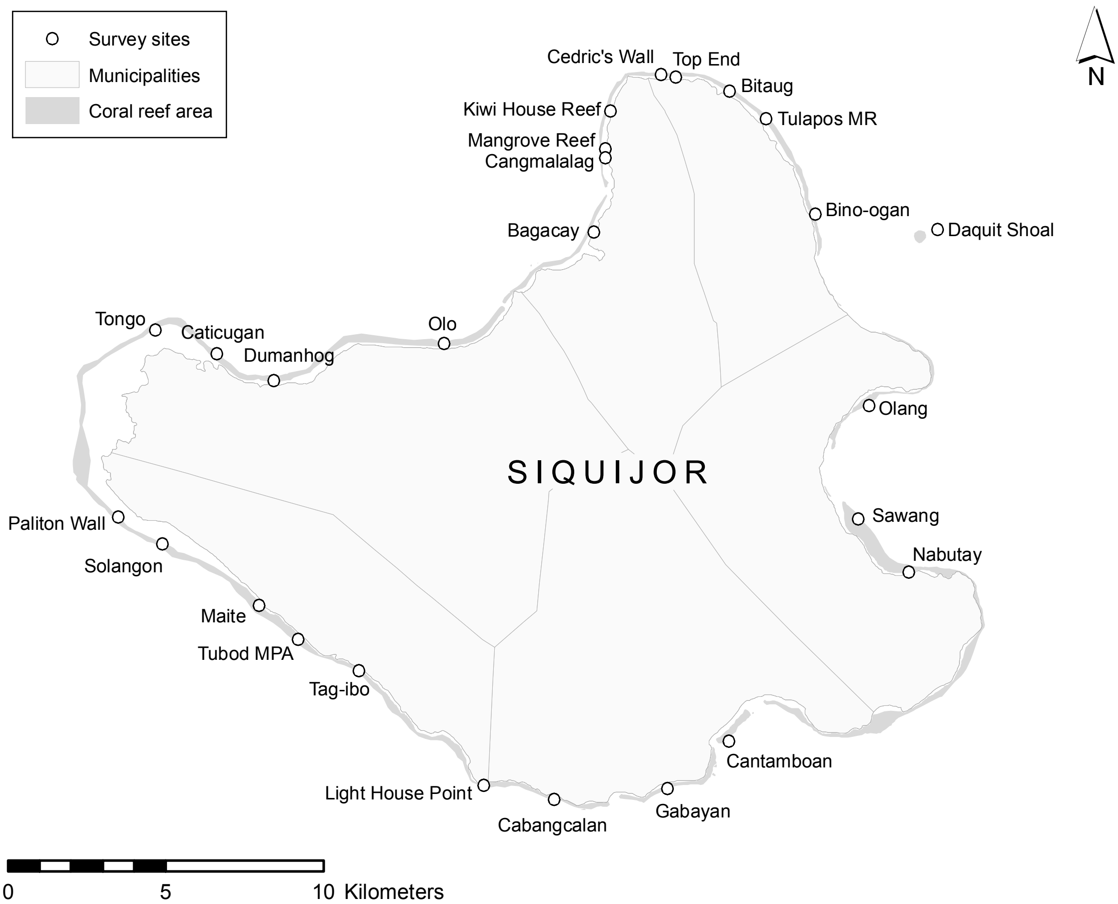


Figure 3.3. The location of 26 reef fish survey sites around Siquijor Island.

An unconstrained ordination was performed using non-metric multidimensional scaling (nMDS). Non-metric MDS provides a graphic depiction in two dimensions of similarities between sites: sites in close proximity to one another in the ordination space share similar species composition, whereas those farther apart are less similar. The goodness of fit of the ordination is measured by the stress index, which indicates how well the ordination summarises the observed distances among sites. Stress values range from 0 to 1: values below 0.05 indicate a good fit, those greater than 0.2 are considered to be a poor fit (Kruskal 1964). Stress values were plotted against dimensionality to identify the most appropriate configuration for analysis. Non-metric MDS was performed using the *metaMDS* function in the *Vegan* package for *R* (Oksanen et al. 2008).

Although nMDS has been demonstrated to be a robust unconstrained ordination procedure for ecology (Minchin 1987), high variability and correlation among unrelated variables can obscure ecologically important patterns in any unconstrained ordination procedure (Anderson & Willis 2003). For this reason, I further analysed the site groupings using canonical analysis of principal coordinates (CAP), a constrained ordination procedure that initially calculates unconstrained principal coordinate (PCO) axes, followed by canonical discriminant analysis on the principal coordinates to maximise separation between groups (Anderson & Willis 2003). A limitation of many constrained ordination techniques is that they are limited to a particular distance measure; CAP produces a constrained ordination on the basis of any dissimilarity matrix, including Bray-Curtis dissimilarities, as I have used here. The CAP procedure provides misclassification errors using a 'leave one out' method, whereby each observation is removed from the analysis and then placed in the canonical space determined by the rest of the observations (Anderson & Willis 2003). The percentage of correct classifications provides a measure of the goodness of fit. Finally, species' correlations with the first two CAP axes were plotted to identify those that had greatest influence on site classifications. CAP analysis was performed using the *CAPdiscrim* function in the *BiodiversityR* package for *R* (Kindt & Coe 2005).

3.2.3. Bioregional Classification

Results from the multivariate analyses were interpreted and refined to produce bioregional classifications that would be informative for conservation planning. In arriving at a final classification scheme, I applied the following principles: (1) bioregions should be spatially contiguous; (2) bioregions should be informed by biogeographic, rather than human or political subdivisions; (3) bioregions should be at an appropriate spatial scale for conservation planning and, without specifying maximum or minimum sizes, be of approximately equal size; and (4) bioregional delineation should be guided by parsimony. Boundaries between bioregions were approximated as the midpoint along the coastline between survey sites classified into different bioregions (Shears et al. 2008).

3.2.4. Evaluation of the Western Bohol Sea bioregionalisation

I evaluated the bioregional classification for the Western Bohol Sea region using the second, independently collected dataset. The evaluation survey sites were classified into bioregions based on their geographic location. This classification was then used as an *a priori* hypothesis for CAP analysis, and the percentage of correct classifications used to evaluate how well these data support the bioregionalisation. A similar appraisal of the Siquijor bioregionalisation was not possible, due to the low number of survey sites from the evaluation dataset in this region (n = 6); these data were instead used to inform the delineation of bioregions for Siquijor.

3.3. Results

3.3.1. Western Bohol Sea

Hierarchical cluster analysis with SIMPROF identified 11 significant site clusters, with four sites not placed in any group. For further analysis, the cluster dendrogram was cut at the point where all clusters contained a minimum of two sites (Bray-Curtis dissimilarity = 0.33), resulting in nine clusters (Fig. 3.4). These groups can be visually discerned on a non-metric MDS plot (Fig. 3.5a). Although the ordination fit is relatively poor (stress = 0.17), an ANOSIM test confirmed that differences between the groups were significant ($R = 0.7858$, $p < 0.001$). Furthermore, the site clusters correspond reasonably well with their geographic location (Fig. 3.4).

CAP analysis provided further support for the site classification: groups were more distinctly separated in the ordination space in the CAP analysis than that provided by the nMDS axes (Fig. 3.5b), and the percentage of correct classifications using the leave-one-out procedure was high (81 %, $p < 0.001$). The first seven PCO axes explained 71.33% of the variability in the dissimilarity matrix. Species' correlations with the first two CAP axes did not provide much insight as to the role of individual species in defining site groupings (Fig. 3.5c). The majority of species showed strong negative correlations with the first axis, suggesting that species richness may have been important in distinguishing the regions.

Based on the results of the multivariate analyses, and subjective knowledge of habitat types, reef types and oceanographic currents in the region, I identified six 'reef fish bioregions' within the Western Bohol Sea (Fig. 3.6).

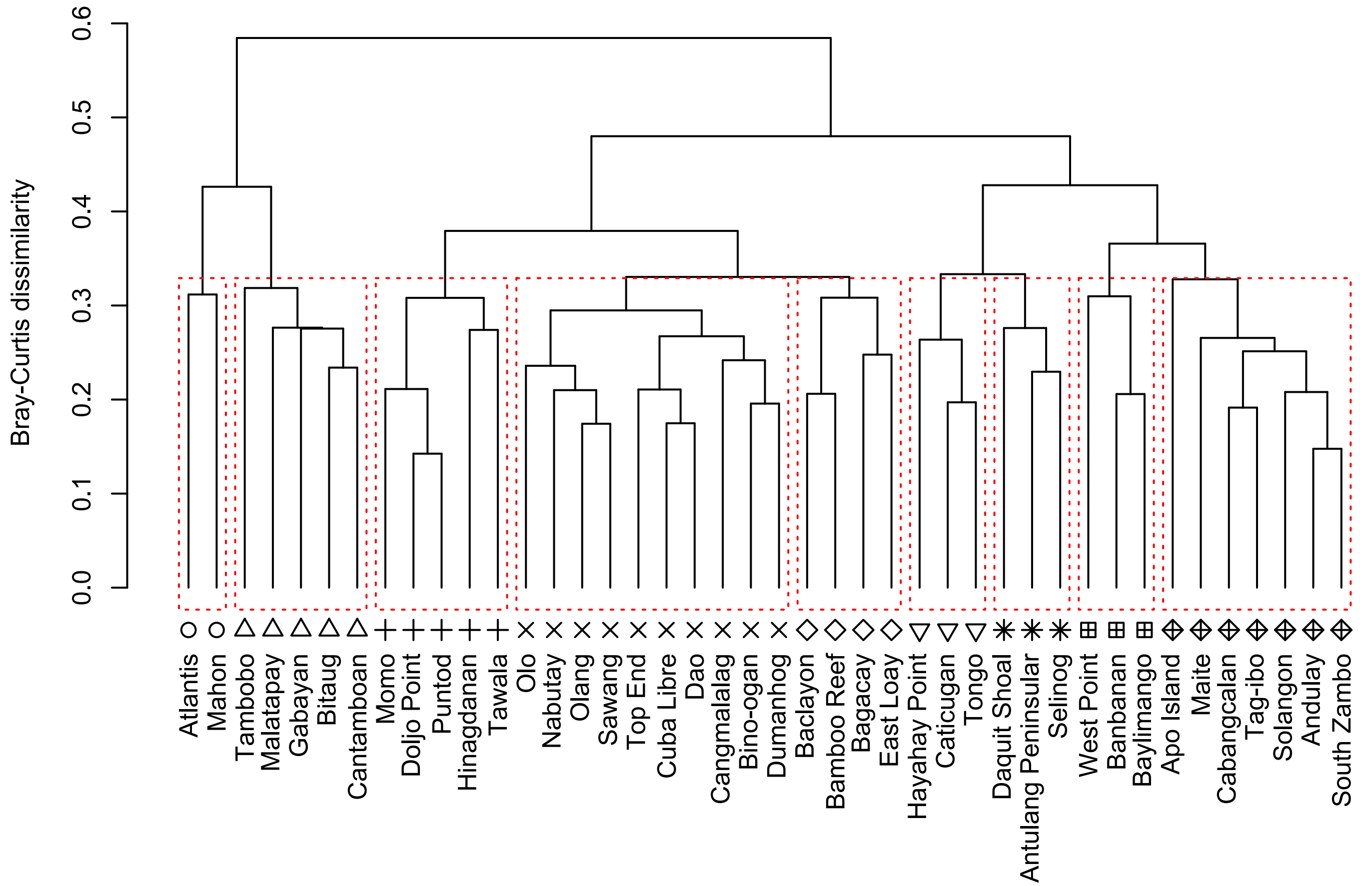
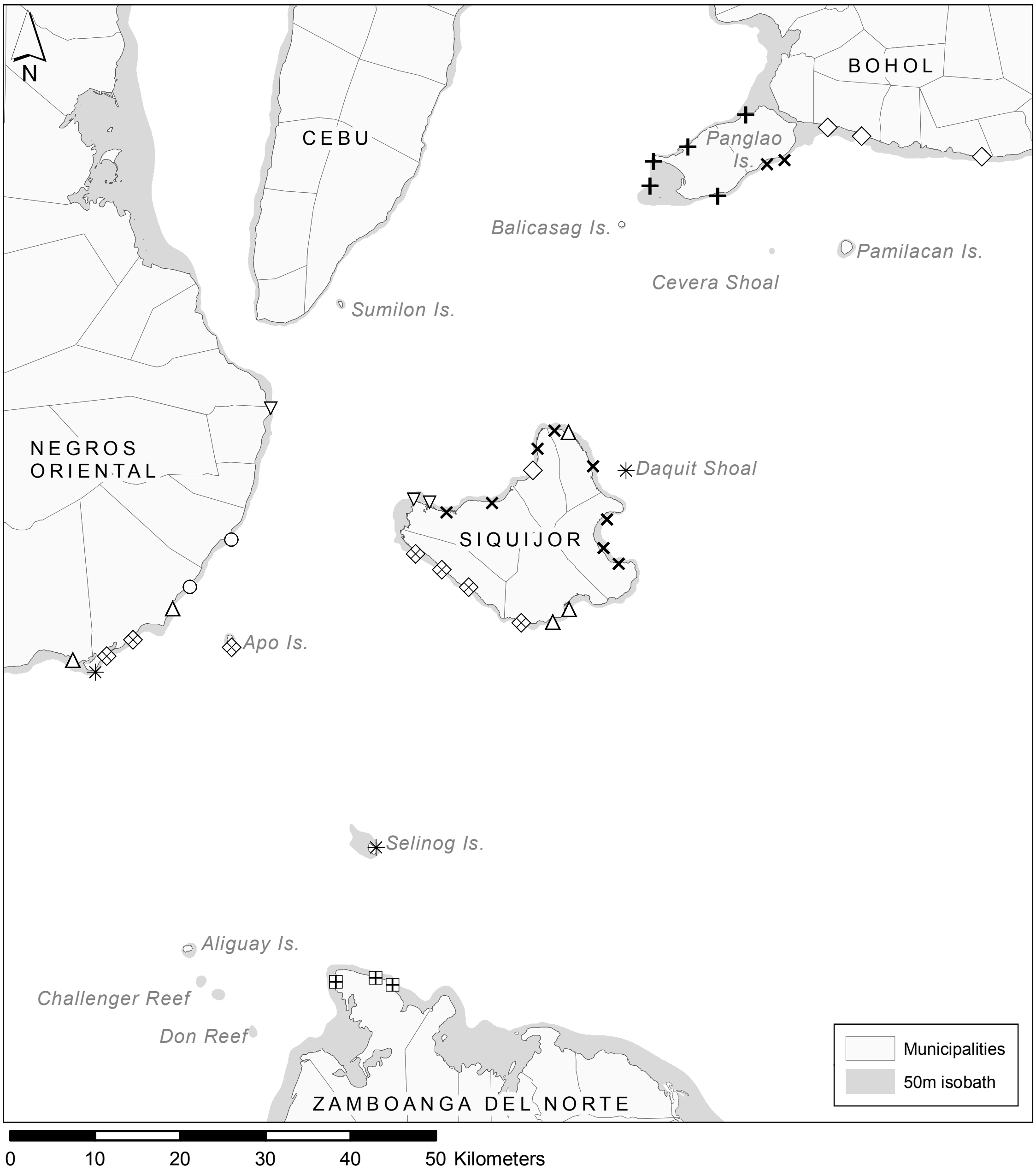


Figure 3.4. Results of hierarchical cluster analysis of sites based on Bray-Curtis dissimilarities of log4 abundance data of reef fish species. Significant clusters are indicated by dashed rectangles on the dendrogram, and by location symbols on the study region map (following page).


N
CEBU
BOHOL
Panglao Is.
Balicasag Is.
Pamilacan Is.
Cevera Shoal
Sumilon Is.
NEGROS ORIENTAL
Daquit Shoal
SIQUIJOR
Apo Is.
Selinog Is.
Aliguay Is.
Challenger Reef
Don Reef
ZAMBOANGA DEL NORTE
Municipalities
50m isobath
0 10 20 30 40 50 Kilometers

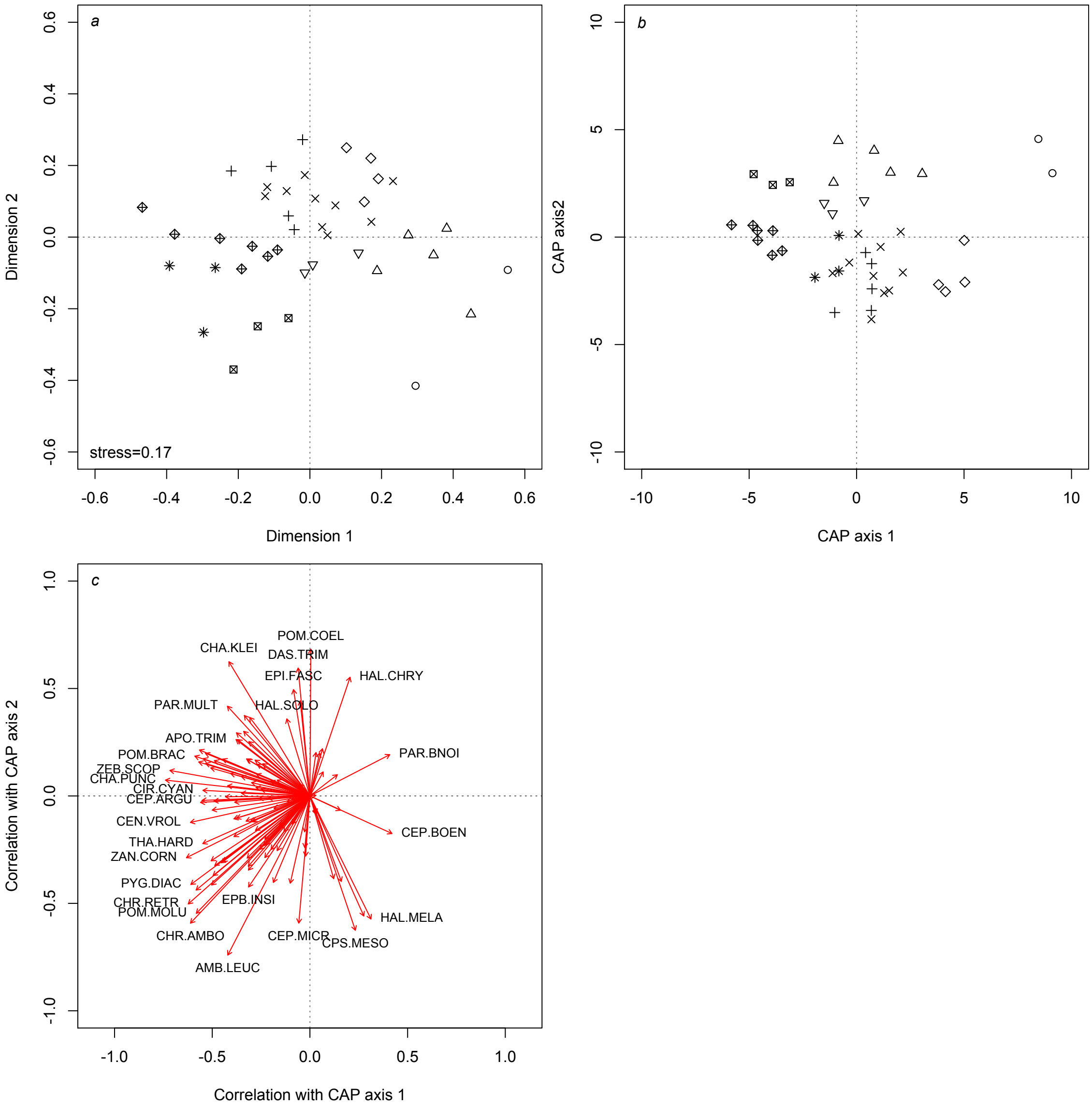


Figure 3.5. (a) Non-metric and (b) CAP ordination plots of reef fish assemblages at survey sites in the Western Bohol Sea region. Plot symbols indicate significant site clusters (see the cluster dendrogram and location map in Fig. 3.4). Species' correlations with the first two CAP axes are shown in (c); see Table 3.1 for species codes.

Figure 3.6. Proposed bioregions for the Western Bohol Sea, on the basis of reef fish community assemblages.

Sites in Zamboanga del Norte, including the islands of Selinog and Aliguay, were grouped with those in the far south of Negros Oriental. The reef fish community composition at sites within this bioregion is likely influenced by their proximity to the Sulu Sea. Survey sites in the municipality of Dauin (directly inshore from Apo Island – see Fig. 3.6) were differentiated from others on the Negros Coastline (Fig. 3.4). This municipality has 10 no-take MPAs, which likely have a significant influence on reef fish community structure in the surrounding area. Consequently, I did not recognise this as a bioregional effect, and included these sites in the South-western Siquijor bioregion (Fig. 3.6).

Unfortunately, no data were available to classify the coastal fringing reefs in Southern Cebu. Presence-absence data were available from a preliminary survey undertaken at Sumilon Island (Figs. 3.2, 3.6). Hierarchical cluster analysis based on presence-absence data placed this site in a cluster with Daquit Shoal and Selinog Island. Similarities in fish community compositions between these sites likely reflect their similar habitats (Sumilon and Selinog are both offshore coralline islands, Daquit Shoal is an offshore reef rising to a depth of 5m at its shallowest point, Fig. 3.6), and thus do not provide information to classify the coastal fringing reefs on Cebu. I ultimately grouped the Southeast coast of Cebu with the sites on the Western side of Panglao Island, as these sites may have similar community composition as a result of influence from the Cebu Strait. Likewise, the Western coast of Cebu was classified in the Tañon Bioregion due to probable influence of the strong current from the Tañon Strait. This bioregion also includes the northern section of Negros Oriental and the Northwest tip of Siquijor (Fig. 3.6).

The Northeast coast of Siquijor showed similar species composition to sites on the East of Panglao Island (Fig. 3.6). This bioregion also encompasses Daquit and Cevera Shoals and Pamilacan Island. Again, similarities between Daquit Shoal and other small, offshore coralline islands were interpreted as a habitat, rather than bioregional, effect. The three survey sites along the southern coast of Bohol were distinct from those on Panglao, and so formed a separate bioregion of southern Bohol (Fig. 3.6).

To provide a test of the goodness of fit of the bioregional classification, survey sites from a second dataset, collected independently (Fig. 3.2), were assigned to bioregions based on their geographic locations. The percentage of correct classifications from a CAP analysis conducted on reef fish data collected at these locations was 83 % ($p < 0.001$), indicating strong support for the classification. An ANOSIM test also confirmed that these groupings were significant (ANOSIM $R = 0.4959$, $p < 0.001$).

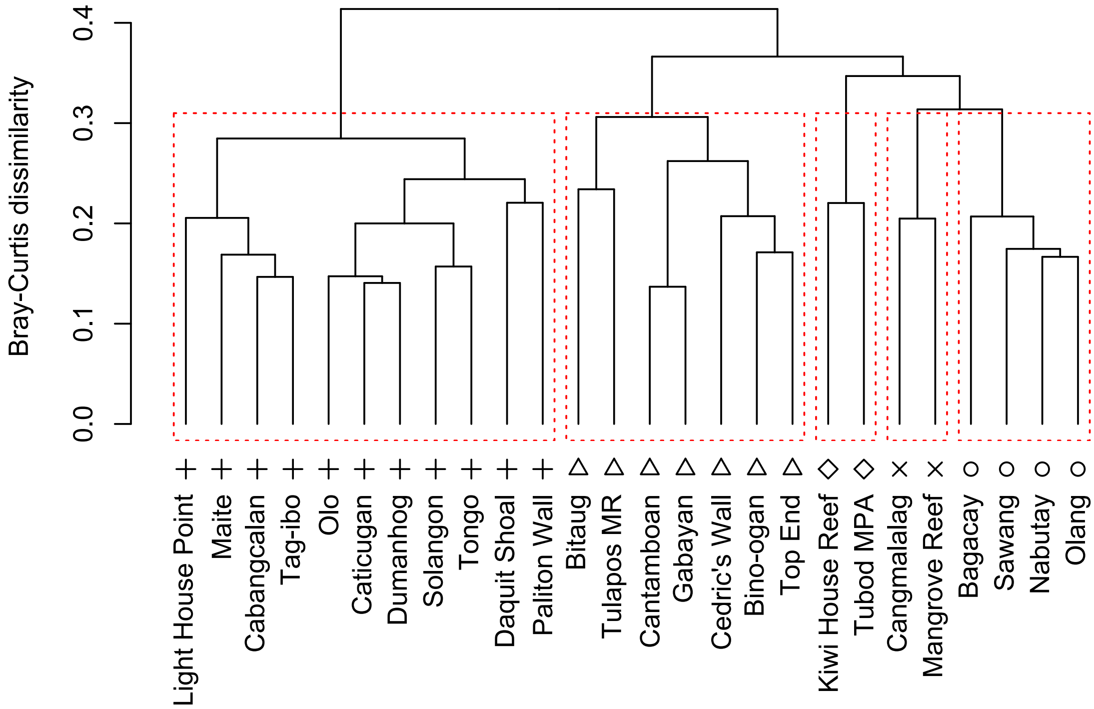


Figure 3.7. Cluster dendrogram of reef fish species assemblages surveyed at sites around Siquijor. Clustering was hierarchical using complete linkages and based on Bray-Curtis dissimilarities. Dashed boxes and site symbols indicate five significant clusters identified by SIMPROF tests.

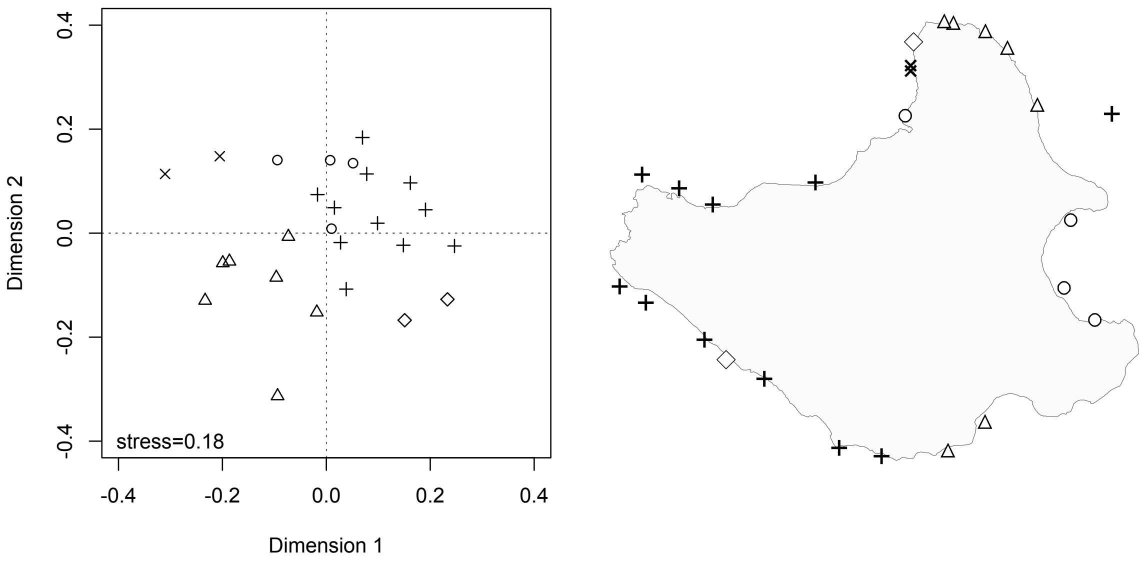
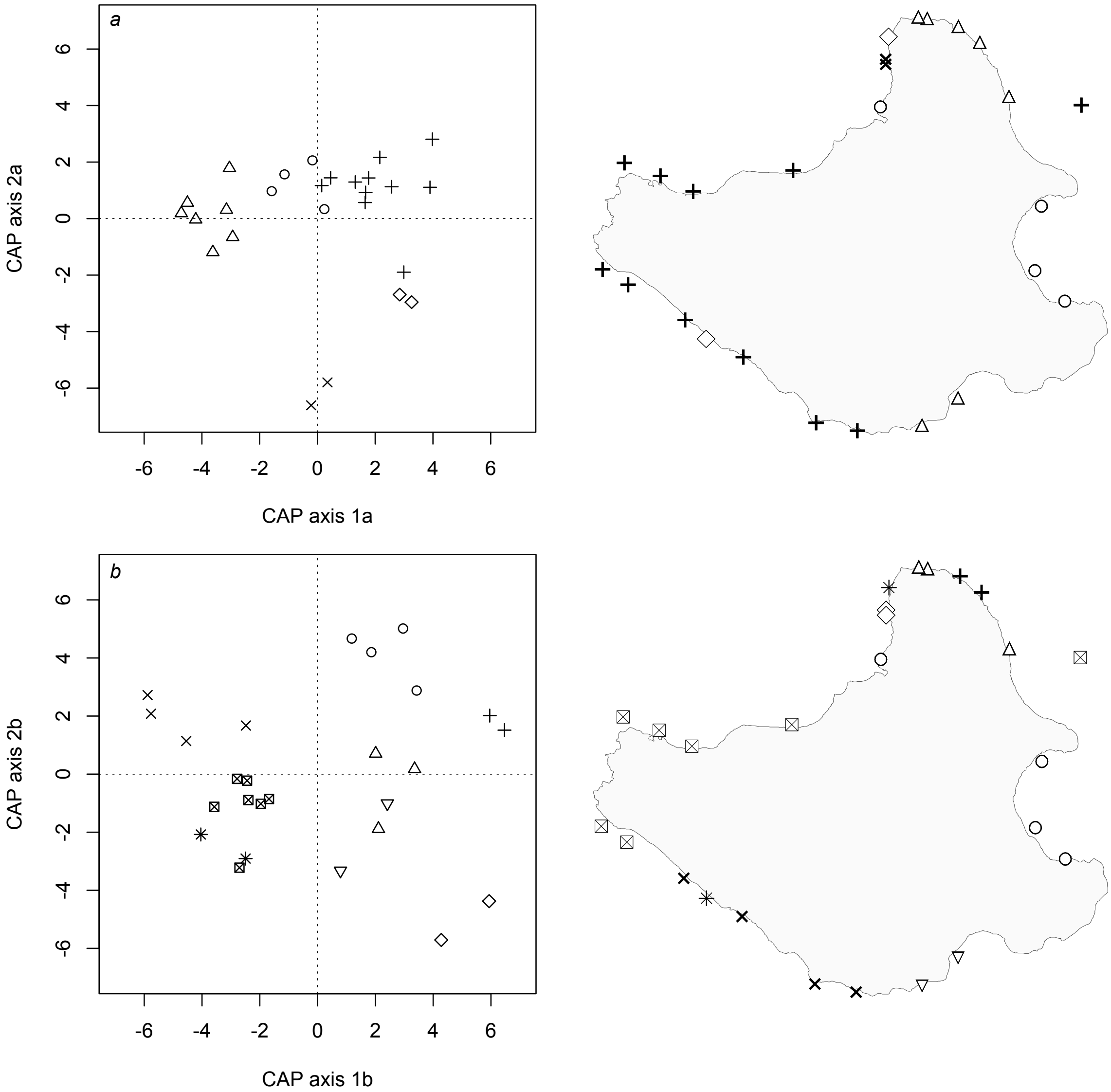


Figure 3.8. Non-metric MDS ordination plot of reef fish assemblages around Siquijor. Location symbols represent significant site clusters identified by SIMPROF analysis, which are overlaid on the study region to the right of the ordination.



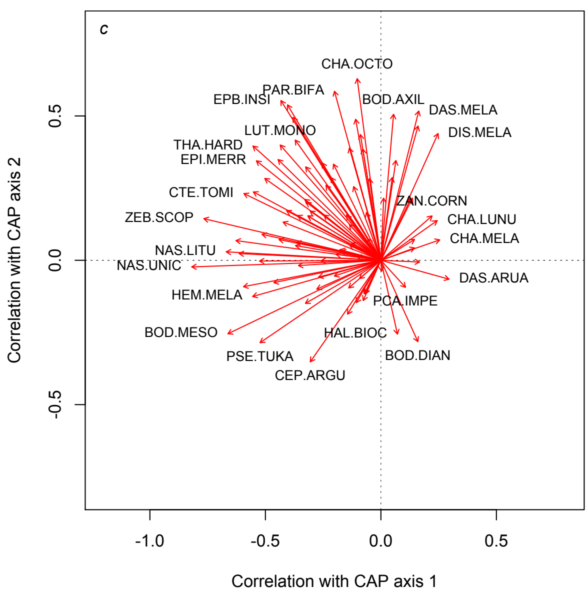


Figure 3.9. Canonical analysis of principal coordinates (CAP) plots of presence-absence data for reef fishes from survey sites around Siquijor. Plot symbols indicate (a) significant site clusters identified by SIMPROF analysis, and (b) further subdivisions of the cluster dendrogram which were found to be significant using ANOSIM and CAP tests. Location symbols are overlaid on the study region to the right of the ordinations. Species' correlations with the CAP axes are shown in (c); see Table 3.1 for species codes.

3.3.2. Siquijor

Hierarchical cluster analysis with SIMPROF identified five significant clusters of sites (Fig. 3.7). The largest cluster featured all but one of the sites in the West and Southwest of the island, with the addition of the offshore Daquit Shoal, which lies off the Northeast coast. Sites on the Northeast coast (with the exception of Daquit Shoal) were grouped with two sites in the Southern area of Lazi Bay. The three sites in Maria Bay formed a cluster with Bagacay, on the Northwest coast in Larena (Fig. 3.7). This group was most closely related to a distinct cluster comprising the two other sites in Larena. The remaining two sites, Tubod MPA and Kiwi House Reef (Figs. 3.3, 3.7) formed a somewhat anomalous cluster, in that they are located on opposite sides of the island.

These groups can be visualised on a non-metric MDS plot (Fig. 3.8). Again, although the ordination fit is relatively poor (stress = 0.18), an ANOSIM test confirmed that differences between the groups were significant ($R = 0.6568$, $p < 0.001$). CAP analysis provided further support for the site classification: a plot of the first two canonical axes shows a clear pattern of differences between the groups (Fig 3.9a), and 85 % of observations were classified correctly using the leave-one-out procedure ($p < 0.001$). The first ten PCO axes explained 98.17% of the variability in the dissimilarity matrix.

Although the SIMPROF test only identified five significant clusters, further subdivision of the cluster dendrogram into eight groups was supported by the geographical location of sites around the island (Fig. 3.9b). The large Western group is separated into sites in the Northwest (plus Daquit Shoal) and those in the Southwest, and the two sites in Lazi Bay are differentiated from those on the Northeast coast of Enrique Villanueva (Figs. 3.3, 3.9b). These groupings were found to be significant when tested using ANOSIM ($R = 0.7928$, $p < 0.001$) and CAP (77 % classification success, $p < 0.001$).

Species' correlations with the first two CAP axes are shown in Figure 3.9c. The majority of species are located in the upper left quadrant of the ordination, indicating that species richness was an important factor in site clustering. However, there is evidence that individual species also influenced the site clusters: *Dascyllus melanurus* was observed exclusively at sites within Maria Bay, with *Bodianus axillaris* and *Dischistodus melanotus* each observed at only a single site outside this cluster. These species and sites all showed positive correlations with both axes, appearing in the upper right quadrant of the ordination.

In the final bioregionalisation for Siquijor (Fig. 3.10) I chose to incorporate the subdivision of the Western sites to create two bioregions (Western Siquijor and Southern), as this

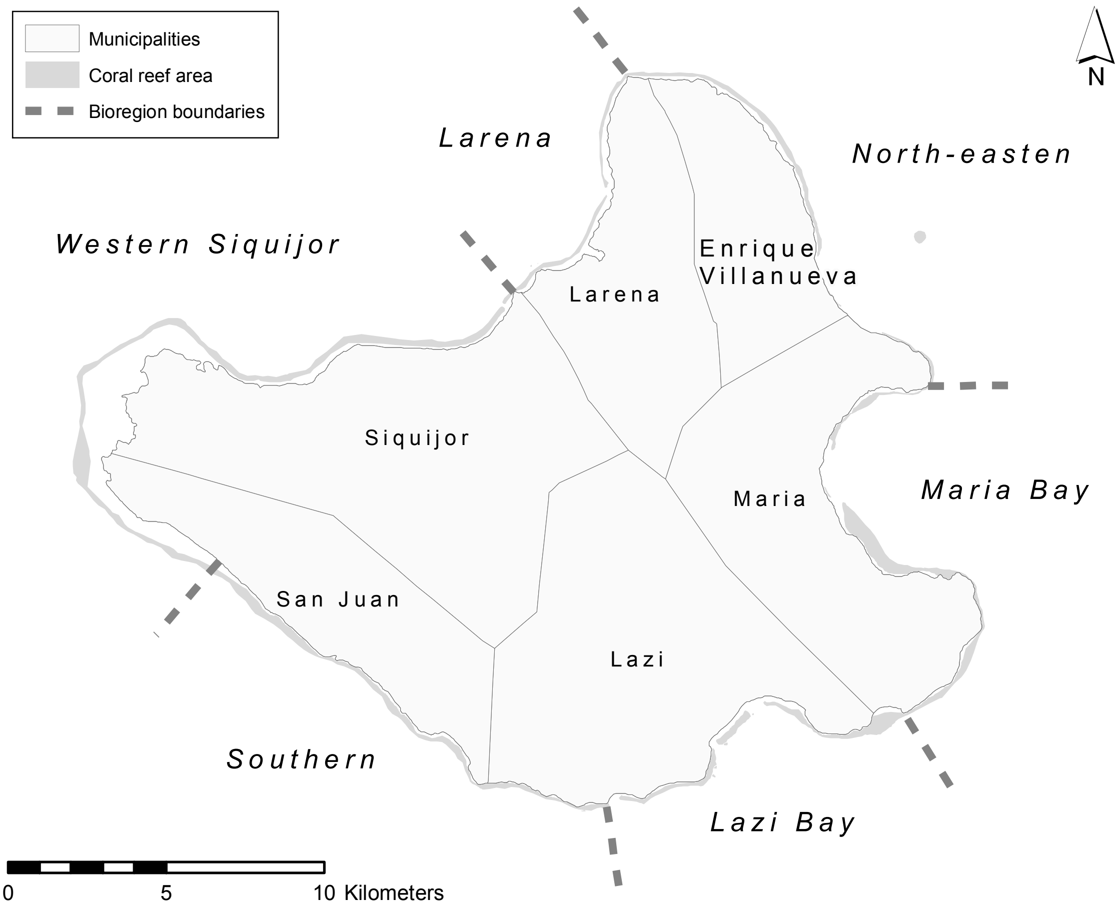


Figure 3.10. Proposed bioregions for Siquijor, on the basis of groupings of locations with similar reef fish species assemblages.

boundary is consistent with a change in reef structure, from a steep slope preceded by wide seagrass beds (Western) to a more gradual fringing reef (Southern). I did not subdivide the Northeastern bioregion, as this division would have created either discontinuous or excessively small bioregions. I included Daquit Shoal in the Northeastern bioregion, as similarities in the reef fish assemblage between this site and those in the Western bioregion are likely a result of similar reef structure, rather than a biogeographic effect. Finally, I disregarded the anomalous grouping of Kiwi House Reef and Tubod MPA (see Fig. 3.3) altogether.

3.4. Discussion

Multivariate analyses highlighted significant differences in reef fish community composition at sites within the Western Bohol Sea study region and around Siquijor island. These differences enabled the delineation of bioregions at a finer spatial resolution than that provided by previous schemes, allowing application to conservation planning problems. Given that the analytical techniques applied here are typically used to identify bioregions at the mesoscale (hundreds of kilometres), the results show remarkable correlation with the geographic locations of sites. Furthermore, evaluation with an independent data set collected using a different survey technique and focal species indicated that the proposed bioregions for the Western Bohol Sea accurately depict variation in reef fish community composition.

The multivariate analysis of reef fish data provided an objective basis for biogeographic classification, and a means to statistically evaluate the reliability of the resulting bioregions using an independent dataset. Nevertheless, the final bioregionalisations were ultimately the result of my subjective interpretation of the statistical results, and are dependent on the quality of the underlying data. Bioregions were identified using a single taxonomic group: reef fishes. Reef fish species have relatively restricted geographic distributions compared to other marine taxa (e.g. corals), and are widely accepted as the most appropriate taxonomic surrogates for marine conservation planning in the absence of more complete data (Roberts et al. 2002; Beger et al. 2003; Mumby et al. 2008). Although it might be expected that data for different taxa would result in different biogeographic classifications, Shears et al (2008) found high concordance between marine classifications for New Zealand developed using different species groups. Evaluation with a different set of reef fish data showed the Western Bohol Sea bioregions to be robust to variation in survey methodology. However, data were not available to assess generality for different taxonomic groups.

The number of bioregions identified and the locations of boundaries between them are likely to be influenced by the spatial distribution and number of sampling locations. The data I used to develop the bioregionalisations were collected at randomly selected sites, and are therefore not subject to typical spatial biases (Grand et al. 2007). Nevertheless, logistical constraints meant that no data were collected for Southern Cebu, and few surveys were undertaken close to Dumaguete City. Additional surveys in these regions may have resulted in a different bioregional classification for the Western Bohol Sea. Given that this section of the study region was also poorly sampled in the evaluation dataset, it is difficult to assess how much impact the reduced survey intensity in this area had on the accuracy of the classification. The distribution of survey sites around Siquijor was more comprehensive and, although no independent data were available to assess the accuracy, likely resulted in a rigorous classification.

In common with other marine biogeographic classifications (Spalding et al. 2007), the bioregions identified here focus on inshore reef ecosystems only, and do not attempt to classify pelagic or deep-water communities. This bias is appropriate, given the intended application to MPA network design: to date, MPAs in the Philippines have exclusively been established to protect reef-associated habitats (Weeks et al. 2010a, Chapter 2). Although the bioregionalisation was based entirely on biological data, with no consideration of human divisions of the land- or seascape, the bioregions for Siquijor correspond surprisingly well with municipal boundaries (Fig. 3.10). This is fortuitous for conservation planning, given that coastal resource management is typically undertaken within the context of political boundaries rather than ecological ones, and MPA designation is undertaken at municipal level in the Philippines.

The bioregions proposed here provide biogeographic context within which to undertake systematic conservation planning for the Western Bohol Sea region, that can be applied to assess the extent to which existing MPAs represent biodiversity (see Chapter 4) and will facilitate the design of ecologically representative MPA networks on Siquijor (Chapters 5 and 6). Nevertheless, these systems should not be considered definitive. Biogeographic boundaries in marine environments are characteristically indistinct and dynamic, and the boundaries between bioregions may change as more data become available, or as spatial patterns in biodiversity change over time.

4. How well does an *ad hoc* system of community-based MPAs fulfil local- and regional-scale objectives for fisheries management, biodiversity conservation and socioeconomic development?

4.1. Introduction

Systematic conservation planning is the process of designing, implementing, and managing protected area networks to achieve explicit objectives for biodiversity conservation (Margules & Pressey 2000). Initially developed in response to the recognition that many protected areas constitute 'residual' conservation (Pressey & Bottrill 2008), systematic conservation planning has since evolved into a comprehensive, science-based framework (Pressey & Bottrill 2009) that has been applied in terrestrial (Cowling & Pressey 2003), freshwater (Nel et al. 2009) and marine (Leslie 2005) systems. Although theoretical studies still outnumber practical applications, regional-scale planning increasingly underpins the conservation strategies of NGOs and governments worldwide (e.g. Groves et al. 2002; Pressey & Bottrill 2009). A systematic conservation planning approach undertaken on the Great Barrier Reef (GBR, Australia) developed existing MPAs, covering 4.5% of the GBR, into a comprehensive, adequate, and representative MPA network that protects 33% of the GBR within no-take zones (Day et al. 2002; Fernandes et al. 2005; McCook et al. 2010). Similar approaches have been applied to design MPA networks in California (Airame et al. 2003; Gleason et al. 2010), the UK (Smith et al. 2009), Papua New Guinea (Green et al. 2009) and elsewhere.

Despite recent interest in systematic approaches to MPA network design, most existing MPAs have not been located on the basis of scientific criteria (Pressey et al. 1993; Roberts 2000; Agardy 2005). In the Philippines, systematic conservation planning has had little influence on coastal resource management thus far. For the most part, MPAs have been established through 'bottom-up', community-based approaches, whereby local communities are the primary decision makers responsible for MPA implementation and management (Alcala 1988; Christie & White 1997; Alcala 1998; White et al. 2002; Alcala & Russ 2006). Similar approaches are employed throughout the Pacific (King & Faasili 1999; Johannes 2002; Aswani 2007; Govan et al. 2009), often in the form of customary management (Cinner & Aswani 2007). The locations of community-based MPAs in the Philippines are typically guided by local community objectives: sites that fishers are willing to give up and that might create opportunities for tourism-related income (Walmsley & White 2003; Alcala & Russ 2006). Although it has been demonstrated that these MPAs can achieve local-scale fisheries objectives (e.g. Alcala & Russ 2006), at present it is clear that they do not constitute a comprehensive national MPA network (Alino et al. 2000; Weeks et al. 2010a, Chapter 2). Yet a system of comprehensive, ecologically representative, connected and resilient MPAs and MPA networks is required not only to conserve the Philippines' rich marine biodiversity (Carpenter & Springer 2005), but also to safeguard the livelihoods and food security of coastal communities (Coral Triangle Initiative 2008).

Whilst the need to rapidly and significantly scale-up existing MPA systems is clear, identifying the most effective approach to accomplish this goal is not straightforward (Christie & White 2007). Although systematic conservation planning appears to offer an ideal approach to develop regional-scale MPA networks, applying this framework in the context of the Philippines will not be easy (Christie et al. 2009; Mills et al. 2010; Weeks et al. 2010a, Chapter 2). The success of many community-based MPAs, despite their opportunistic design (Roberts 2000), is widely attributed to their ability to achieve strong stakeholder support and compliance: a result of extensive local community participation in planning and management (Alcala 1988; Pomeroy et al. 1997; Alcala 1998; White & Vogt 2000; Pollnac et al. 2001; Walmsley & White 2003; Crawford 2004; Granek & Brown 2005; Alcala & Russ 2006; Samoilys et al. 2007). This is in contrast to regional-scale planning, which is inherently a 'top-down' approach (Rodriguez et al. 2007; Abrams et al. 2009). Failure to adequately incorporate socioeconomic factors in the design and implementation of MPA networks is a frequent criticism of systematic conservation planning initiatives (Knight & Cowling 2007), and can lead to poor management effectiveness or MPAs that exist on paper only (Christie & White 1997; Alcala & Russ 2006; Polasky 2008; Green et al. 2009). Generating local community ownership of and support for regional-scale plans is an important problem that still needs to be resolved (Rodriguez et al. 2007).

Regional-scale planning typically requires more time, money, data and expertise than community-based approaches (Didier et al. 2009). As a minimum, systematic MPA network design requires data on the spatial distribution of targeted biodiversity features and socioeconomic costs throughout the planning region (Williams et al. 2002; Ban & Klein 2009). If these data are lacking, or are of insufficient spatial resolution for decision making, as is often the case in many tropical developing countries (Ban et al. 2009a), time and funds may need to be allocated to data collection. In addition to community liaison officers, who will be required to ensure effective consultation with stakeholders, systematic planning also requires scientific advisors and technical staff to oversee the use of GIS systems and decision support tools. The Nature Conservancy estimated that the total cost of designing a scientific MPA network for Kimbe Bay, Papua New Guinea was c. USD $400,000, including scientific research, staff and workshops, but excluding community engagement and implementation (Green et al. 2009).

Although these challenges are not insurmountable, it is nevertheless appropriate to consider whether the benefits of systematic conservation planning outweigh the costs. The aim of this chapter therefore, is to determine whether a systematic approach to MPA network design is required to develop ecologically functional MPA networks in the Philippines, or whether community-based efforts can achieve the same objectives. If the latter is true, international funding programs such as the Coral Triangle Initiative on Coral Reefs, Fisheries and Food Security (CTI, Coral Triangle Initiative 2008) may do better to support grassroots conservation efforts to expand MPA networks, rather than to invest in regional-scale planning.

The Central Visayas has the highest concentration of MPAs in the Philippines (Weeks et al. 2010a, Chapter 2): a result of a long history of community-based management (Alcala 1998; Alcala & Russ 2006) and the continued efforts of NGOs and academic institutions that facilitate coastal resource management initiatives in the region (White et al. 2006b; Eisma-Osorio et al. 2009). This region represents an optimistic scenario for community-based MPA establishment in its current form, and therefore an ideal system to assess the potential of such approaches to achieve both local- and regional-scale objectives for fisheries management, biodiversity conservation and socioeconomic development. MPAs in the Western Bohol Sea region of the Central Visayas have been the focus of a disproportionately large body of scientific literature on community-based management and the ecological effects of no-take MPAs (references in Alcala & Russ 2006, and others). I first critically review this body of research, to assess the extent to which these MPAs achieve local-scale objectives for fisheries management, biodiversity conservation and socioeconomic development. I then conduct a series of new spatial analyses to assess whether the MPA system as a whole possesses the characteristics of an ecologically connected and functional MPA network.

4.2. Methods

4.2.1. Study region

The Western Bohol Sea study region (Fig. 4.1) encompasses a marine area of 12,300 km^2, with a coastline of almost 600 km. The region includes 37 coastal Municipalities in five Provinces, with a total population of c. 1.1 million. The coastal environment is characterised by fringing coral reefs with areas of seagrass and mangroves. These ecosystems support extensive small-scale and commercial fisheries, which form an important source of food and income for a large proportion of the population. Although heavily exploited, the region has been identified as a high conservation priority for corals and reef fishes, and is an important migratory corridor for cetaceans, whale sharks, manta rays and turtles (Ong et al. 2002).

MPA data were extracted from the national MPA database (developed in Chapter 1), and supplemented with additional information collected from site visits (Alcala et al. 2008) and monitoring (CCEF 2009). At present, the region has 90 MPAs (Fig. 4.1, Table 4.1), all except four of which are community-based (of these, three were initially community-based no-take areas that were subsequently designated under the national integrated protected areas system (NIPAS)). I define community-based MPAs as those that are designated under municipal-level ordinances and are managed primarily by local community groups (e.g. fisherfolk associations, people's organisations), often with assistance from local government units or NGOs. Community-based MPAs typically consist of a core zone, in which all fishing and extractive activities are prohibited, and an adjacent buffer zone, in which limited fishing or gleaning is permissible, for example restricted by gear type or to members of the local community (White et al. 2002).

The MPAs in the region vary in age (2-36 years of protection) and management effectiveness (Table 4.1). Of the 77 MPAs rated under the Marine Protected Area Rating System (White et al. 2006b), two are rated as excellent, with 'institutionalised' management, 31 have 'sustained' management, 22 are rated as 'enforced', 14 as 'established', and the remaining eight as 'initiated'. In most cases MPA establishment and site selection has been driven by local-scale fisheries management objectives, although potential income from tourism is often an additional motivating factor (Walmsley & White 2003; Alcala & Russ 2006). Specific MPA objectives are not consistently documented, although Municipal Ordinance legislations may contain broad objective statements, for example *'to generate food security and ensure that fish stocks shall remain for future generations'* (Guiwanon Punta Cruz Fish Sanctuary Municipal Ordinance).

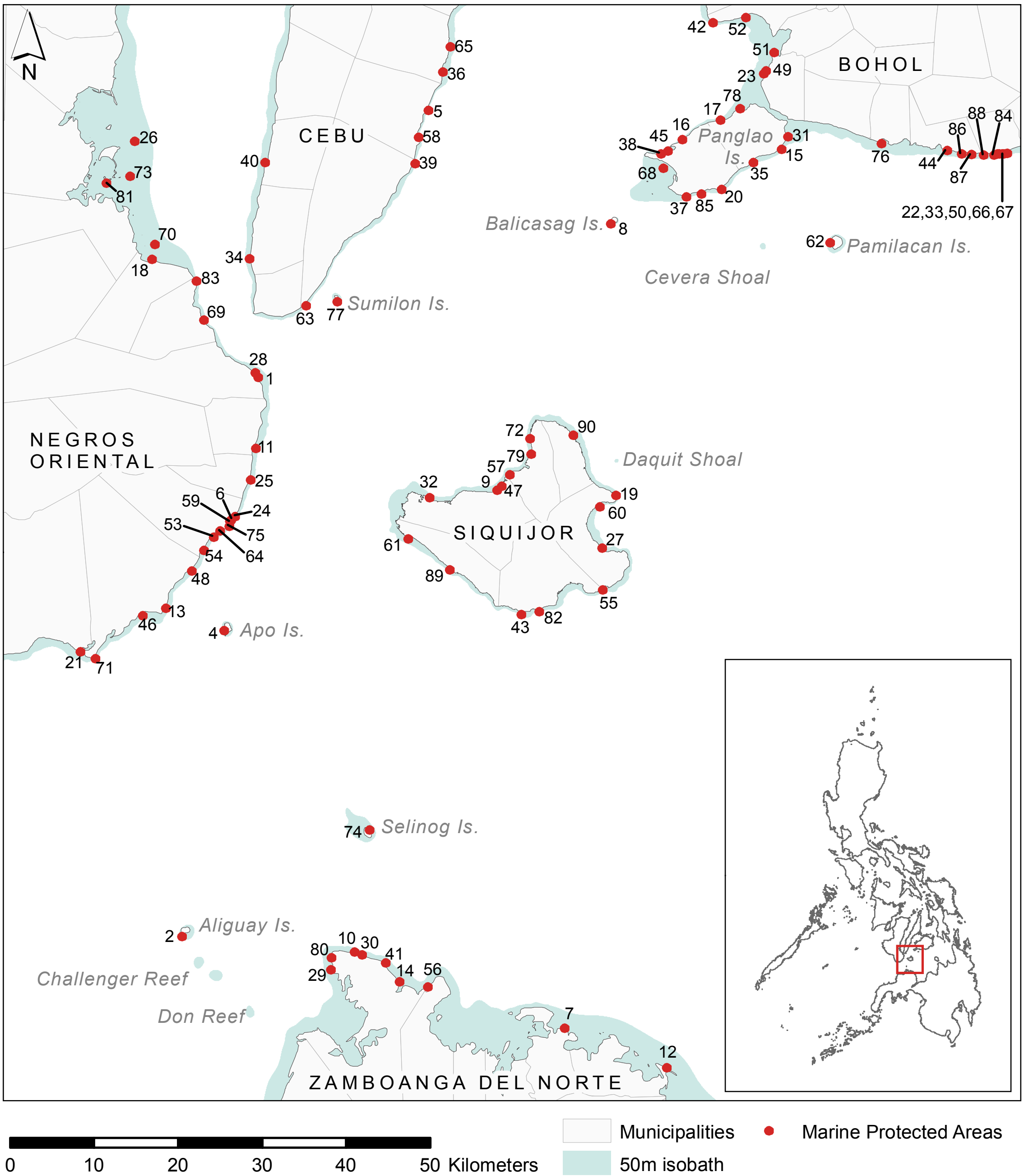


Figure 4.1. Marine protected areas in the Western Bohol Sea region of the Philippines.

Table 4.1. Marine protected areas in the Western Bohol Sea study region, with references

Marine protected area name	Map reference (Fig. 1)	Year Established	No-take area (km2)	MPA rating [a]	References
Agan-an Marine Reserve	1	1999	0.06	3	Raymundo et al. 2009
Aliguay Island Protected Landscape and Seascape	2	1999	0.12	1	
Andulay Marine Reserve	3	1993	0.06	3	Nillos-Kleiven and Stockwell 2008; Stockwell et al. 2009
Apo Island Protected Landscape and Seascape	4	1985	0.15	4	Vogt 1997; Cadiz and Calumpong 2000; Maypa et al. 2002; Pido et al. 2003; Russ and Alcala 2003; Russ et al. 2003; Walmsley and White 2003; Russ and Alcala 2004; Russ et al. 2004; Abesamis and Russ 2005; Alcala et al. 2005; Abesamis et al. 2006a; Abesamis et al. 2006b; Leisher et al. 2007; Raymundo et al. 2008; Raymundo et al. 2009; White et al. 2010
Arbor Marine Sanctuary	5	2001	0.09	3	
Bahura Marine Reserve	6	2005	0.04	1	
Baliangao Protected Landscape and Seascape	7	1991	0.05	2	de Guzman 2003, 2004
Balicasag Fish Sanctuary	8	1985	0.03	4	Christie et al. 2002; Walmsley and White 2003; Christie 2004; Abesamis et al. 2006; White et al. 2007; PADAYON Bohol Marine Triangle Management Council 2009; Raymundo et al. 2009
Banban-Luyang Marine Sanctuary	9	2007	0.05	2	
Banbanan Marine Sanctuary	10	2003	0.12	4	
Banilad Marine Reserve	11	2001	0.76	4	
Baobaon Fish Sanctuary	12	2002	0.05	2	Stockwell et al. 2009
Basak Marine Sanctuary	13	2006	0.08	NA	Stockwell et al. 2009
Baylimango Marine Sanctuary	14	2001	0.22	4	Stockwell et al. 2009
Biking Marine Sanctuary	15	2005	0.05	2	

Table 4.1. (*continued*)

Bil-isan Fish Sanctuary	16	1998	0.08	4	White et al. 2007; Raymundo et al. 2008; PADAYON Bohol Marine Triangle Management Council 2009
Bingag Marine Sanctuary	17	2003	0.07	4	Flores-Salgado and Parras 2005; PADAYON BMT Council 2009
Bio-os Marine Reserve	18	1999	0.09	4	Stockwell et al. 2009
Bogo Marine Sanctuary	19	2008	0.21	NA	
Bolod Fish Sanctuary	20	1998	0.05	4	White et al. 2007; PADAYON Bohol Marine Triangle Management Council 2009
Bonbonon Marine Sanctuary	21	1995	0.05	4	Russ et al. 2005; Nillos-Kleiven and Stockwell 2008
Bonkokan Ubos Marine Sanctuary	22	2001	0.01	2	
Bool Fish Sanctuary	23	2000	0.22	NA	
Bulak Marine Reserve	24	2005	0.07	4	
Buntis Marine Sanctuary	25	2000	0.06	1	
Campuyo Marine Sanctuary	26	1994	0.51	3	
Candaping B Marine Sanctuary	27	2003	0.20	3	
Cangmating Marine Reserve	28	1997	0.06	4	White et al. 2002; Uychiaoco et al. 2003; Russ et al. 2005
Canlucani Marine Sanctuary	29	2002	0.05	4	Russ et al. 2005; Stockwell et al. 2009
Carang Marine Sanctuary	30	2003	0.09	3	Stockwell et al. 2009
Catarman Fish Sanctuary	31	1997	0.07	3	
Caticugan Fish Sanctuary	32	1989	0.14	3	White et al. 2010
Catugasan-Cayupo Marine Sanctuary	33	2001	0.01	2	
Colase Marine Sanctuary	34	2002	0.16	3	
Da-o San Isidro Marine Sanctuary	35	2002	0.11	3	White et al. 2007; PADAYON Bohol Marine Triangle Management Council 2009
Daan Lungsod and Guiwang Marine Sanctuary	36	2002	0.23	4	
Danao Fish Sanctuary	37	1998	0.08	4	Russ et al. 2005; PADAYON Bohol Marine Triangle Management Council 2009

Table 4.1. (*continued*)

Doljo Fish Sanctuary	38	1998	0.08	3	Russ et al. 2005; White et al. 2007; PADAYON Bohol Marine Triangle Management Council 2009
Gawi Marine Sanctuary	39	2003	0.13	4	
Ginatilan Marine Sanctuary	40	2005	0.12	1	
Guimputlan Marine Sanctuary	41	2003	0.28	4	Stockwell et al. 2009
Guiwanon Punta Cruz Fish Sanctuary	42	1995	0.10	4	
Lalag-Bato Marine Sanctuary	43	2003	0.08	3	White et al. 2010
Las Salinas Sur Marine Sanctuary	44	2003	0.04	1	
Looc Marine Sanctuary	45	2003	0.03	4	PADAYON Bohol Marine Triangle Management Council 2009
Lutoban Marine Reserve	46	2002	0.10	NA	Stockwell et al. 2009
Luyang Mangrove Reserve	47	1997	0.05	NA	
Maayong Tubig Marine Reserve	48	2000	0.07	3	Raymundo et al. 2008
Mabaw Marine Sanctuary	49	1997	0.02	NA	
Malinao East-West Jambawan Marine Sanctuary	50	2001	0.01	2	
Manga Fish Sanctuary	51	2001	0.15	NA	
Maraag Marine Sanctuary	52	2002	0.12	2	
Masaplod Norte Marine Reserve	53	1997	0.03	5	Russ et al. 2005; Stockwell et al. 2009
Masaplod Sur Marine Reserve	54	2001	0.06	5	Stockwell et al. 2009
Minalunan Shell and Fish Sanctuary	55	2003	0.15	3	
Napo Marine Sanctuary	56	2002	0.08	3	Russ et al. 2005
Nonoc Marine Sanctuary	57	1996	0.04	4	
North Granada Marine Sanctuary	58	2001	0.09	4	
North Lipayo Marine Reserve	59	2005	0.02	4	
Olang Marine Sanctuary	60	1988	0.21	4	
Paliton Marine Protected Area	61	2008	0.07	1	White et al. 2010

Table 4.1. (*continued*)

Pamilacan Island Fish Sanctuary	62	1986	0.12	4	Christie et al. 2002; Pido et al. 2003; Walmsley and White 2003; Samonte-Tan et al. 2007; White et al. 2007
Pasil Marine Sanctuary	63	2002	0.10	4	
Poblacion 1 Marine Reserve	64	2000	0.09	4	Stockwell et al. 2009
Poblacion Alcoy Marine Sanctuary	65	2002	0.06	3	
Poblacion Lila Marine Sanctuary	67	2001	0.23	NA	
Poblacion Panglao Marine Sanctuary	68	1986	0.12	NA	PADAYON Bohol Marine Triangle Management Council 2009
Poblacion San Jose Marine Reserve	69	1994	0.05	1	
Polo Marine Reserve	70	1998	0.02	2	
Salag Marine Reserve	71	2001	0.10	NA	
Sandugan Marine Sanctuary	72	2003	0.10	4	
Sanlagan Marine Sanctuary	73	1994	0.01	NA	
Selinog Island Protected Landscape and Seascape	74	2000	0.06	2	Nillos-Kleiven 2007
South Lipayo Marine Reserve	75	2005	0.08	1	
Sta. Filomena Marine Sanctuary	76	2001	0.12	3	
Sumilon Island Fish Sanctuary	77	1974	0.40	4	Alcala and Russ 1990; Russ and Alcala 1998a; Russ and Alcala 1998b; Russ and Alcala 1999; Russ and Alcala 2003; Walmsley and White 2003; Russ and Alcala 2004; Alcala et al. 2005; Russ and Alcala 2006
Tabalong Marine Sanctuary	78	2003	0.16	4	PADAYON Bohol Marine Triangle Management Council 2009
Taculing / Cangmalalag Marine Sanctuary	79	1998	0.13	3	
Tag-ulo Marine Sanctuary	80	2004	0.10	3	
Talabong Mangrove Swamp Forest Reserve	81	1986	0.00	NA	Pido et al. 2003
Talayong Marine Reserve	82	2003	0.07	3	White et al. 2010

Table 4.1. (*continued*)

Tandayag Marine Reserve	83	1996	0.06	3	Russ et al. 2005; Stockwell et al. 2009
Taug Marine Sanctuary	84	2001	0.27	2	
Tawala Fish Sanctuary	85	1998	0.03	3	Russ et al. 2005; White et al. 2007; Raymundo et al. 2008; PADAYON Bohol Marine Triangle Management Council 2009
Tayong Occidental Marine Sanctuary	86	2001	0.24	2	
Tayong Oriental Marine Sanctuary	87	2002	0.18	2	
Tiguis Marine Sanctuary	88	2001	0.18	NA	
Tubod Marine Sanctuary	89	1989	0.08	4	White et al. 2010
Tulapos Marine Sanctuary	90	1987	0.27	4	

[a] From the Marine Protected Area Rating System (White et al. 2006b). Level 1 = MPA has been initiated; 2 = MPA is established; 3 = MPA is enforced; 4 = MPA is sustained; 5 = MPA is institutionalised; NA = not yet rated.

The MPAs in the region were not planned as a network. Initially, MPAs were established at Sumilon Island (1974), Apo Island and Balicasag Island (both 1985) with assistance from Silliman University (Alcala & Russ 2006). In subsequent years the success of these MPAs spread by word of mouth throughout coastal communities in the region, resulting in widespread adoption of no-take MPAs as a fisheries management tool (Alcala & Russ 2006). Although 'community-based' MPAs increasingly benefit from improved local government capacity for coastal resource management, national and international funding schemes, MPA placement is still largely determined by the objectives of local communities. Attempts to develop ecological and social MPA networks around existing MPAs have begun retroactively (Christie et al. 2009; Eisma-Osorio et al. 2009; Lowry et al. 2009); however these efforts are still in their infancy.

4.2.2. Local-scale objectives for fisheries management, biodiversity conservation and socioeconomic development

Assessment of MPA effectiveness requires explicit statements of objectives and measurable outcomes. Given that information on the specific objectives of the MPAs in the Western Bohol Sea region was unavailable, I identified fifteen commonly stated fisheries management, biodiversity conservation and socioeconomic development objectives for MPAs, and potential indicators of effectiveness for each, from the literature (Table 4.2, adapted from Ward et al. 2001; Pomeroy et al. 2004; Sobel & Dahlgren 2004). I systematically reviewed evidence from the literature to determine whether MPAs in the Western Bohol Sea region achieve these local-scale objectives. I sought relevant publications from both peer-reviewed and grey literature. I searched the ISI Web of Knowledge database and Google (first 50 results) for the name of each MPA within the study region, and possible variations, e.g. "marine sanctuary" vs. "marine reserve". I also searched the online resource libraries of Philippine NGOs working in the region (Coastal Conservation and Education Foundation, Fisheries for Improved Sustainable Harvest, Silliman University Angelo King Center for Research and Environmental Management) for relevant publications or data. I scanned the titles and abstracts (where available) of these results, and selected for further reading any that appeared relevant to the effects of MPA implementation (as opposed to, for example, taxonomic studies undertaken on samples collected from within the MPA). The bibliographies of selected articles were checked for any additional sources.

Selected articles were read in full and searched for evidence relevant to the objectives and indicators outlined in Table 4.2. Articles were rejected if they did not contain information specific to one or more MPAs within the study region (as opposed to generalised

Table 4.2. MPA objectives and indicators of effectiveness

Objectives / Indicators	Increase in species abundance / biomass	Change in species / population structure	Change in community structure	Spillover and recruitment subsidy of target species to adjacent fished area	Increase in functional diversity	Increase in habitat quality or complexity	Type, level and return on fishing effort	Water quality	Increased understanding of human impacts on resources	Perceptions of local resource harvest	Improved quality of human health	Improved household income and quality of life	Reduced dependence on marine resources as a result of alternative livelihoods	Improved community infrastructure	Perceptions of equality in MPA management
Fisheries Management Objectives															
Sustain or enhance populations of target species	■	■					■								
Protect critical spawning stock biomass of species from fishery-related depletion	■	■													
Increase fecundity and production of eggs and larvae	■	■													
Export biomass of target species to surrounding fished areas				■			■								
Sustain or improve catch yields in adjacent fishing areas				■			■								
Biodiversity Conservation Objectives															
Protect species from negative impacts	■	■				■	■								
Protect habitat structure from negative impacts						■									
Protect rare or endangered species	■	■			■										
Restore community and ecosystem structure			■												
Maintain or restore ecosystem processes and functions			■		■			■							
Maintain or enhance ecosystem stability and resilience					■	■									
Socioeconomic Objectives															
Maintain or enhance food security	■						■		■	■					
Improve the economic status and relative wealth of coastal residents and/or resource users							■					■	■	■	
Enhance the environmental awareness and knowledge of coastal communities									■						
Distribute monetary and non-monetary benefits equitably to and through coastal communities							■				■		■	■	■

statements about MPA effectiveness), and have either temporal monitoring data or spatial comparisons between protected and unprotected sites (one-time opinion or interview data were accepted as evidence for socioeconomic objectives). Meta-analytical studies (e.g. Pollnac et al. 2001; Christie et al. 2009; Maliao et al. 2009; Pietri et al. 2009) were only included if they contained information not presented elsewhere. Studies based on the same dataset (e.g. Russ & Alcala 2003; Russ et al. 2004) were considered if the data were analysed differently; otherwise, only the most recent source was included.

From 122 initial search results, 31 references were selected for analysis. These sources contained information on 40 MPAs (44% of all MPAs in the study region). Of these 31 references, 20 appeared in peer-reviewed journals or conference proceedings, with the remainder comprising reports from NGOs or academic institutions. A full list of these references is provided in Table 4.1.

Due to the wide variety of study aims and methodologies, a meta-analytical approach to summarising MPA effectiveness was not appropriate. I therefore simply recorded evidence of MPA effectiveness for each of the 15 objectives (Table 4.2) as positive, negative or inconclusive. For example, a significant increase in the abundance of target species within an MPA over time would be recorded as positive evidence that the MPA was effectively protecting critical spawning stock biomass of species from fishery-related depletion and sustaining and enhancing the populations of target species. In cases where results required subjective interpretation, I recorded the conclusions presented in the source paper, rather than my own opinions. For example, if the authors concluded that their data supported a positive effect of MPA implementation on benthic habitat quality, I recorded a positive result, even if my own interpretation of the results differed.

In many cases multiple sources provided evidence for the same MPA and objective. I summarised this information to give a single measure of effectiveness (positive, negative or inconclusive) based on the weight of evidence in each direction and the reliability of the sources. Sources with spatial and temporal comparisons were deemed to be more reliable than those that presented only a single comparator, or gave anecdotal evidence for an MPA effect. Where an equal number of sources presented positive and negative evidence, results were recorded as inconclusive.

4.2.3. Regional-scale MPA network objectives

Scientific principles for MPA network design have been developed to guide the selection of sites. These principles attempt to make MPA networks comprehensive, representative, adequate and replicated (ANZECC TFMPA 1998). More recently, these guidelines have been supplemented to ensure that MPA networks are resilient (McLeod et al. 2009) and connected (McCook et al. 2009). Table 4.3 gives examples of how these broad guidelines have been translated into operational principles for the design of MPA networks for the Great Barrier Reef (Australia), California (US) and Kimbe Bay (Papua New Guinea). Here, I apply principles for MPA network design in retrospect to assess whether the MPA system in the Western Bohol Sea region possesses characteristics of an ecologically functional MPA network. I focus on biophysical principles for MPA network design, although these are often complemented by a set of socioeconomic operating principles (Fernandes et al. 2005; Green et al. 2009).

Comprehensive and representative

Design principles to ensure that MPA networks are comprehensive and representative typically emphasise the need to include examples of all biogeographic regions (e.g. '*represent at least 20% of each reef bioregion in no-take areas'* (Fernandes et al. 2005)), and habitat types (e.g. *'include at least 20% of the area of each habitat type'* (Green et al. 2009)), as these are the data most commonly available for marine conservation planning. I identified six distinct biogeographic regions on the basis of reef fish survey data collected at 42 survey sites across the region (see Chapter 2). I summed the total number and area of MPAs in each bioregion to determine their level of representation in the MPA system. I set a target for bioregion representation at 10%, following the Philippine Marine Sanctuary Strategy (Arceo et al. 2004), which calls for 10% of coral reef area to be protected within no-take areas by 2020 (a target that has been adopted by the CTI (Coral Triangle Initiative 2008)). The area of each bioregion was taken as the area from the coastline to the 50 m isobath. Coarse-filter habitat data for coral reefs, shoals, seagrass and mangroves were digitised from satellite imagery and bathymetric charts. However, the resolution of these data, combined with the small size of MPAs and potential inaccuracies in MPA locations, was insufficient for a GIS-based gap analysis. I therefore relied on habitat data recorded during MPA site visits (Alcala et al. 2008) to assess habitat representation.

In addition to including a representative sample of habitats and bioregions, MPA networks should also aim to represent the full range of environmental conditions encountered in the region (e.g. '*…encompassing gradients such as latitude, inshore–offshore location, depth and influences such as upwellings or river runoff'* (McCook et al. 2009)). I assessed the diversity of reef slope, current speed and exposure to river runoff in the MPA system. Reef slope was derived from bathymetric data (digitised from nautical charts) using the Slope tool in the Spatial Analyst extension for ArcGIS. Mean annual current speeds were calculated

Table 4.3. Case studies demonstrating the application of MPA network design principles

MPA network objective	Biophysical operational principles of the Great Barrier Reef Representative Areas Program (Fernandes et al 2005)	California Department of Fish and Game Master Plan for Marine Protected Areas (2008)	Biophysical design principles for an MPA network for Kimbe Bay (Green et al 2009)
Comprehensive and Representative	Represent at least 20% of each reef bioregion in no-take areas.	Every 'key' marine habitat should be represented in the MPA network.	Aim to include at least 20% of the area of each habitat type.
	Maintain geographic diversity: represent cross-shelf and latitudinal diversity in the network of no-take areas.	Each of these habitats should be represented in multiple MPAs across biogeographic regions, upwelling cells, and environmental and geographical gradients.	All else being equal choose representative areas (areas that are typical of a habitat type within which it is located) based on knowledge (high biodiversity areas, complementarity) to maximise the number of species protected.
	Represent all habitats: represent a minimum of each community type and physical environment type in the overall network.	MPAs should extend from the intertidal zone to deep waters offshore.	Where information is available, include a minimum amount of each ecosystem and community type within each habitat type (to ensure that all known communities and habitats that exist within each habitat type are protected.
	Protect uniqueness: include biophysically special/unique places.		Include special and unique sites, including: cetacean preferred habitats; turtle nesting areas; nursery grounds; spawning aggregations; migratory corridors; areas of high species diversity.
Adequate	No-take areas should be at least 20 km long on the smallest dimension; for coastal bioregions, protect at least six no-take areas, each at least 10 km long.	MPAs should have an alongshore span of 5-10 km of coastline, and preferably 10-20 km.	Where possible, include entire biological units (e.g. whole reefs, seamounts), including a buffer around the core area of interest.

Table 4.3. (*Continued)*	Great Barrier Reef	California	Kimbe Bay
Adequate	Protect fewer, larger areas rather than more, smaller areas. Where a reef is incorporated into no-take areas, the whole reef should be included.	Larger MPAs should be required to fully protect marine birds, mammals, and migratory fish.	Where entire biological units cannot be included, chose bigger vs. smaller areas.
Replicated	Recommend 3-4 no-take MPAs in each bioregion to safeguard against negative impacts on some part of a bioregion.	At least three to five replicate MPAs should be designed for each habitat type within a biogeographic region.	Include a sufficient number and area of each habitat type, and spread them out geographically to reduce the chances that they will all be negatively affected at the same time. Aim to include at least three areas of each habitat type.
Connected	Accommodate what is known about migration patterns, currents, and connectivity among habitats; include consideration of sea and adjacent land uses.	MPAs should be placed within 50-100 km of each other.	Take a system-wide approach that recognises patterns of connectivity within and among ecosystems.
Resilient			All else being equal choose sites that are more likely to be resistant or resilient to global environmental change.

from monthly average Naval Research Laboratory Layered Ocean Model (NLOM) data for the Bohol Sea. Exposure to river runoff was calculated as a function of distance from river mouth and the expected magnitude and sediment load of the river plume (large rivers were assumed to have an influence up to 5 km from their mouth, small rivers 2 km and streams 1km).

Finally, in line with guidelines to protect 'special and unique' places (Fernandes et al. 2005; Green et al. 2009), I sought to determine whether small islands, shoals, spawning grounds and sites frequented by cetaceans, elasmobranchs and turtles were adequately represented in the MPA system.

Adequate

Guidelines for ensuring that MPA networks are adequate typically focus on the size of individual MPAs, e.g. *'protect fewer, larger areas rather than more, smaller areas'* (Fernandes et al. 2005), '*MPAs should have an alongshore span of 5-10 km of coastline, and preferably 10-20 km'* (CDFG 2008). A common recommendation is that whole biological units, for example entire reefs, be protected (Fernandes et al. 2005; Green et al. 2009). I assessed the adequacy of the MPA system by evaluating the no-take area of individual MPAs. Given that much of the region is characterised by continuous fringing reefs, it would be difficult to adhere to the principle of including entire biological units (as this would suggest designating an entire coastlines as an MPA). Nevertheless, I qualitatively assessed how well this guideline was achieved for non-fringing reefs.

Replicated

To determine the degree of replication provided by the existing MPA system, I evaluated the number of MPAs containing each habitat type within each bioregion. I set a target of three occurrences for each habitat in each bioregion.

Connected

McCook et al (2009) provide a set of guidelines for maximising connectivity in the design of MPA networks. In addition to reiterating many of the design principles stated above, they suggest that MPA networks should aim to provide for a wide range of dispersal distances by varying the spacing between MPAs, with a maximum distance between protected areas of 30 km and most connections in the region of 10-20 km (McCook et al. 2009). I measured the distance from each MPA in the region to its nearest neighbour in ArcGIS, first irrespective of oceanographic currents, and then considering prevailing currents (measuring downstream distances only) to determine whether MPAs in the

Western Bohol Sea adhere to these recommendations. I also sought to determine whether MPAs provide contiguous protection for habitats that encompass different life stages (Mumby 2006)

Resilient

Guidelines for designing resilient MPA networks emphasise the importance of incorporating sites that demonstrate natural resistance or resilience to coral bleaching (West & Salm 2003; Marshall & Schuttenberg 2006; McLeod et al. 2009). Proximity to deep water is frequently employed as an indicator of likely reef resilience (West & Salm 2003; Obura 2005; Marshall & Schuttenberg 2006; McLeod et al. 2009); nevertheless, specific guidelines (i.e. how close? how deep?) for identifying resilient sites are lacking. I assessed the potential resilience of MPAs in the Western Bohol Sea to climate change-induced coral bleaching by measuring the distance from each MPA to the nearest 50m and 100m isobath (digitised from nautical charts) using the Proximity Toolset for ArcGIS. Areas subject to strong currents or with steep slopes may also have increased resistance to bleaching, as a result of enhanced water mixing (with associated cooling effect) and reduced light stress (from shading) respectively (Obura 2005). I therefore also determined whether sites with high annual mean current and / or steep slopes were included in the MPA system.

4.3. Results

4.3.1. Local-scale objectives

A summary of the effectiveness of individual MPAs in terms of local-scale objectives for fisheries management, biodiversity conservation and socioeconomic development is presented in Figure 4.2 and Table 4.4.

The objective most frequently assessed was the effectiveness of MPAs at sustaining or enhancing populations of species targeted by fishers. The strongest evidence for a positive effect comes from Apo Island, where a threefold increase in the density and biomass of *Naso vlamingii*, a targeted surgeonfish, was observed during 20 years of protection (Russ et al. 2003; 2004; Abesamis & Russ 2005). Increases in target species abundance, density or biomass were observed or inferred for a further 11 MPAs (Table 4.4). At five sites no significant change in target species abundance was observed, indicating that populations have been sustained, rather than enhanced. Two MPAs were not found to be effective at sustaining target species populations: at Pamilacan Island, the density of target species decreased by approximately 40% between 1986 and 2007, both inside and outside the MPA (White et al. 2007b), and at Bolod Fish Sanctuary, target fish abundance did not

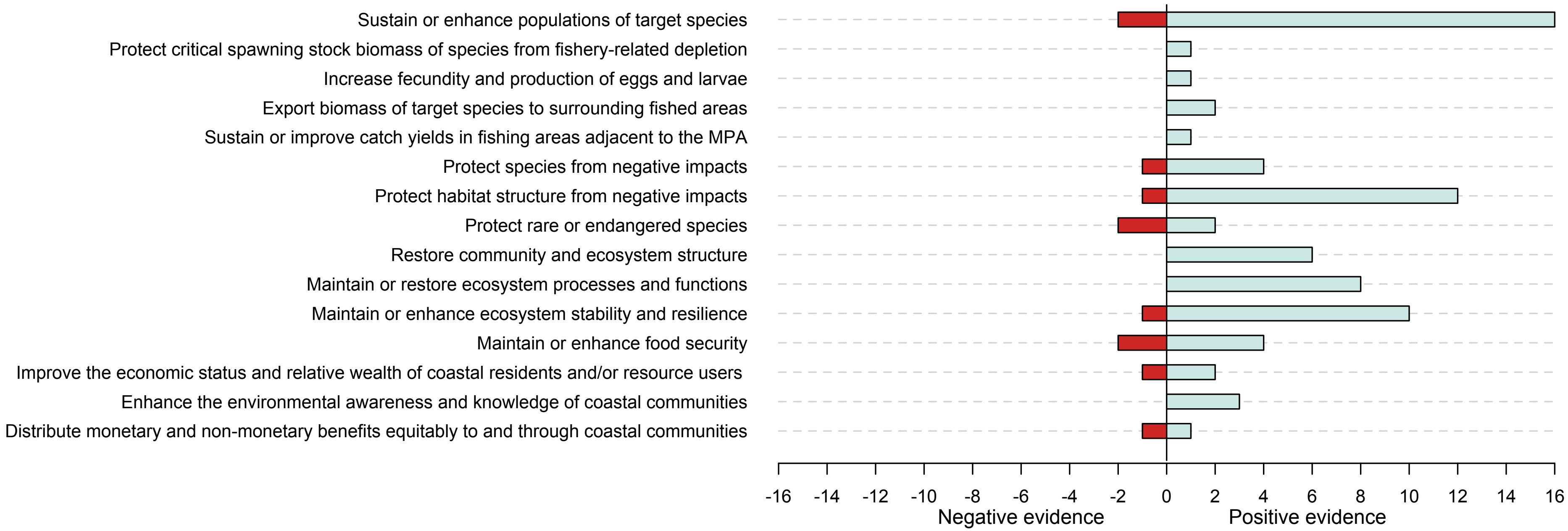


Figure 4.2. Summary of evidence that MPAs in the Western Bohol Sea region achieve local scale objectives for fisheries management, biodiversity conservation and socioeconomic development. Numbers on the x axis indicate the number of MPAs for which positive or negative evidence of effectiveness were available. MPAs for which evidence of effectiveness were equivocal are not shown.

Table 4.4. Summary of evidence for MPA effectiveness against local-scale objectives for fisheries management, biodiversity conservation and socioeconomic development

Marine Protected Area	Agan-an	Andulay	Apo Island	Baliangao	Balicasag	Baobaon	Basak	Baylimango	Bil-isan	Bingagy	Bio-os	Bolod	Bonbonon	Cangmating	Canlucani	Carang	Caticugan	Da-o San Isidro	Danao	Doljo
Fisheries Management Objectives																				
Sustain or enhance populations of target species			+	+	/				/			-	+	+	+		+	/	+	/
Protect critical spawning stock biomass of species from fishery-related depletion			+																	
Increase fecundity and production of eggs and larvae			+																	
Export biomass of target species to surrounding fished areas			+		/															
Sustain or improve catch yields in fishing areas adjacent to the MPA			+																	
Biodiversity Conservation Objectives																				
Protect species from negative impacts			+		/				+	/		+	-	+				/	/	/
Protect habitat structure from negative impacts		/	+	/	+				/	+		+	-				/	/	+	/
Protect rare or endangered species																				
Restore community and ecosystem structure																				
Maintain or restore ecosystem processes and functions		+				+	+	+			+				+	+				
Maintain or enhance ecosystem stability and resilience	+	+	+		+	+	+	+	-		+				+	+				
Socioeconomic Objectives																				
Maintain or enhance food security			+	+						+				+						
Improve the economic status and relative wealth of coastal residents and/or resource users			+	-																
Enhance the environmental awareness and knowledge of coastal communities			+		+															
Distribute monetary and non-monetary benefits equitably to and through coastal communities			/		/															

+ evidence of positive effect; - evidence of negative effect; / inconclusive evidence.

Table 4.4. (continued)

Marine Protected Area	Guimputlan	Lalag-Bato	Looc	Lutoban	Maayong Tubig	Masaplod Norte	Masaplod Sur	Napo	Paliton	Pamilacan Island	Poblacion 1	Poblacion Panglao	Selinog Island	Sumilon Island	Tabalong	Talabong	Talayong	Tandayag	Tawala	Tubod
Fisheries Management Objectives																				
Sustain or enhance populations of target species		+				+		+	+	-				+			+	+	+	+
Protect critical spawning stock biomass of species from fishery-related depletion														+						
Increase fecundity and production of eggs and larvae																				
Export biomass of target species to surrounding fished areas														+						
Sustain or improve catch yields in fishing areas adjacent to the MPA														+						
Biodiversity Conservation Objectives																				
Protect species from negative impacts			/		+					/		-	+	+	/				/	-
Protect habitat structure from negative impacts										/			+	/	/				/	/
Protect rare or endangered species																				
Restore community and ecosystem structure														+						
Maintain or restore ecosystem processes and functions	+			+		+	+				+							+		
Maintain or enhance ecosystem stability and resilience	+			+	+	+	+				+			+				+		
Socioeconomic Objectives																				
Maintain or enhance food security										-				/		-				
Improve the economic status and relative wealth of coastal residents and/or resource users										+				+						
Enhance the environmental awareness and knowledge of coastal communities										+										
Distribute monetary and non-monetary benefits equitably to and through coastal communities										+				-						

+ evidence of positive effect; - evidence of negative effect; / inconclusive evidence.

increase in 9 years after establishment, and was observed to be lower inside the MPA than in an adjacent fished area (White et al. 2007b).

In addition to protecting and enhancing populations of target species within MPA boundaries, there is evidence to suggest that Apo Island Fish Sanctuary is effectively exporting biomass of target species to surrounding fished areas (also termed 'spillover'). Surveys undertaken over two decades document greater increases in the biomass of *N. vlamingii* closer to the MPA boundary than further away (Russ et al. 2003), a gradient of decreasing abundance across the MPA boundary (Abesamis et al. 2006b), and a density-dependent mechanism for spillover (Abesamis & Russ 2005). Catch per unit effort (CPUE) was also found to be higher close to the MPA boundary (Russ et al. 2004; Alcala et al. 2005; Abesamis et al. 2006a).

The effectiveness of MPAs at protecting non-target species and habitats from negative impacts was equivocal (Table 4.4). Whilst increases in the density and diversity of reef fish species were observed at six MPAs, declines in species richness were observed at Tubod Marine Sanctuary (White et al. 2010), Pamilacan Island Fish Sanctuary (Walmsley & White 2003) and Bonbonon Marine Sanctuary (Nillos-Kleiven & Stockwell 2008). Many studies were inconclusive, documenting inconsistent trends in density or species richness. For example, at Balicasag Island Fish Sanctuary, species richness initially increased, then subsequently declined and increased again (Walmsley & White 2003). Studies that addressed the effectiveness of MPAs at protecting habitat structure typically did so by exploring spatial or temporal trends in live coral cover. In most cases, no significant trends were observed. Christie et al (2002) note that although MPA implementation at Balicasag and Pamilacan Islands eliminated dynamite fishing, coral damage still occurred as a result of the increasing number of anchors dropped on the reef by tourist boats. No studies explicitly assessed whether MPAs were effective at protecting rare or endangered species.

Russ and Alcala (1998) reported significant temporal changes in community structure at Sumilon Island Fish Sanctuary, which occurred in response to sequential periods in which the MPA was opened and then re-closed to fishing (Russ & Alcala 1999). This indicates that, when functional, the MPA was effective at maintaining community and ecosystem structure. Positive evidence that MPAs can maintain or enhance ecosystem stability and resilience came from two recent studies. Stockwell et al (2009) inferred a positive relationship between MPA protection and the functional diversity of herbivores for 11 MPAs in the study region, documenting an increase in the density and biomass of herbivorous fishes, changes in herbivore community composition and an associated reduction in macroalgal cover with increasing duration of protection. Raymundo et al (2009) found that MPAs had significantly higher fish taxonomic diversity and lower incidences of coral disease than adjacent unprotected areas at seven sites. The study by Stockwell et al (2009) also lends support to the hypothesis that MPAs can restore ecosystem processes and functions.

Relatively few studies explored whether MPAs were successful at achieving socioeconomic objectives. At six MPAs, fishers were asked about their perceptions of catch or CPUE in the time since MPA establishment: at Apo Island (Russ & Alcala 1996; Pido et al. 2003; Leisher et al. 2007), Baliangao Protected Landscape and Seascape (de Guzman 2003), Bingag Marine Sanctuary (Flores-Salgado & Parras 2005) and Canmating Marine Reserve (Uychiaoco et al. 2003) fishers perceived that their catch had increased as a result of MPA implementation. At Pamilacan Island Fish Sanctuary and Talabong Mangrove Swamp Forest Reserve fishers believed their catch to be decreasing (Pido et al. 2003). Only one study measured catch rates directly: Russ et al (2004) used published estimates of fishery catch and effort, and fisher interviews to demonstrate that the total catch of two targeted families of reef fish at Apo Island was significantly higher after MPA establishment.

The economic status and relative wealth of coastal residents and resource users was found to have improved at Apo Island (Vogt 1997; Cadiz & Calumpong 2000; Russ et al. 2004; Leisher et al. 2007) and Pamilacan Island (Samonte-Tan et al. 2007), largely as a result of income from tourism associated with the MPAs. However, fishers' monthly income remained below the poverty level in Baliangao Protected Landscape and Seascape (de Guzman 2003). Enhanced environmental awareness and knowledge was documented for coastal communities at Apo, Balicasag and Pamilacan Islands (Walmsley & White 2003), and the MPA at Apo Island was also found to have had a positive effect on residents' health and nutrition (Leisher et al. 2007).

Socioeconomic benefits resulting from MPA implementation were not distributed equitably through coastal communities. Although Walmsley and White (2003) found that more than 90% of interviewees at Apo, Balicasag and Pamilacan Islands felt that the whole community was involved in the management of the MPA, community members at Apo Island later reported conflicts over MPA management and the distribution of funds resulting from tourism (Leisher et al. 2007). Similarly, at Balicasag Island, Christie (2004) reported that national government control of the MPA has led to a loss of local community

Table 4.5. Summary of MPA network objective results

MPA Network Objective	Design principles	Evidence
Comprehensive and representative	Represent all marine habitat types	Coral reef habitats are well represented; seagrass, macroalgal beds and mangroves have reasonable levels of representation; lagoons, rocky intertidal and soft bottom habitats are underrepresented.
	Represent at least 10% of each bioregion	Bioregional representation varies between 0.6% and 4.7%.
	Encompass gradients in current strength, slope and exposure to river runoff.	MPAs are biased towards areas with low to moderate currents, shallower slopes and low exposure to river runoff.
	Include ecologically important sites, e.g. sites used by cetaceans, elasmobranchs and turtles, and known spawning aggregations	No data available for spawning aggregations. Cetaceans, elasmobranchs and turtles are protected by alternative methods.
	Include biophysically special and unique places, e.g. small islands and shoals	Small islands are well represented in the MPA system: 100% have MPAs, which protect a relatively high proportion of their area. The MPA system provides no protection for shoals.
Adequate	Bigger, rather than smaller areas selected for protection	All MPAs in the system have no-take areas <1 km^2, with 98% smaller than 0.5 km^2.
	Where possible, include entire biological units or reefs	Difficult to achieve due to the predominance of fringing reefs and the need to leave some area of the reefs surrounding small islands accessible to communities dependent upon fishing.
Replicated	Include at least three examples of each habitat type in each bioregion	Coral reef and seagrass habitats are well replicated in the MPA system; other habitat types are inconsistently replicated.
Connected	Provide for a wide range of dispersal distances between protected areas, particularly dispersal distances up to 10–20 km	All MPAs are within 20 km of another MPA, providing for a wide range of dispersal distances; 80% are within 4 km of another MPA.
	Ensure contiguous protection for habitats that encompass different life stages	Occurs naturally in many cases as MPAs typically extend from the shoreline to beyond the reef crest.
Resilient	Protect sites that are likely to be more naturally resistant to coral bleaching events.	MPAs are likely to have good resistance to bleaching events due to their proximity to deeper, cooler water: more than 80% of MPAs are within 1km of the 50 m isobath.

support. At Sumilon Island Fish Sanctuary, respondents felt they had no power or control over MPA regulations and felt no sense of ownership (Walmsley & White 2003).

4.3.2. MPA network objectives

In this section I discuss how well the MPA system in the Western Bohol Sea region adheres to ecological principles for MPA network design. This information is summarised in Table 4.5.

Comprehensive and representative

The 90 MPAs in the Western Bohol Sea region have a combined area of 56.73 km^2, of which 10.12 km^2 is no-take. In total, 1.95 % of area to the 50 m isobath is within no-take MPAs. None of the six bioregions achieved the representation target of 10%. The area (to the 50 m isobath) included in MPAs varied between 0.6% (Sulu-Zamboanga bioregion) and 4.7% (Southern Bohol bioregion)(Figure 4.3, Table 4.6). Likewise, the MPA system is not fully representative of environmental variation in the region (Fig. 4.4). MPAs were more likely to be situated in areas with low to moderate current, although a few (e.g. Pasil Marine Sanctuary, Colase Marine Sanctuary, Sumilon Island Fish Sanctuary) are in areas subject to strong currents (Fig. 4.4a). Sites with steep slopes are less well represented than those with lower gradients (Fig. 4.4b) and MPA locations are strongly biased away from sites that are likely to be exposed to river runoff, with more than 90% not exposed to any river influence (Fig. 4.4c).

All except five MPAs have been established primarily to protect coral reef habitats: Minalulan Shell and Fish sanctuary contains lagoon and soft bottom habitat with patchy coral; Looc Marine Sanctuary and Poblacion Panglao Marine Sanctuary consist predominantly of seagrass beds; and Luyang Mangrove Reserve and Talabong Mangrove Swamp Forest Reserve primarily protect mangroves. Whilst coral reefs are well represented in the MPA system, other marine and coastal habitat types are afforded less protection (Table 4.7). In particular, rocky intertidal, soft bottom and lagoonal habitats are underrepresented.

MPAs have not been established specifically to protect sites used by cetaceans, elasmobranchs and turtles, although the region is known to be an important migration corridor for these species, and has a history of directed cetacean and manta ray fisheries (Dolar et al. 1994; Eckert et al. 2002). On Pamilacan Island, the whale shark and manta ray fishery was a primary source of income until it was prohibited by national law in 1999 (Walmsley & White 2003); conservation efforts by local NGOs have focused on re-training fishers to provide whale- and dolphin-watching trips for tourists. No data were available

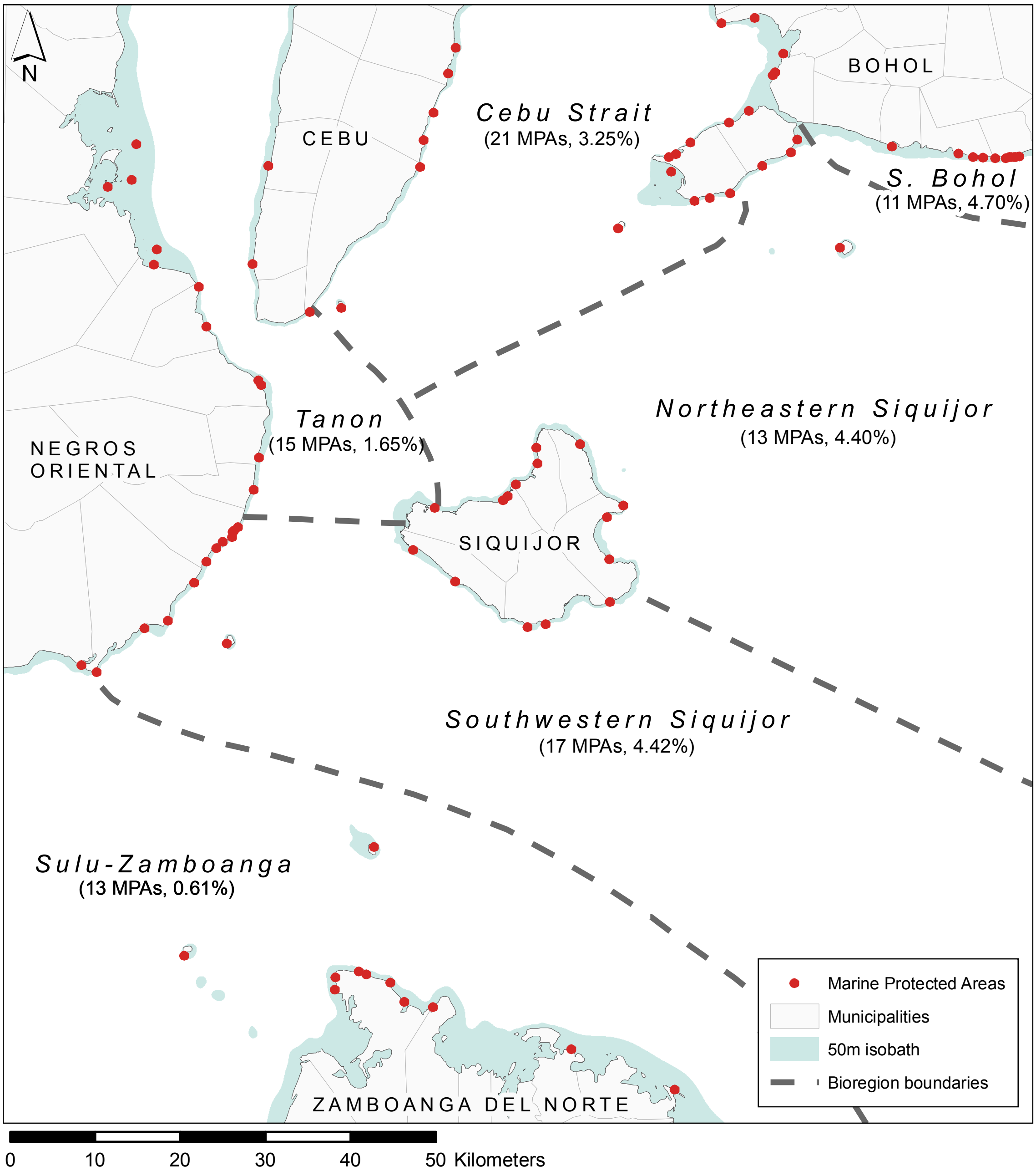


Figure 4.3. Representation of marine bioregions in the Western Bohol Sea MPA system. Dashed grey lines indicate the boundaries between bioregions. Numbers in parentheses indicate the number of existing no-take MPAs and the percentage of the area to the 50 m isobath (shaded in blue) currently protected in no-take MPAs within each bioregion.

Table 4.6. Representation of bioregions in the MPA system

Bioregion	Number of MPAs	Percentage of area to the 50 m isobath in MPAs
Cebu Strait	21	3.25%
North-east Siquijor	13	4.40%
Southern Bohol	11	4.70%
Sulu-Zamboanga	13	0.61%
South-west Siquijor	17	4.22%
Tanon	15	1.65%

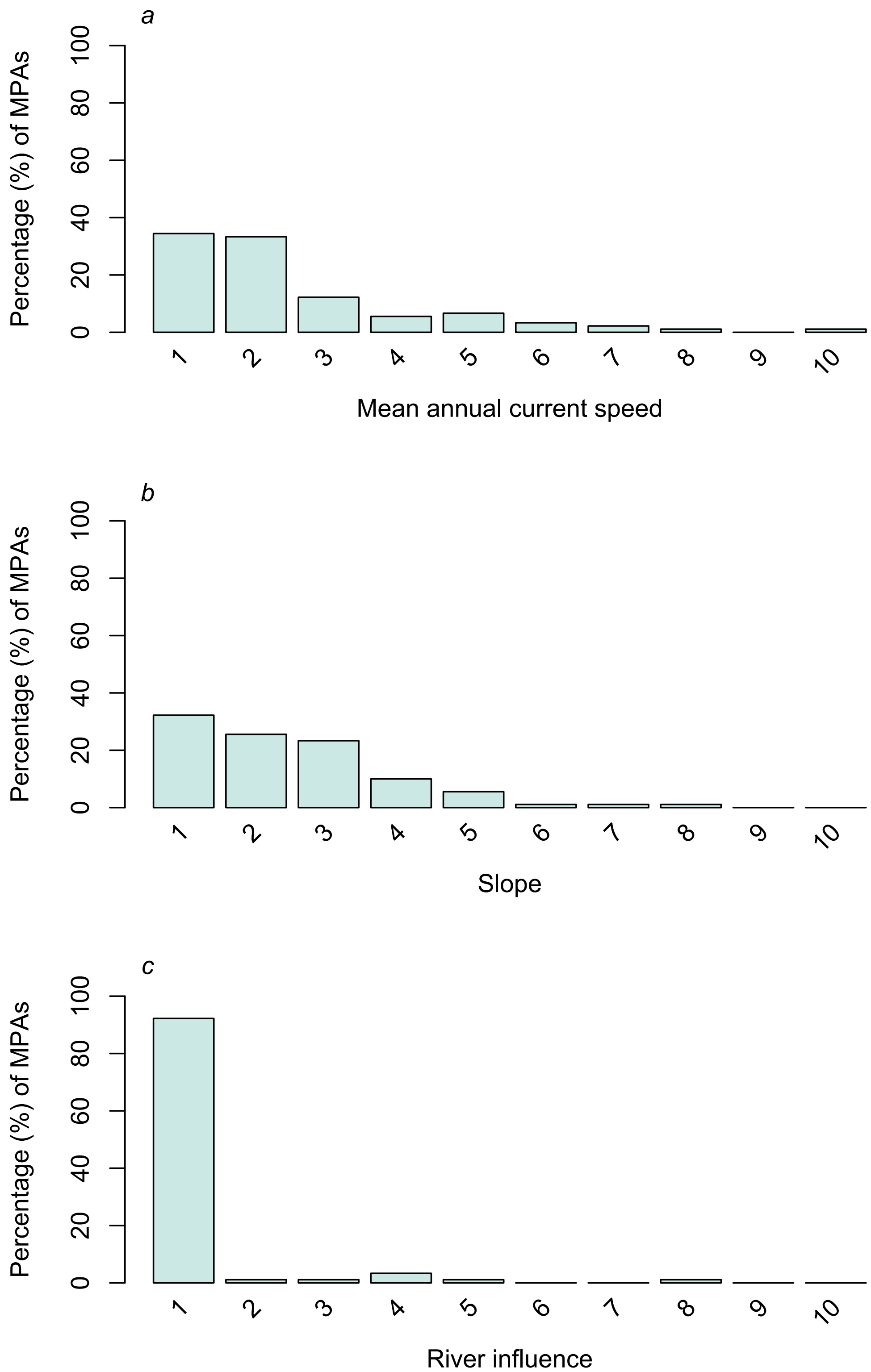


Figure 4.4. Representation of environmental variation in the Western Bohol Sea MPA system: (a) current speed; (b) slope; and (c) exposure to river runoff. X axes represent the range of values for each environmental factor that occur within the area to the 50 m isobath on a relative scale, with 1 indicating the lowest value and 10 the highest. All distributions differ significantly from a uniform distribution (Kolmogorov-Smirnov tests, $P < 0.05$)

Table 4.7. Representation and replication of habitat types in the MPA system, by bioregion [a]

	Cebu Strait	North-east Siquijor	Southern Bohol	Sulu-Zamboanga	South-west Siquijor	Tanon	Total
Coral reef	20	12	11	13	17	14	87
Seagrass	15	7	9	2	3	3	39
Mangrove	0	6	1	2	0	2	11
Macroalgae	6	0	4	0	1	2	16
Open water	3	1	4	0	0	0	8
Rocky intertidal	2	0	3	0	0	0	5
Soft bottom	2	4	0	0	0	2	5
Lagoon	0	0	0	0	1	0	1

[a] Numbers indicate the number of MPAs containing each habitat type in each bioregion

to assess whether the MPA system provides specific protection for fish spawning aggregations.

Small coralline and volcanic islands are very well represented in the MPA system. All six small islands in the region (Aliguay, Apo, Balicasag, Pamilacan, Selinog, and Sumilon) have no-take MPAs, and the proportion of small island reefs protected is higher than that for any bioregion (mean percentage of bioregions within MPAs = 3.14%, small islands = 13.8%). However, MPAs have not been established on any of the region's four shoals (Cevera Shoal, Daquit Shoal, Challenger Reef and Don Reef).

Adequate

The MPA system does not adhere to recommendations to select bigger, rather than smaller areas for protection. Only four MPAs have a total area greater than 1 km^2, and all have no-take areas smaller than 1 km^2, with 98% smaller than 0.5 km^2 (Fig. 4.5). The largest no-take MPA is Banilad Marine Reserve, which has an area of 0.76 km^2. The mean no-take area of MPAs in the Western Bohol Sea region is 0.11 km^2, which does not differ significantly from the national average of 0.12 km^2 (Weeks et al. 2010a, Chapter 2). Additionally, the principle of protecting entire biological units is not adhered to. This is not surprising, given that the region is characterised by continuous fringing reefs. However, even where MPAs are located on small islands, only a partial area of the reef is protected.

Replicated

Coral reef ecosystems were well replicated in the MPA system, with all bioregions containing a minimum of 11 coral reef MPAs (Table 4.7). Seagrass habitats were also reasonably well replicated, with only one bioregion (Sulu-Zamboanga) failing to achieve the target of three occurrences. Other habitat types were less well replicated. Six MPAs in the North-east Siquijor bioregion contained mangroves, but there were no more than two mangrove occurrences in any other bioregion. Macroalgae, rocky intertidal, soft bottom, lagoon and open water ecosystems showed similarly inadequate replication in the MPA system (Table 4.7).

Connected

The MPA system follows guidelines for inter-MPA spacing (McCook et al. 2009) well. More than 80% of MPAs are within 5 km of their closest neighbour, and all MPAs are within 20 km of another MPA (Fig. 4.6). When measured unidirectionally to account for prevailing currents, the distances between MPAs are slightly greater (Fig. 4.6). Nevertheless, only three MPAs are further than 20 km from their nearest neighbour, and 87% are within 10

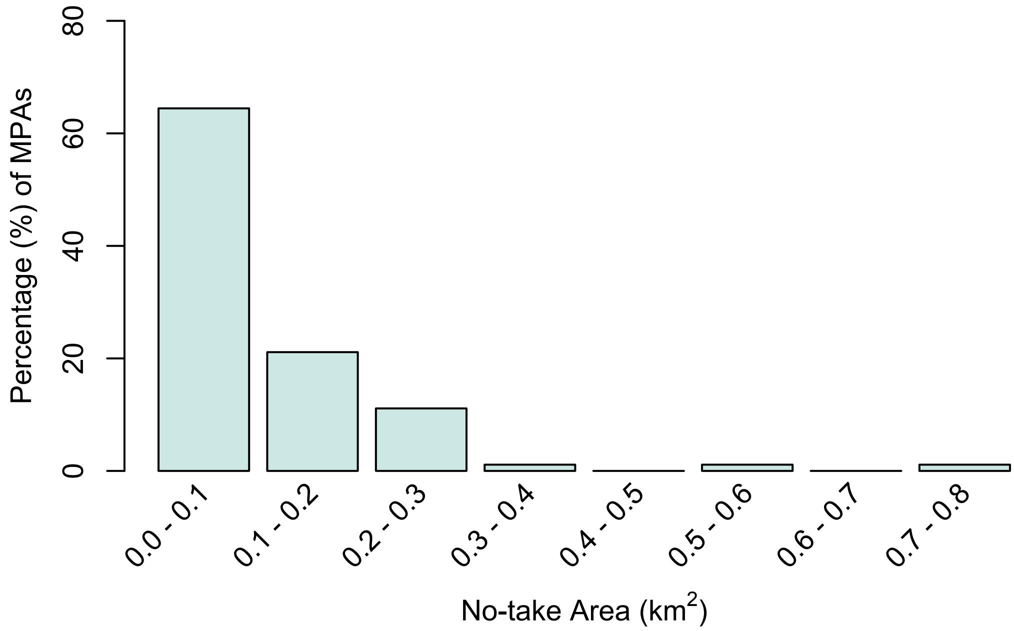


Figure 4.5. Size frequency distribution of MPAs in the Western Bohol Sea study region.

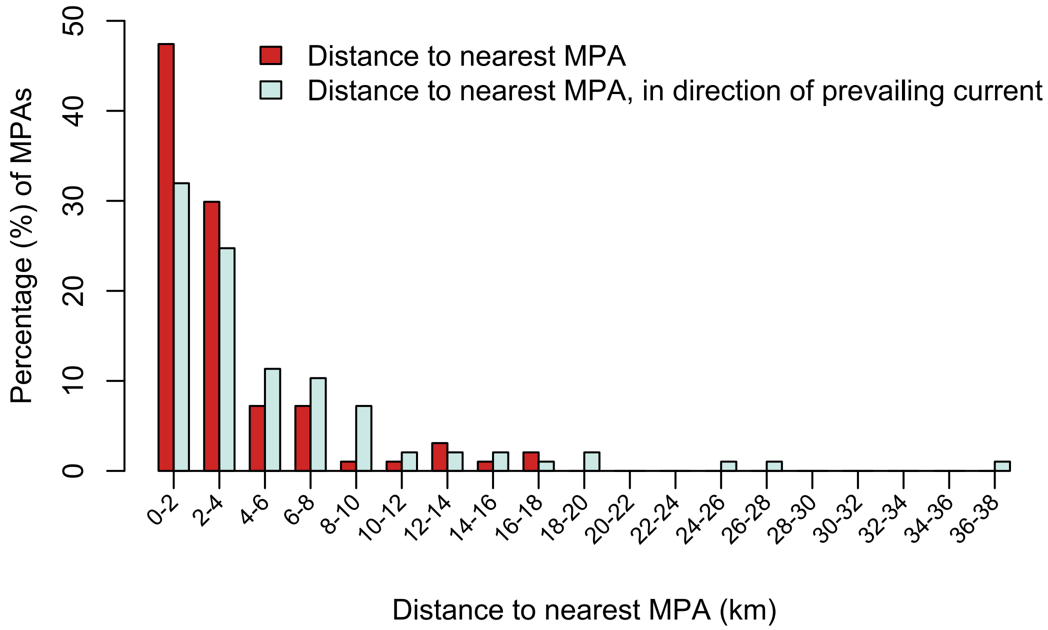


Figure 4.6. Distance between MPAs in the Western Bohol Sea system. Red bars show the distance to the nearest MPA irrespective of oceanographic currents; blue bars show distance to the nearest MPA in the direction of prevailing currents.

km. Given the frequency distribution of distances between MPAs (Fig. 4.6), medium and long dispersal distances will automatically be provided for (McCook et al. 2009).

Although there is no evidence to suggest that MPAs have been designed to protect contiguous habitats that encompass different life stages, in many cases this has occurred naturally. The majority of MPAs are designed to have a no-take area that extends from the shoreline to the bottom of the reef slope (occasionally with a narrow buffer zone adjacent to the shore to allow beach access). Thus, where mangroves and / or seagrass beds precede the coral reef area, these habitats are typically also protected. Seven MPAs contain coral reef, seagrass and mangrove habitats; many more include both coral reef and seagrass or macroalgal beds.

Resilient

More than 80 % of MPAs are within 1 km of the 50 m isobath, and more than 40% are within 1 km of the 100 m isobath (Fig. 4.7). This proximity to deeper and cooler water indicates that the MPA system is likely to demonstrate good resistance to coral bleaching events, despite poor representation of areas with steep slopes and high currents.

4.4. Discussion

The MPA system in the Western Bohol Sea was not designed to achieve many of the objectives examined here. Assessments of the effectiveness of protected areas should be made against the original objectives for which they were established; thus, MPAs in this region should be considered successful if they achieve local-scale objectives for fisheries management and socioeconomic development. However, given the global importance of the Philippines for marine biodiversity, and the widespread dependence of coastal communities on marine resources, it is important to also assess whether these MPAs can achieve objectives for conservation and fisheries management at broader spatial scales.

Previous meta-analyses of the effects of no-take MPAs have sought to identify factors that may explain heterogeneity in effectiveness, e.g. duration of protection, location, enforcement, size, or distance from other MPAs (Cote et al. 2001; Pollnac et al. 2001; Halpern 2003; Claudet et al. 2008; Garia-Charton et al. 2008; Lester et al. 2009; Molloy et al. 2009). My aim here was not to determine why some MPAs are more effective than others, but to look more broadly at the effectiveness, in terms of objectives for fisheries management, biodiversity conservation and socioeconomic development, of an MPA system that has been established following an *ad hoc*, community-based approach.

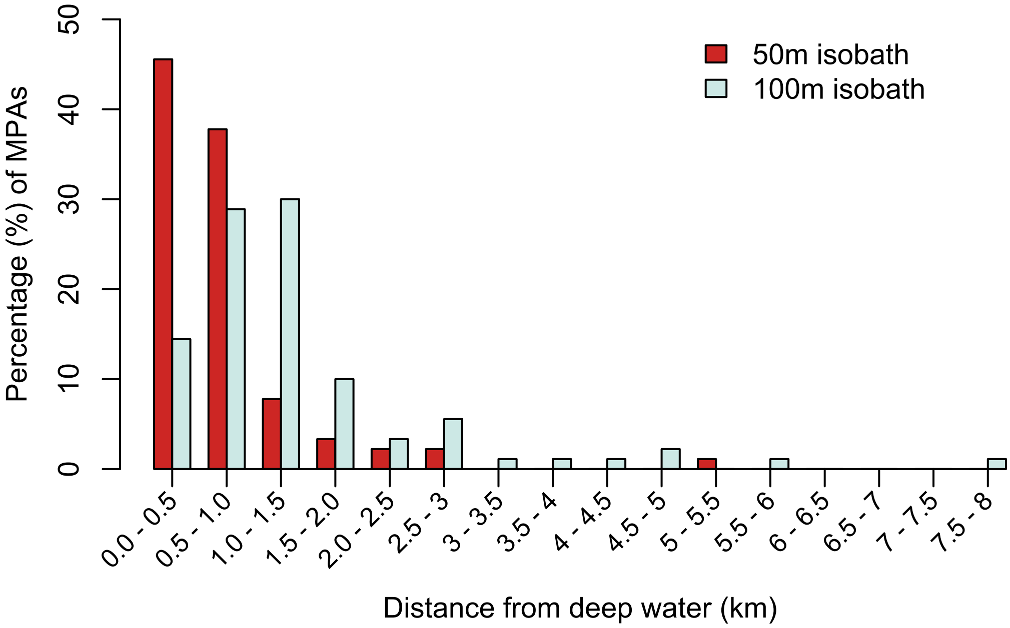


Figure 4.7. MPA distance from deep water: red bars show distance to the nearest 50 m isobath; blue bars to the 100 m isobath. Proximity to deep water is employed as an indicator of likely resilience to coral bleaching events.

The results presented here indicate that a system of MPAs that individually achieve many local-scale objectives for implementation will not necessarily function as a comprehensive, adequate and representative MPA network. Evidence from the literature indicates that MPAs in the Western Bohol Sea region are effective at achieving a range of local-scale objectives. Perhaps surprisingly, given that their location is typically driven by fisheries management objectives, MPAs appear to be equally capable of achieving local-scale objectives for biodiversity conservation. However, whilst the Western Bohol Sea MPA system performs well against criteria for connectivity, resilience and replication, it is not fully representative of the range of habitats and environmental conditions that occur in the region, and may not be adequate to ensure the persistence of biodiversity.

The location of MPAs is biased towards coral reef habitats in areas with low to moderate currents, shallower reef slopes and low exposure to river runoff. These biases likely reflect a focus on local-scale objectives during MPA establishment: sites that are dominated by seagrass, mangroves or lagoonal habitats, or those with turbid water are unlikely to generate income from dive tourism. At the municipal scale, MPA location appears to have been influenced by political, rather than ecological factors. The municipality of Dauin has established nine MPAs (only one coastal *barangay* does not have an MPA (Leisher et al. 2007)), partly as a result of a highly supportive mayor. Similarly, the election of unsupportive political leaders resulted in the removal of no-take status from the MPA at Sumilon Island in 1984 (Russ & Alcala 1999).

Small coralline and volcanic islands are very well represented in the MPA system. This is significant, given that they are likely to have high conservation value. Small islands may act as ‘stepping stones’ connecting fringing reefs on large islands (McCook et al. 2009), and often have environmental conditions that promote resistance to coral bleaching (West & Salm 2003). It may be easier to establish no-take MPAs on small islands due to their discrete geographical nature (White & Vogt 2000; Beger et al. 2005). Where coastal resources are exploited by a single community, motivation to protect those resources is greater, as there is a lower risk that benefits resulting from MPA implementation will be received by other communities that have not given up any of their fishing grounds (Foale & Manele 2004). It is likely that shoals are underrepresented in the MPA system for similar reasons: they are typically exploited by fishers from multiple communities, may be subject to unclear jurisdiction and their offshore location makes enforcement more difficult (McClanahan et al. 2006).

The widely applied MPA network design principle of protecting entire biological units is not generally practicable in the ecological or socioeconomic context of the Philippines. Given that much of the coastal environment comprises continuous fringing coral reefs, and coastal population density is consistently high, 'split zoning', where some parts of a reef are no-take and others are open to fishing, will be unavoidable. Moreover, it will not be socioeconomically feasible to include entire patch reefs, shoals or island fringing reefs in no-take MPAs. The coral reef area of Apo Island is just 0.7 km^2 (to the 20 m isobath) and Balicasag 0.3 km^2 (to 25 m) (Abesamis et al. 2006b). Although MPAs that protect the entire reef area of these islands would still be considered small by global standards (Wood 2008), both islands support c. 800 people, who are largely dependent upon fishing for income and subsistence.

Given the large number of MPAs in the region, it would be difficult not to achieve some level of connectivity between them. The spacing between MPAs is such that species with a wide variety of dispersal distances are provided for. Although it is generally recommended that bigger, rather than smaller areas be protected to allow for local-scale connections (McCook et al. 2009), recent empirical studies indicate that even MPAs smaller than 0.5 km^2 may be capable of self-replenishment (Jones 2005; Almany et al. 2007; Planes et al. 2009). The large number of MPAs also ensures reasonable levels of replication in the system, at least for coral reef and seagrass habitats.

To assess the extent to which individual MPAs achieve local scale objectives I reviewed existing literature. This approach has several limitations. Systematic reviews and meta-analyses are always limited by the quality and quantity of the original sources upon which they are based. Of the 90 MPAs in the Western Bohol Sea region, data were available for 40. Although less than 50% of MPAs are represented in this analysis, the number of data sources and focal MPAs is comparable to recent reviews of the effects of MPAs undertaken at continental and global scales (Claudet et al. 2008; Stewart et al. 2009). I am not aware of any similar studies that have analysed a community-based MPA system of this size. However, few of the studies that I analysed comprised comparisons of protected and unprotected areas through time. Many provided only temporal data from within MPAs, with no unprotected control site, or once-only spatial comparisons, that may be confounded by habitat differences between protected and control sites or the redistribution of fishing effort (Edgar et al. 2004b). Given that many of the sources are more than five years old, it should be noted that their results might not accurately represent current MPA effectiveness.

The majority of sources that I analysed, in particular those in peer-reviewed literature, presented positive evidence for MPA effectiveness (73 positive records, c.f. 11 negative and 32 inconclusive). Rather than indicating that MPAs in the Western Bohol Sea are generally successful at achieving a broad range of objectives, this result could be an artefact of reporting bias, as conservation failures are rarely recorded in the literature (Knight 2006).

By far the most evidence for the effectiveness of individual MPAs in the region comes from two MPAs, at Apo and Sumilon Islands. The next best-studied MPAs are those at Balicasag and Pamilacan. These four MPAs are atypical, in that they are located on small islands. Is there any evidence to suggest that the results observed at these sites are indicative of the effectiveness of the majority of MPAs that are situated on mainland fringing reefs? Russ et al (2005) compared relationships between the biomass of large predatory reef fish and duration of reserve protection from once-only surveys at 13 MPAs in the region to those measured over 18 years of temporal monitoring at Apo and Sumilon Islands. They found that inferred rates of recovery of coastal MPAs were similar to measured rates on offshore islands, indicating that the ecological effects observed at Apo and Sumilon may have generality for other MPAs in the region. However, it is unlikely that the significant social and economic benefits observed at Apo Island can be replicated for MPAs across the region. Vogt (1997) predicted that the extent to which coastal communities in the region may benefit from MPA-related tourism will vary, depending on site accessibility, infrastructure and the attractiveness of the MPA habitat to divers.

It is much easier to design comprehensive, adequate and representative MPA networks than to realise them within the constraints of implementation (Knight et al. 2006b; Leslie & McLeod 2007; Lowry et al. 2009; Banks & Skilleter 2010). Implementation is typically a sequential process (Meir et al. 2004), constrained by budget and site availability, and social, economic or political factors may prevail over site selections based on ecological representativeness (Margules & Pressey 2000). Biodiversity objectives frequently conflict with those that address socioeconomic, cultural or management feasibility perspectives, which were not considered here. Conservation planners attempting to achieve both sets of objectives simultaneously face trade-offs that will ultimately result in imperfect achievement of some, or all targets (e.g. Fernandes et al. 2005). Thus, it is important to note that failure to fulfil biodiversity objectives does not indicate failure of an MPA network *per se*. Assessments of effectiveness must consider the degree to which all objectives are fulfilled; for example, it would be preferable to narrowly miss all targets than to achieve one at the expense of all others.

To adequately represent the full range of habitats and environmental conditions occurring in a region, it will be necessary to establish MPAs in sites that are not 'attractive' to tourists, and therefore do not provide associated economic benefits. Correspondingly, it may be necessary to develop social networks and revenue-sharing mechanisms to ensure that the economic benefits of MPA implementation are equitably distributed amongst coastal communities that have committed to establishing no-take areas (Emerton et al. 2006).

Although community-based approaches have been successful in establishing MPAs to achieve local-scale objectives, a broader spatial context for planning will be required to develop ecologically functional MPA networks. Nevertheless, community-based approaches should not be abandoned in favour of regional-scale conservation planning. Local community support is essential to MPA success, and this will only be achieved if MPAs are designed to achieve local-scale objectives (Russ & Alcala 1996). Thus, the challenge is to develop MPA networks that simultaneously deliver local- and regional-scale benefits. To achieve this ambitious goal, conservation planners will need to better integrate social, economic and political considerations in the design of MPA networks, and adapt conservation planning frameworks and tools to the context in which they are to be applied. Some progress is being made in this respect (Knight et al. 2006b; Klein et al. 2008; Pressey & Bottrill 2008; Ban et al. 2009a). However, many commonly stated principles for MPA network design are not achievable, or appropriate, in the Philippines. Placing emphasis on unattainable targets, for example for the size of individual MPAs or the total area of habitat to be protected, may be counterproductive (Agardy et al. 2003; Carwardine et al. 2009). Instead, guidelines for MPA network development should be tailored to the region in which they are to be applied.

A key objective of the Coral Triangle Initiative is the development of MPA networks throughout the Philippines, Indonesia, Malaysia, Papua New Guinea, the Solomon Islands and Timor Leste (Coral Triangle Initiative 2008). Amongst these nations, the Philippines leads the way in coastal resource management and MPA implementation. Nevertheless, even here, MPA systems that have been established through community-based initiatives do not constitute ecologically functional networks. To achieve their objective, the CTI should seek to build capacity for regional-scale conservation planning within local NGOs. These groups benefit from established relationships with local communities and are best positioned to integrate regional-scale planning with local-scale implementation.

5. Shortcuts for marine conservation planning: the effectiveness of socioeconomic data surrogates

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5.1. Introduction

A unifying theme emerging from recent empirical studies of marine conservation initiatives is that socioeconomic, cultural and political factors are critically important to their success (Lundquist et al. 2005). For conservation plans to be implemented successfully, they must have support from stakeholders and local communities. This is especially true in regions where resources are insufficient for active enforcement of management initiatives, and compliance must therefore be voluntary (Walmsley & White 2003; Cinner et al. 2005; Alcala & Russ 2006; McClanahan et al. 2006).

One such region is the Coral Triangle (comprising Indonesia, Malaysia, the Philippines, Solomon Islands, Papua New Guinea and Timor Leste, Fig. 5.1). Widely acknowledged as a global priority for marine conservation (Roberts et al. 2002; Carpenter & Springer 2005), the region is home to 33% of the world's coral reefs, which, in addition to their biodiversity value, provide food security and livelihoods for millions of people (Coral Triangle Initiative 2008). No-take marine protected areas (MPAs) have become a key management tool to mitigate threats to marine resources in this region, both from the impacts of global climate change and local threats from overfishing, unsustainable fishing practices and habitat degradation (Govan et al. 2009; Weeks et al. 2010a). However, failure to adequately incorporate socioeconomic factors has resulted in poor management effectiveness of many MPAs (Christie & White 1997), and plans that fail to be translated into conservation action (Knight et al. 2008; Polasky 2008). As planners attempt to scale up existing MPAs into ecologically representative, connected and resilient MPA networks (Coral Triangle Initiative 2008; Lowry et al. 2009), they will need to carefully balance the potential biodiversity conservation benefits and socioeconomic viability of their designs (Klein et al. 2008).

Systematic conservation planning is the process of locating, configuring, implementing and maintaining areas that are managed to promote the persistence of biodiversity (Margules & Pressey 2000). This approach is often supported by software designed to assist planners to identify protected area networks that achieve quantitative targets for biodiversity representation for a minimal cost. In this context, 'cost' may refer to the

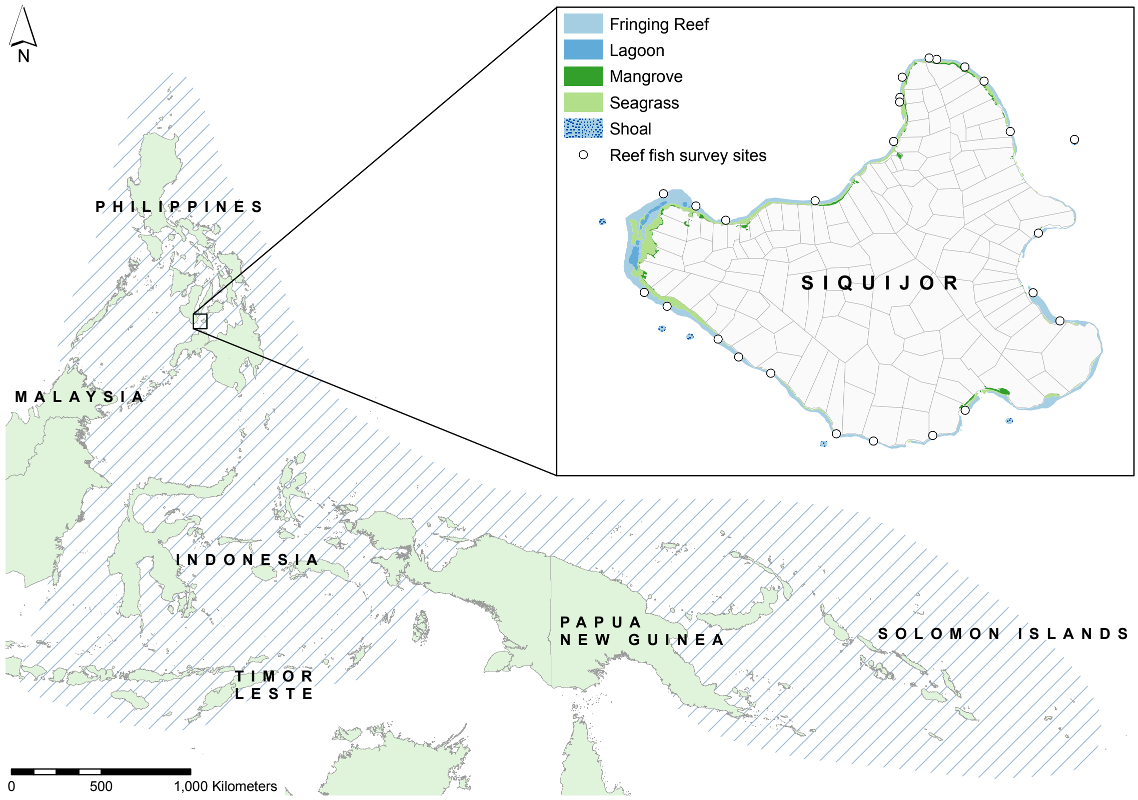

Figure 5.1. Location of the study site: Siquijor province, Philippines. Administrative subdivisions on Siquijor indicate the boundaries between *barangays* (the smallest political unit in the Philippines, analogous to a village or city ward). The shaded area shows the extent of the Coral Triangle region.

financial cost of acquiring, designating and managing protected areas or foregone opportunity costs to resource users (see Naidoo et al. 2006). Considering spatial variation in costs enables planners to use limited funds available for conservation to best effect, and to reduce social conflict by minimising the overlap between protected areas and resource use (Ban & Klein 2009).

Despite widespread agreement that they ultimately determine the success of conservation efforts (Sale et al. 2005), much research to date has neglected the social and economic aspects of planning (Naidoo & Adamowicz 2006). Early attempts at conservation planning considered socioeconomic factors as a post-hoc filter for areas identified using biological data alone (Stewart & Possingham 2005). Emphasis on the 'efficiency' of conservation plans (Pressey et al. 1993) typically focussed on minimising the financial acquisition cost of terrestrial protected area networks (Ando et al. 1998), or the area and boundary length of MPA networks (Leslie et al. 2003), as a crude surrogate for management costs. Only relatively recently have conservation planners turned their attention to the problem of socioeconomic viability, by considering opportunity costs to resource users to explicitly account for the social costs of establishing protected areas (e.g. Cameron et al. 2008; Klein et al. 2008).

Efforts to explicitly incorporate socioeconomic factors into the design of MPA networks are likely to be constrained by a lack of quantitative data (Naidoo & Adamowicz 2006; Carwardine et al. 2008). Conservation planning undertaken at regional scales allows for the consideration of complementarity (Vane-Wright et al. 1991) and connectivity (Almany et al. 2009) between protected areas. However, this approach also demands that biological and socioeconomic data are consistent in extent and resolution across the planning region, to avoid data-driven bias in site prioritisation (Grand et al. 2007).

Although many socioeconomic factors require consideration during the implementation of marine conservation initiatives, MPA location is primarily influenced by opportunity costs to resource users (Ban & Klein 2009). Several studies have demonstrated that incorporating data on the distribution of commercial and recreational fishing effort can substantially reduce economic losses incurred by the fishing industry as a result of MPA implementation (Stewart & Possingham 2005; Richardson et al. 2006; Klein et al. 2008). However, in the Coral Triangle, small-scale fishers are the primary stakeholders in conservation planning for coastal waters. The poor availability of data for small-scale fisheries in particular, compared to logbooks and records typically kept for commercial fisheries, is likely to frustrate conservation planners in the region (Russ 2002; Ban et al. 2009a).

Coastal resource use in the Coral Triangle is extremely complex; a large number of fishers from different communities employ different gear types to target different species. In this context, collecting empirical data on the spatial distribution of resource use at scales relevant to conservation planning is logistically challenging (Green et al. 2009). For this reason, 'shortcuts' that take advantage of existing data sources, such as population census data, are of interest. Just as biodiversity surrogates are employed to compensate for gaps in biological data (Rodrigues & Brooks 2007), socioeconomic surrogates can be used to represent the cost of conservation in the absence of empirical data (Ban & Klein 2009). Previous studies have used surrogates based on population pressure (Ban et al. 2009a) or the density of small boats (Sala et al. 2002) as a proxy for fishing pressure.

How well socioeconomic surrogates reflect the true cost of conservation action to stakeholders and communities is an important and unresolved question (Adams et al. 2009; Ban & Klein 2009). If simple functions of population or fishing boat registration data can predict the spatial distribution of fishing effort accurately, these data can (and should) be used to inform regional-scale conservation planning. However, if they represent real resource use patterns poorly, using such data may result in less effective conservation plans that do not earn community support. In this case, using coarse-resolution surrogates may be no better than assuming costs are homogenous, or proportional to the area protected (Richardson et al. 2006).

Much effort has been directed at assessing the adequacy of biodiversity surrogates in marine conservation planning (e.g. Ward et al. 1999; Gladstone & Alexander 2005; Beger 2007; Mumby et al. 2008). In contrast, where socioeconomic surrogates have been employed, they are typically assumed to be an accurate representation of true conservation costs. Although a few studies have compared different cost surrogates (e.g. Carwardine et al. 2008; Ban et al. 2009a), I am not aware of any that have tested their accuracy empirically.

I investigated the ability of socioeconomic surrogates to predict the opportunity cost to small-scale fishers of establishing an MPA network for Siquijor Province, Philippines. I compared MPA networks that were designed without consideration of socioeconomic data with those that incorporated four different surrogates for fishing effort and empirical data on the spatial distribution of fishing effort collected through interviews. My aim was to assess the effectiveness of each surrogate at predicting empirical cost, and their ability to reduce the impact of an MPA network on fishers.

5.2. Methodology

5.2.1. Study Area

My study area was Siquijor island, Philippines (Fig 5.1). More than two thirds of the population (c. 88,000) live in coastal *barangays* (the smallest political unit in the Philippines, analogous to a village or city ward), and small-scale fisheries contribute significantly to income and food security. Following the Philippine Local Government Code of 1991 (Republic Act 7160) and Philippine Fisheries Code of 1998 (Republic Act 8550), commercial fishers are excluded from operating within 15 km of the coastline. These coastal waters are reserved for small-scale fishers, who use small, typically non-motorised, outrigger boats and employ a large variety of fishing gears to target a diverse range of species. At present, Siquijor has 15 designated no-take MPAs, all smaller than 0.5 km^2 (Weeks et al. 2010a, Chapter 2).

5.2.2. Fishing Effort Data

Data on the number of fishers and boats, and the spatial distribution of fishing effort in each *barangay* were collected through semi-structured interviews (n = 57) with members of coastal communities. Time constraints limited my survey to one interview per *barangay*, and interviewees were asked to summarise patterns of fishing effort for all fishers in their community. Where possible, interviewers targeted those likely to have extensive knowledge of fishing practices throughout their *barangay*: members of a Fisherfolks' Association committee or *barangay* councillors. Interviews were conducted between April – July 2009 by A. Bucol, an ex-resident of Siquijor.

Interviewees were shown enlarged maps of Siquijor, with the location of *barangays*, prominent coastal features and marine habitats (coral reefs, seagrass, mangroves) marked. The maps were laminated to allow fishers to draw and write information directly onto the map. Where interviewees were not comfortable with this approach, they were prompted to describe the limits of fishing grounds verbally, and this information was transferred onto the map by the interviewer. Interviewees were first asked to estimate the total number of fishers in their *barangay* and the number of motorised and non-motorised boats. They were then asked to identify the locations of fishing grounds used by members of their community on the map, and to estimate the number of fishers from their *barangay* using each site.

5.2.3. MPA Network Selection

I used the conservation planning software Marxan (Ball & Possingham 2000; Possingham 2000) to identify MPA networks that fulfilled a conservation objective whilst minimising foregone opportunity costs to small-scale fishers. Marxan uses a simulated annealing algorithm with iterative improvement to generate multiple MPA network solutions that satisfy specified conservation objectives for a low cost. These solutions typically have different spatial configurations, providing planners with flexibility. In addition to identifying the 'best' MPA network for each scenario (that which meets targets for the lowest cost), Marxan reports selection frequency scores for each planning unit: the number of times each planning unit was selected in (in this instance) 1000 runs of the algorithm. Selection frequency scores provide a useful indication of the conservation importance of each planning unit (Stewart et al. 2007).

MPAs in the Philippines are typically established to conserve coral reef associated habitats and ensure the sustainability of the fisheries dependent upon them. To date, no MPAs have been established to target open water habitats (Weeks et al. 2010a, Chapter 2). I thus defined my planning region by the extent of targeted conservation features (see below, and Fig 5.1).

I divided the planning region into regular hexagonal planning units of 0.05 km^2. This planning unit size was selected to be at a scale relevant to management: the median size of no-take MPAs in the Philippines is 0.12 km^2, and the minimum for Siquijor, 0.04 km^2 (Weeks et al. 2010a, Chapter 2). Thus, a single planning unit, or two contiguous units would be a typical size range for a no-take MPA in the region. Socioeconomic constraints, including the low spatial mobility of fishers, make the implementation of larger MPAs less feasible. Consequently, in contrast to conservation planning studies located in developed countries (e.g. Fernandes et al. 2005; Klein et al. 2008), a scattered system of small MPAs is more practical than fewer, larger ones. For this reason, I did not use Marxan's 'boundary-length modifier' (Possingham 2000) to express a preference for spatially compact MPA networks.

Following the Philippine Marine Sanctuary Strategy (Arceo et al. 2004), which has been adopted by the Coral Triangle Initiative (Coral Triangle Initiative 2008), my conservation objective required representation of 10% of the area of coral reef-associated habitat types identified from satellite imagery: fringing reefs, sunken shoals, seagrass, lagoons and mangroves. I identified six distinct reef fish 'bioregions' using multivariate statistics on underwater visual census data collected at 26 survey sites around the island (Fig. 5.1, Chapter 3). I required 10% of each habitat type to be represented in each bioregion, resulting in 22 targeted conservation features (not all habitat types were present in all bioregions). I used ArcGIS to calculate the area of each conservation feature within each planning unit. Cost values were then assigned to planning units as described below (section 5.2.4).

5.2.4. Data Scenarios

I implemented Marxan for six different reserve selection scenarios, each with different socioeconomic costs (Table 5.1). I compared a scenario in which socioeconomic data were not considered (scenario *a*) with four different surrogates of small-scale fishing effort (scenarios *b* – *e*) and empirical data for the spatial distribution of fishing effort (scenario *f*; Table 5.1). I selected socioeconomic surrogates that could be derived from population census data (scenario *b* and *c*), as these data are typically available even in otherwise data-poor regions. I also tested surrogates based on the number of fishers or boats (scenario *d* and *e*): these data may be pre-existing, and are otherwise relatively cheap and quick to collect (compared to mapping the spatial distribution of fishing effort). Fishing effort surrogates were intended to represent the foregone opportunity cost to small-scale fishers if that site were to be designated as a no-take MPA. I assumed that minimising opportunity costs to fishers would increase the likelihood that they would support and comply with MPA implementation, resulting in more effective conservation. As my planning region was limited to inshore coastal waters, I did not consider the interests of commercial fishers as stakeholders.

For scenario *a*, costs were considered to be spatially homogenous, with the cost of each planning unit (each site available for inclusion in the MPA network) equal to its area. Under this scenario, sites were selected based on their biodiversity value alone, with no explicit consideration of socioeconomic values, and the site-selection objective was to minimise the total area of the MPA network. For all other scenarios, the objective was to reduce the cost of the MPA network. For scenarios *b* and *c*, costs were modelled from population census data (http://www.census.gov.ph/; accessed May 2009), as a function of *barangay* population (scenario *b*) or coastal population density (scenario *c*). Coastal population density was derived by dividing *barangay* population by the length of the *barangay* coastline, calculated in ArcGIS (version 9.2; ESRI, Redlands, California). Cost values were assigned to planning units using a linear distance decay, such that units closer to the coastline had a higher cost than those further offshore (out to 10 km) and units adjacent to *barangays* with larger populations had greater cost than those adjacent to smaller populations. For scenario *d*, costs were modelled from the estimated number of fishers (from interview data) in each *barangay*, following the same approach. In scenario

Table 5.1. Summary of socioeconomic data scenarios

Scenario	Cost value assigned to each planning unit
a (area)	Area [a]
b (population)	Linear distance function of barangay population, with influence distance of 10 km
c (coastal density)	Linear distance function of coastal population density, with influence distance of 10 km
d (fishers)	Linear distance function of estimated number of fishers per barangay, with influence distance of 10 km
e (boats)	Function of estimated number of fishing vessels of different types, and the spatial mobility of that vessel type (motorised boats = 10 km, non-motorised boats = 3 km, beach-based gears = 1 km)
f (empirical)	The estimated number of fishers using each site, derived from interviews

[a] (*Planning units were typically 0.05 km^2 except at the land-sea interface where units were clipped to the coastline, resulting in smaller units)*

e, costs were modelled as a function of the estimated number of fishing vessels (from interview data) of different types and the distance typically travelled using that vessel type. Based on responses given during interviews, fishers using motorised boats were assumed to have spatial mobility of 10 km, those using non-motorised boats 3 km, and beach-based gears (beach-seine, gleaners and spearfishers) 1 km.

Finally, scenario *f* used empirical cost values: the number of fishers using each site, from interview data. Spatial information collected during interviews was digitised in ArcGIS. Each fishing ground polygon was assigned a score of the estimated number of people fishing there; scores were summed for overlapping polygons, and then rescaled to match the other cost layers (all costs were relative, on a scale of 0 – 1000). The spatial precision of fishing ground polygons identified during interviews was variable: whilst some interviewees identified precise locations of favoured sites, others gave a broader summary of the areas fished by members of their community. This variability resulted in sharp gradients in cost values at the edges of fishing ground polygons, which I felt did not accurately represent real spatial resource use and would have excessively influenced my results. To reduce this effect, I smoothed the cost values for the empirical data scenario using the Neighbourhood Statistics tool in the Spatial Analyst extension for ArcGIS. In all cases, cost values represent a relative measure of the importance of each site to small-scale fishers, rather than an estimate of economic value.

My surrogate models would be unable to predict the absence of fishing effort within existing no-take MPAs or increased effort along their boundaries (Alcala & Russ 2006). Thus, including areas of zero fishing effort inside existing MPAs in my empirical data would have resulted in large differences between predicted and empirical values at those sites. To avoid this bias, which may have excessively driven my results, I did not include existing MPAs in my empirical maps of fishing effort and redistributed effort in adjacent fished areas across the MPAs.

5.2.5. Analyses

Effectiveness of surrogates at predicting empirical cost

I assessed the ability of surrogates to predict the spatial distribution of fishing effort by comparing the cost values assigned to planning units under each data scenario with pairwise Spearman's rank correlation (ρ). All statistical analyses were undertaken with the statistical software package '*R*' (R Development Core Team 2008).

Comparing MPA networks

For each scenario, I categorised the conservation importance of planning units based on their Marxan selection frequency: units selected no more than would be expected by chance (1 standard deviation to the right of the mean) were considered of low importance; those selected slightly more than by chance (2 standard deviations to the right of the mean) of medium importance; and those selected much more frequently than chance (the rest of the tail) were considered highly important.

I used two methods to compare the spatial similarity of Marxan outputs for the different scenarios (i.e. how similar the planning units selected for one network are to the planning units in another network). First, I examined the overlap of conservation importance categories for each scenario using Cohen's Kappa statistic (κ): a chance-corrected measure of spatial agreement. Kappa values range from +1, indicating complete agreement (or perfect overlap between categories), to -1, indicating complete disagreement. A Kappa statistic of 0 indicates overlap due to chance. Second, I compared the selection frequency of planning units under different surrogate data scenarios using hierarchical cluster analysis (Euclidean distance, Ward's clustering method).

Effectiveness of surrogates at reducing the cost of MPA networks

To assess the ability of different socioeconomic surrogates to reduce the impact of an MPA network on fishers, I calculated the mean cost of MPA networks identified by Marxan as the sum of the empirical cost of selected planning units. I tested the significance of cost differences with one-way analysis of variance (ANOVA) and post-hoc Tukey's honest significant differences (HSD) tests.

Existing MPAs

Two assumptions are central to my analysis: first, that the spatial distribution of fishing effort accurately represents opportunity costs to small-scale fishers; and second, that minimising opportunity costs to fishers increases the likelihood of successful MPA implementation. I tested these assumptions using the hypothesis that existing MPAs would have a lower cost than other sites under the empirical data scenario. Existing MPAs have been established following an opportunistic approach, with locations determined largely by the likelihood of successful implementation; I therefore expect that these sites would have lower fishing pressure (causing less conflict) than other areas.

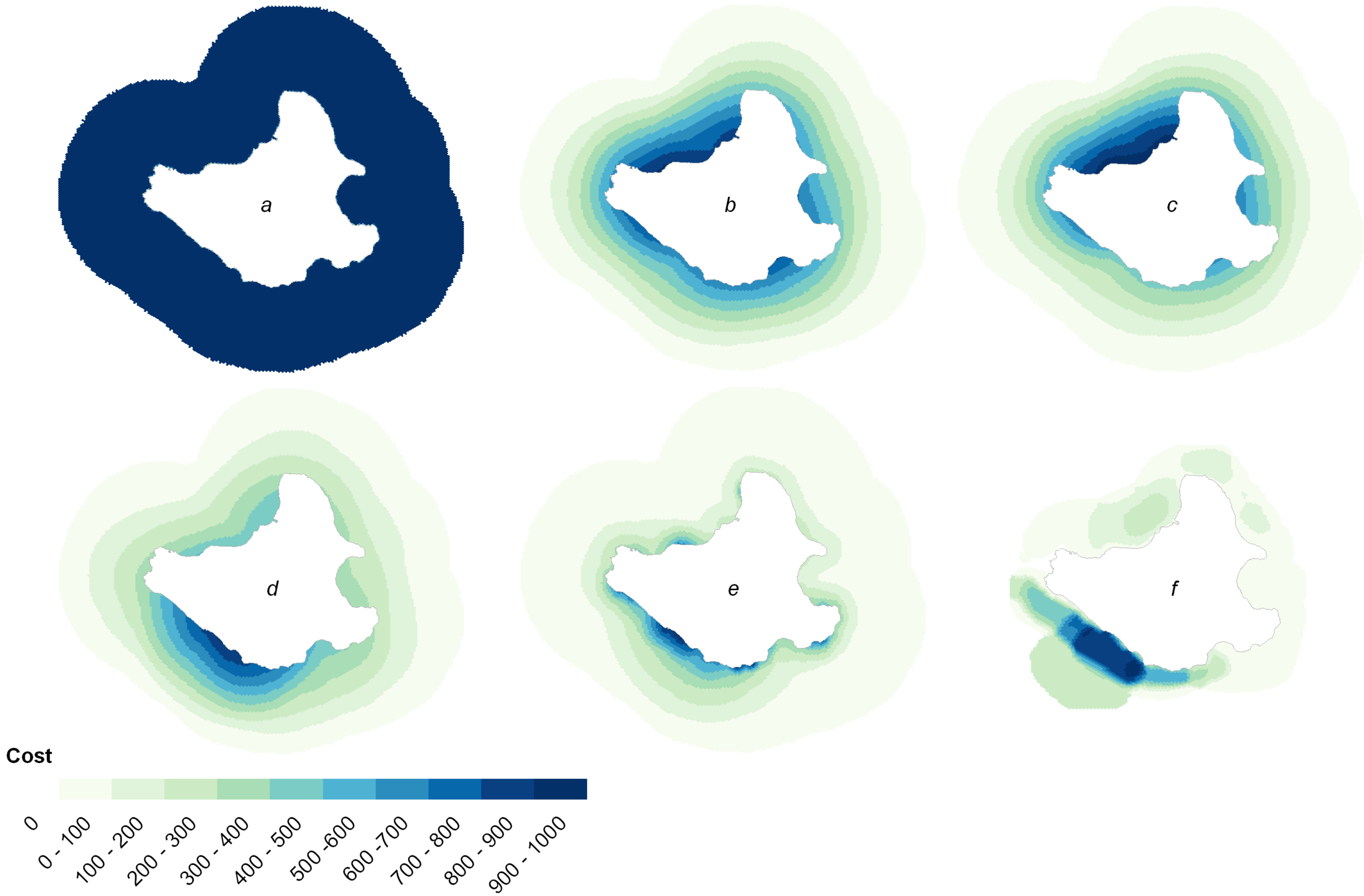

Figure 5.2. The foregone opportunity cost to small-scale fishers of MPA implementation under different socioeconomic data scenarios used in analyses: area (*a*), population (*b*), coastal population density (*c*), fishers (*d*), boats (*e*) and empirical data (*f*). In all cases, cost values are relative, and not intended to represent the economic value of a site.

5.3. Results

5.3.1. Effectiveness of surrogates at predicting empirical cost

Surrogates varied in their ability to predict empirical cost patterns (Fig. 5.2). Population (scenario *b*) and coastal population density (scenario *c*) were poor predictors of the spatial distribution of fishing effort (ρ = 0.19 and ρ = -0.14 respectively, $p < 0.001$ for both). These scenarios predicted greatest cost values along the north-west coast of Siquijor, whereas empirical data showed that fishing pressure was, in fact, highest along the south-west coast (Fig. 5.2). Surrogates based on the number of fishers (scenario *d*) and boats (scenario *e*) were positively correlated with empirical cost, although not strongly so (ρ = 0.62 and ρ = 0.55 respectively, $p < 0.001$ for both).

5.3.2. Spatial similarity of MPA networks identified using different surrogates

All data scenarios produced MPA network solutions that achieved the conservation objective (Table 5.2). Kappa statistics indicated that scenarios *e* (boats) and *d* (fishers) were most similar to empirical data in terms of the conservation importance of planning units (κ =0.377 and κ =0.312 respectively, $p < 0.001$ for both; Table 5.3, Fig. 5.3). MPA networks designed without socioeconomic data (scenario *a*) showed no spatial agreement in selection frequency categories with those based on empirical data (κ = -0.058, $p < 0.05$).

Hierarchical cluster analysis of Marxan results for each scenario (Fig. 5.4) indicates that the spatial pattern of selected sites varied greatly between scenarios; clustered solutions have similar spatial configurations of selected sites. With the exception of scenarios *b* (population) and *c* (coastal density), which cannot be distinguished, each of the socioeconomic data scenarios forms a distinct cluster. The empirical data scenario cluster appears as an outlier, indicating low levels of similarity with MPA networks identified using any of the surrogates (Fig. 5.4).

5.3.3. Effectiveness of surrogates at reducing the cost of MPA networks

MPA networks identified without socioeconomic data (scenario *a*) cost up to five times as much as those designed using empirical data (Fig 5.5). All of the surrogates that I tested improved on this scenario (Fig. 5.5, Table 5.2). Scenario *e* (boats) was the most cost-effective for stakeholders; however the mean cost of MPA networks for this scenario was still 183% of the least costly network using empirical data. All of the surrogate scenarios were significantly more costly to fishers than the MPA networks identified using empirical data (one-way ANOVA, $p<0.001$).

Table 5.2 Summary of Marxan results for different surrogate data scenarios. Values shown are means for 1000 replicate runs.

Scenario	Objective function score [a]	Cost	Number of planning units	Penalty [b]	Shortfall [c]	Missing Values [d]	Total Area (km^2)	Empirical cost
a (area)	63097.59	63094.84	101.54	2.75	0.38	0.07	3.15	9718
b (population)	38176.98	38175.24	66.89	1.74	0.38	0.03	3.17	6080
c (coastal density)	31746.30	31744.76	67.43	1.53	0.39	0.04	3.18	6984
d (fishers)	24311.63	24310.86	66.88	0.77	0.35	0.02	3.17	4649
e (boats)	17965.19	17964.57	68.44	0.62	0.29	0.02	3.25	4348
f (empirical)	2775.93	2775.77	70.66	0.16	0.30	0.03	3.29	2776

[a] The objective function score is the sum of the cost of selected planning units and the penalties for not meeting all targets. Lower scores indicate 'better' solutions.
[b] Penalty scores are accrued if feature targets are not met.
[c] Shortfall is equal to the sum, across all features that have not met their target, of the target minus the amount in the solution. If targets are not met, the shortfall indicates how close the solution was to achieving the target.
[d] The number of features whose representation targets were not met.

Table 5.3. Cohen's Kappa statistics (κ)[a] for spatial agreement between selection frequency categories under each cost scenario

Scenario	*a* (area)	*b* (population)	*c* (coastal density)	*d* (fishers)	*e* (boats)
b (population)	-0.070 **				
c (coastal density)	-0.063 **	0.752 ***			
d (fishers)	-0.072 **	0.540 ***	0.397 ***		
e (boats)	-0.064 **	0.425 ***	0.357 ***	0.461 ***	
f (empirical)	-0.058 *	0.216 ***	0.157 ***	0.312 ***	0.377 ***

[a] A Kappa statistic of 1 indicates complete agreement, 0 indicates overlap due to chance.

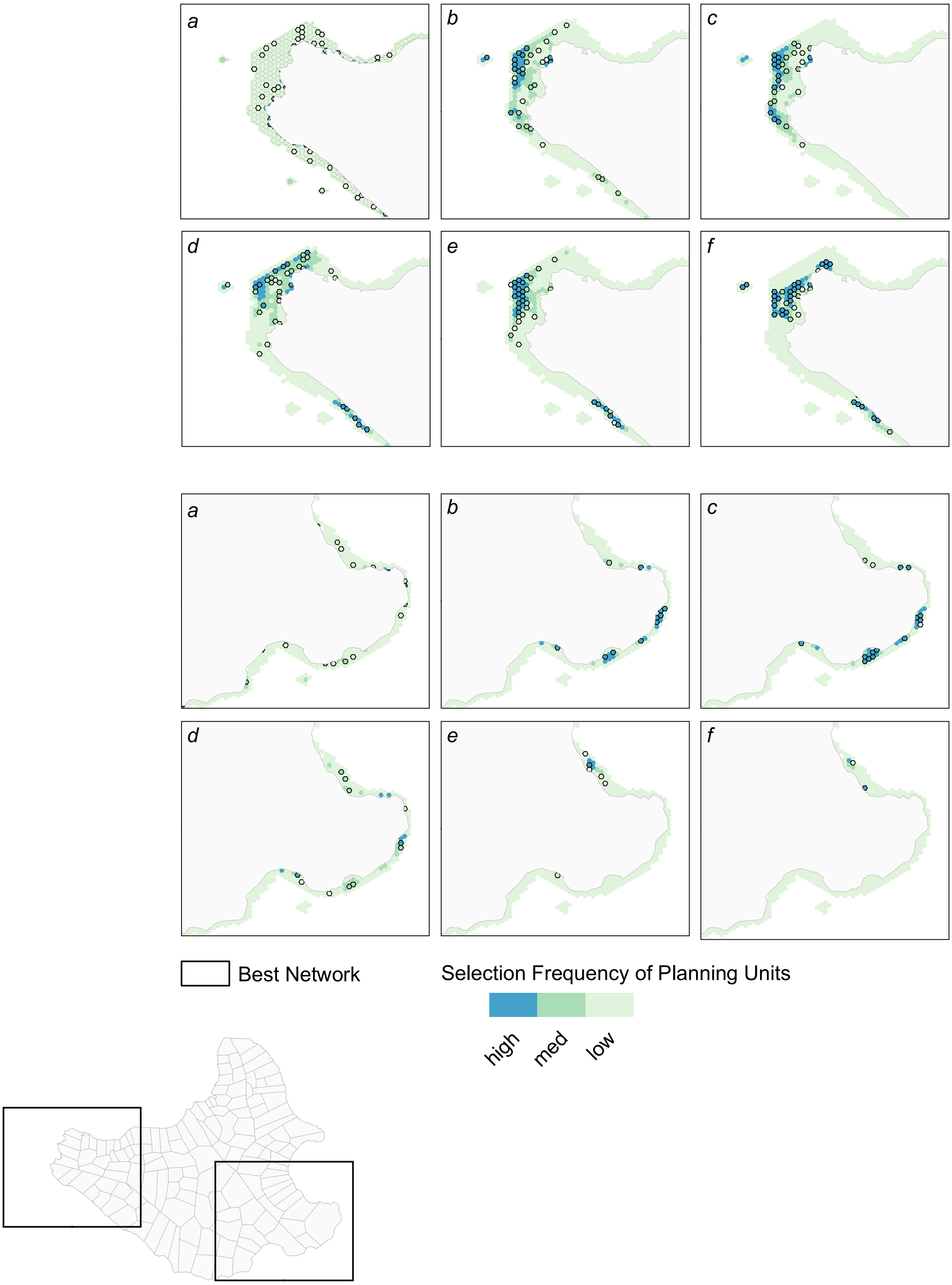

Figure 5.3. Marxan results for two parts of the study region: 'best' networks and selection frequency of planning units under different socioeconomic data scenarios: area (*a*), population (*b*), coastal population density (*c*), fishers (*d*), boats (*e*) and empirical data (*f*).

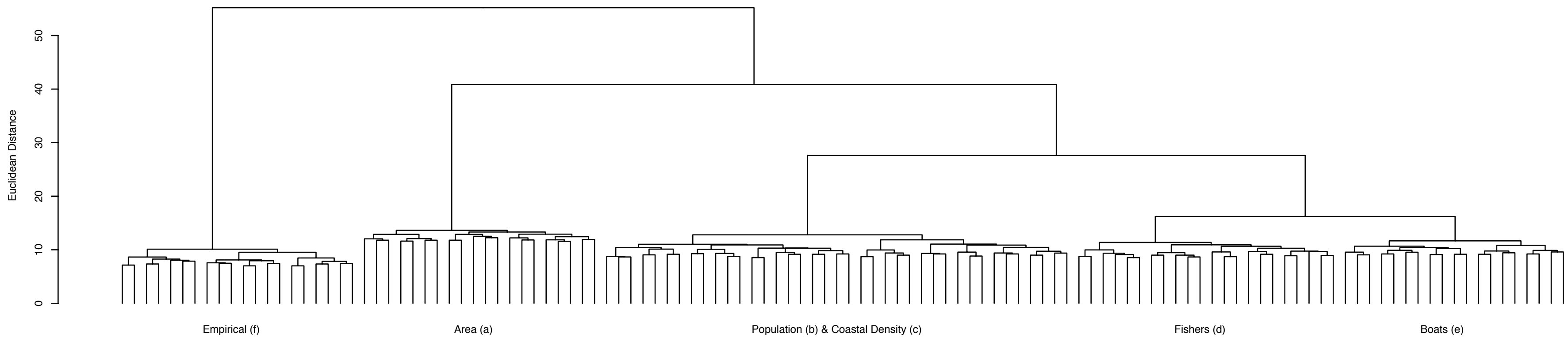

Figure 5.4. Comparison of Marxan solutions from different surrogate data scenarios using hierarchical cluster analysis. For clarity, only 20 solutions are shown for each scenario; the groupings remain the same when all solutions are included. Labels correspond to the surrogate data scenarios: area (*a*), population (*b*), coastal population density (*c*), fishers (*d*), boats (*e*) and empirical data (*f*).

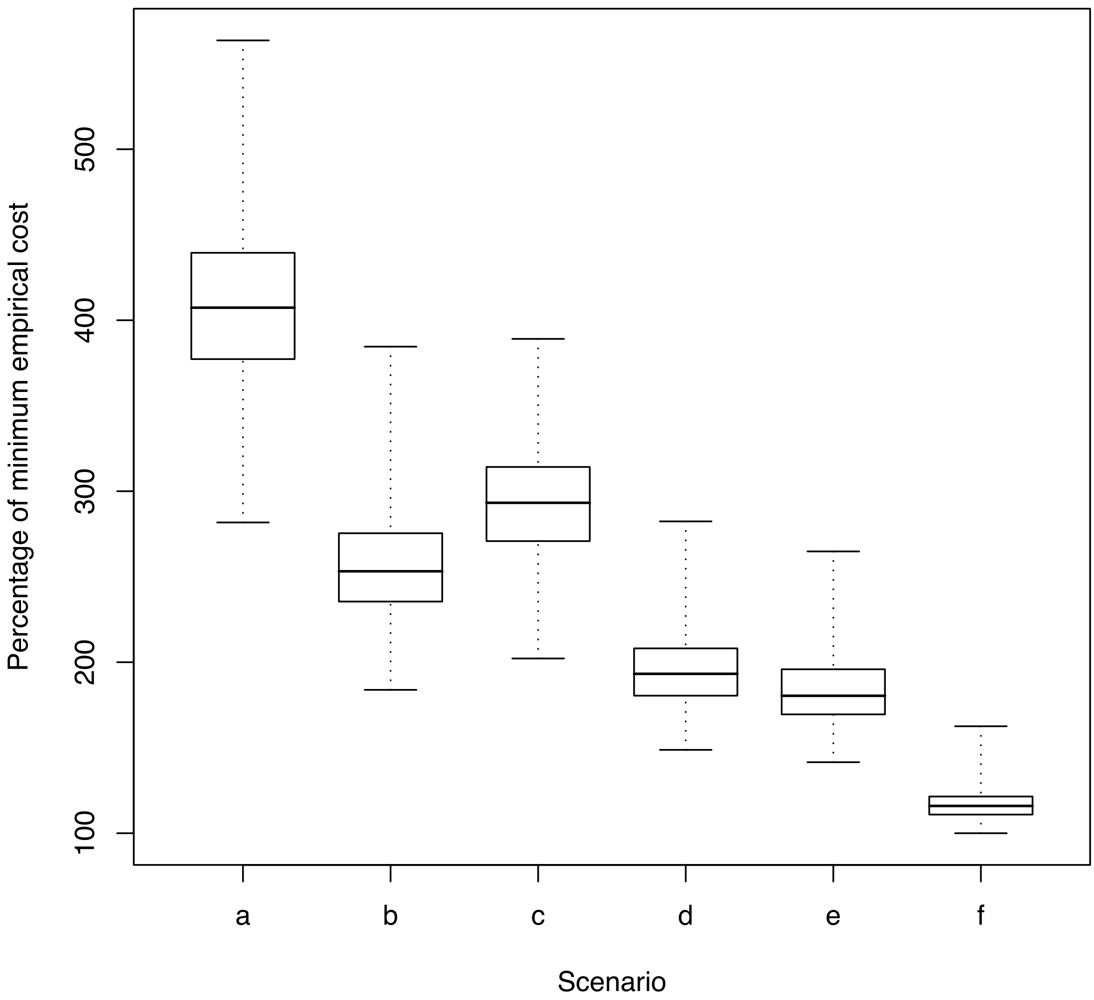

Figure 5.5. The total empirical cost of MPA networks designed using different surrogates, expressed as a percentage of the lowest cost network identified using empirical data; *n*=1000 for each scenario. Labels correspond to the surrogate data scenarios: area (*a*), population (*b*), coastal population density (*c*), fishers (*d*), boats (*e*) and empirical data (*f*). Boxes show the inter-quartile range bisected by the median; whiskers show the full data range.

5.3.4. Existing MPAs

I found that under the empirical data scenario, planning units containing existing MPAs (*n* = 45) had significantly lower cost values than those that were not currently protected (t = -2.6547, df = 553.785, p <0.001). However, these planning units did not have a higher selection frequency than other sites (t = -2.385, df = 55.493, p-value = 0.9897).

5.4. Discussion

Effective conservation of marine resources in the Philippines, and throughout the Coral Triangle region, will be achieved only through local community support for management actions (Alcala & Russ 2006; Green et al. 2009). One of the many challenges that face planners attempting to scale up existing, largely community-based, conservation efforts into effective regional MPA networks is the lack of informative socioeconomic data at the appropriate scale (Ban et al. 2009a; Green et al. 2009). I assessed the ability of different opportunity cost surrogates to reduce the negative impacts of MPA network implementation on small-scale fishers, the primary stakeholders in coastal waters of the Coral Triangle.

Opportunity costs to resource users are the principal costs involved in conserving marine ecosystems, where acquisition and transaction costs (Naidoo et al. 2006) are atypical, due to the common-pool nature of marine resources (Ostrom et al. 1999). I did not consider spatial variation in management costs. Management costs associated with MPA establishment and infrastructure (e.g. marker buoys) are unlikely to exhibit spatial variation at this scale. Although it has been assumed that enforcement costs are reduced when MPAs are within sight of a community (McClanahan et al. 2006; Ban et al. 2009a), existing MPAs in Siquijor show exceptions to this rule: at least two MPAs are located away from their *barangays*, with enforcement achieved through 24-hour surveillance by volunteers stationed in a guard house.

My study is the first to empirically test the adequacy of surrogates for small-scale fishing effort in conservation planning. However, my empirical data are not without their limitations. Time constraints allowed only one interview per community, and interviewees were asked to summarise patterns of fishing effort for all fishers in their community. Anecdotal evidence suggests that interviewees were able to locate fishing grounds accurately: the locations of offshore reefs mapped during interviews matched GPS points provided for these sites by a local dive center almost exactly. Nevertheless, further research is needed to test how well my 'empirical' data correspond to resource use patterns. Ideally, this would be done through direct observations of fishing activity. However, intensive surveys conducted in a subset of *barangays* might be sufficient to determine the accuracy of the dataset. Fishers' spatial preferences were typically not described at a sufficiently fine scale to guide the placement of individual MPAs. In many cases, interviewees identified all of the reef area adjacent to their *barangay* as a fishing ground. In order to place a no-take MPA within that area, more comprehensive consultation with communities would certainly be required.

In agreement with other studies, I found that using spatially homogenous costs (i.e. area) resulted in MPA networks that had a greater impact on resource users than when more specific cost measures were used (Stewart and Possingham 2005; Klein *et al.* 2008a). Socioeconomic surrogates differed in their ability to predict spatial patterns of fishing effort and to reduce the costs of MPA network implementation to stakeholders. Surrogates based on the number of fishers or boats in each *barangay* outperformed those based solely on population census data. My findings support those of Carwardine et al. (2008) who warn that using inappropriate cost measures can lead to costly conservation mistakes. In this context, using population density as a surrogate would direct conservation planners towards locating MPAs in areas that may not be supported by local communities.

It might be argued that seeking to place MPAs in areas with relatively low fishing pressure amounts to little more than 'residual conservation': protecting the areas that least need protection (Pressey 2009). However, it should be noted that a key constraint of the reserve selection process using Marxan is that MPA networks must achieve the specified conservation target. Thus, sites with high fishing pressure are not avoided at all cost. If a conservation feature is spatially associated with areas of high fishing pressure it will still be represented in the MPA network, even though important fishing grounds may be lost. Attempting to minimise opportunity costs ensures that where there is a choice of sites to achieve the conservation objective, the one that will cause the least conflict with fishers is selected. Nevertheless, whilst the MPA networks identified here achieve targets for habitat and bioregion representation, they do not consider other ecological design principles, for example those concerning adequacy, replication, connectivity and resilience (see Chapter 4). These factors are beyond the scope of this analysis, but must be taken into account in 'real world' conservation planning.

None of the surrogates that I tested provided an accurate representation of resource use at fine spatial scales. Nevertheless, my results indicate that models based on the number of fishers or boats can assist in identifying regional-scale conservation priorities. These could be used to identify areas with relatively low fishing pressure, where conservation objectives may be met more easily (Green et al. 2009).

Philippine legislation states that local government units should maintain a registry of municipal fisherfolk and fishing vessels three gross tons and below (Executive Order 305). However, I found that these data were not available at *barangay* level for Siquijor, prompting my surveys. The national legislation does not make it clear which local authority should be responsible for keeping these records; the Municipal Agricultural Offices on Siquijor undertook fishing vessel registration prior to 2000, but these records have not been updated in recent years (A. Bucol, personal communication). Additionally, interviewees indicated that registration fees present a disincentive for fishers to register their boats. My results indicate that, in the absence of empirical data on fishing effort, this information can be used to design conservation plans that reduce opportunity costs to small-scale fishers, which I believe will result in greater community acceptance and likelihood of successful implementation. I therefore recommend that attempts be made to establish and strengthen legislation requiring the collection of fisher and boat registrations, and to provide better incentives for fishers to obtain the relevant licenses.

The effectiveness of socioeconomic surrogates is likely to be sensitive to the spatial scale at which analysis is undertaken. At large spatial scales population may be a good indicator of fishing pressure: data for the number of fishers in each province of the Central Visayas region of the Philippines (Green et al. 2004) indicates a very strong positive relationship between population size and fishing effort at this scale. However, at the local (*barangay*) scale, I found that surrogates based on population data predicted the spatial distribution of fishing effort poorly. In rural areas with lower population density, a high proportion of households are dependent upon fishing as their primary source of income, as there are few opportunities for alternative employment. Interviewees in more urbanised *barangays* indicated that fewer people were engaged in fishing because other livelihoods, such as construction or small-scale enterprise, were more profitable. Some also suggested that lower water quality in areas adjacent to population centres, resulting from the use of detergents and chlorine-based products to wash clothes, were a disincentive to fishing. The fine spatial scale at which coastal resource management is undertaken in the Coral Triangle region requires that conservation planners consider which surrogates are likely to best represent the opportunity costs of conservation at this scale.

As per previous studies (Carwardine et al. 2008; Ban et al. 2009a), I found that conservation priorities emerged as increasingly complex socioeconomic data were incorporated in MPA network design. My conservation objective to represent 10% of reef-associated habitats in MPAs is relatively modest (yet realistic). Sensitivity analyses (not shown here) demonstrated that with higher representation targets, conservation priorities, and differences between the surrogate scenarios, became more pronounced. In regions where the opportunity cost of sites is more variable than their biodiversity value (either real, or perceived due to data limitations), costs should be used to drive the selection of sites for conservation (Naidoo et al. 2006; Perhans et al. 2008). Grantham et al. (2008) found strongly diminishing return on investment for biodiversity survey data: increasing data collection effort did not result in significantly more effective protected area networks. My experience suggests that the quality of socioeconomic data may be improved more quickly and economically than biodiversity data, and may have a more significant impact on the end-result of conservation plans.

Planning units containing existing MPAs were less costly than other sites under the empirical data scenario, indicating that my assumption that patterns of fishing effort reflect the cost of MPA implementation to coastal communities is valid: MPAs have been established in areas with relatively low fishing pressure. However, existing MPAs were not selected for inclusion in MPA networks more than would be expected by chance, indicating that they are not placed in areas of high conservation importance. This is likely a result of the approach to MPA implementation taken in the region, which has been characterised by opportunistic, site-level initiatives, rather than regional-scale systematic site-selection exercises, as I have performed here (Lowry et al. 2009). Nevertheless, although the existing MPAs on Siquijor may not be optimally sited for regional-scale conservation, they may provide local-scale benefits for fisheries management, conservation and socioeconomic development (e.g. income from tourism).

Despite a lack of formal marine tenure (as seen in the Solomon Islands and Papua New Guinea, amongst others (Foale & Manele 2004)), I found that fishers utilising coral reef or seagrass habitats typically identified fishing grounds only in areas adjacent to their *barangay*. Fishers using hook and line in open water travelled beyond their *barangay*, but rarely identified fishing grounds outside of their municipality. This has implications for the way that opportunity costs of conservation should be incorporated into conservation planning. Here, I sought to minimise the opportunity costs to small-scale fishers as a single stakeholder group. However, in this context it may be more important to minimise costs to individual communities separately (Ban & Klein 2009). The effect of local marine tenure on regional-scale conservation planning has not yet been investigated, and should be a priority for future research.

6. Incorporating local tenure in the systematic design of marine protected area networks

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6.1. Introduction

In the last two decades, systematic conservation planning (Margules & Pressey 2000) has evolved from a largely academic discipline to influence conservation action on the ground and in the sea (e.g. Fernandes et al. 2005; Klein et al. 2008; case studies in Pressey & Bottrill 2009). Nevertheless, in many regions of the world where conservation action is urgently needed, systematic planning has had relatively little influence on local-scale management initiatives. One reason for this planning-implementation gap (Knight et al. 2008) is the mismatch between the regional scales at which planning has typically been undertaken, and the local scale at which implementation occurs. Another is that the social, economic and political context in which systematic conservation planning methods and tools have been developed is different to that in which we now need to apply them (Christie et al. 2007; Cinner 2007). The fine spatial scale of governance, existence of formal or informal local tenure, and limited spatial mobility of small-scale fishers (who are often the primary stakeholders for conservation in coastal waters) all present new challenges to the application of conservation planning.

How resource tenure systems shape conservation outcomes in different social and ecological contexts has been identified as one of 100 questions of conservation importance (Sutherland et al. 2009). Local or customary marine tenure has been documented worldwide (Johannes 2002; Cinner & Aswani 2007), and often coincides with forms of customary management (see Cinner & Aswani 2007). Whilst there has been much discussion in the literature as to whether no-take marine protected areas (MPA) are an effective management tool in this context (Foale & Manele 2004; Cinner & Aswani 2007), less attention has been paid to the effect that constraints on spatial resource use brought about by local tenure may have on spatial planning and the design of regional-scale MPA networks (but see Aswani & Lauer 2006; Green et al. 2009).

In regions with local marine tenure, fishing rights are managed by individual communities, often at spatial scales of hundreds of meters to a few kilometers of coastline. This is in contrast to most developed countries, where coastal and marine resources are governed at national or state level. Under local tenure systems, fishers' rights are not as strong, or frequently exercised beyond the boundaries of their own community (Foale & Manele 2004). Thus, local tenure acts to restrict fishers' spatial mobility, as they may not be able to redistribute effort to areas outside of their community following MPA implementation. Whereas commercial fishers might incur increased operating costs associated with travelling to more distant fishing grounds, and opportunity costs if their catch per unit effort in those areas is lower (Ban & Klein 2009), small-scale fishers in regions with local tenure might lose access to fishing grounds entirely if large contiguous areas are protected (Aswani & Hamilton 2004b). Consequently, if they are to be socioeconomically viable, the design of MPA networks needs to consider the spatial scale and location of marine tenure boundaries (Aswani & Hamilton 2004a; Foale & Manele 2004; Cinner 2007).

Previous attempts to reduce the impact of MPA network implementation on small-scale fishers have used population pressure (Ban et al. 2009a), the density of small boats (Sala et al. 2002) or numbers of fishers (Weeks et al. 2010b) as a proxy for opportunity costs. These examples do not account for constraints on the spatial distribution of fishing effort, which can occur either as a result of local tenure or of the limited spatial mobility of small, non-motorised fishing vessels. Approaches that seek to minimise impacts on small-scale fishers as a single stakeholder group are likely to prioritise sites for protection that will disproportionately impact some resource users, leaving others unaffected. Such inequitable distribution of the costs and benefits of conservation among stakeholders may result in social or political conflict, failure during implementation or poor compliance (Cinner 2007; Klein et al. 2009).

There are few examples where local marine tenure has been explicitly incorporated into regional-scale conservation planning. In the development of an MPA network for the Roviana and Vonavona Lagoons, Solomon Islands, knowledge of customary tenure boundaries was used to identify sites at which conflict over natural resources was likely to be minimal, and MPA implementation more successful (Aswani & Lauer 2006). In Kimbe Bay, Papua New Guinea, information on customary marine tenure boundaries was taken into account when refining an MPA network initially designed using reserve-selection software (Green et al. 2009). Whilst adjustments to the design of MPA networks will always be required prior to implementation, plans that incorporate the constraints of local tenure from the outset are less likely to be subject to extensive alterations that may compromise biodiversity objectives.

Here, I demonstrate how spatial zoning software can be used to incorporate local marine tenure into the design of MPA networks in the Philippines. The resulting MPA networks distribute the costs of conservation equitably amongst local fishing communities, producing solutions that are socioeconomically viable without compromising biodiversity objectives. My approach turns the traditional reserve selection problem on its head, treating the areal extent of fisheries as targets to be achieved, rather than costs to be minimised (see also Ban & Vincent 2009; Klein et al. 2009). I show, somewhat counter-intuitively, that larger MPA networks might have a greater likelihood of acceptance by local communities.

6.2. Methods

6.2.1. Study region

I used the island Province of Siquijor, Philippines, as our case study (Fig. 6.1). More than two thirds of the population (c. 88,000) live in coastal *barangays*, (analogous to a village or city ward) and small-scale fisheries (defined as those using vessels of three gross tons or less) contribute considerably to income and food security. Coral reefs in the Philippines are increasingly threatened by overexploitation, destructive fishing techniques and coastal development. Networks of comprehensive, ecologically representative, connected and resilient MPAs are considered necessary both to safeguard the livelihoods of coastal communities (Coral Triangle Initiative 2008) and to conserve the Philippines' rich marine biodiversity.

Prior to colonisation, the Philippines had a long history of traditional marine tenure at the *barangay* level. During the Spanish colonial period, the traditional property rights of *barangays* over their fishing grounds were steadily eroded, and superseded by national government control (Pomeroy & Carlos 1997). More recently, this trend has been reversed, with decentralisation of management of coastal resources (to 15 km offshore) to municipal level following the Philippine Local Government Code of 1991 (Republic Act 7160). While it has been stated that traditional fishing rights and *barangay*-based management systems have disappeared (Pomeroy & Carlos 1997), I found that in my study region the spatial distribution of small-scale fishing effort still conforms to boundaries between *barangays* (Weeks et al. 2010b, Chapter 5). Although MPAs are legislated at the municipal level, *barangay* governments play a central role in planning and implementation. Furthermore, fisherfolk associations, which often play a key role in MPA management, also operate at this scale.

6.2.2. Data

Biodiversity features targeted for inclusion in MPA networks were coral reef-associated habitats (fringing reefs, sunken shoals, seagrass, lagoons and mangroves) and bioregions (Fig. 6.1). Habitat types were identified from satellite imagery and verified using hand-held GPS as part of the Siquijor Coastal Resource Enhancement Project (2003). I subdivided these five habitat types into six bioregions identified on the basis of reef fish community composition (Fig. 6.1, see also Chapter 3). This resulted in 22 targeted biodiversity features (not all habitat types were present in all bioregions).

Data on the spatial distribution of fishing effort were collected through semi-structured interviews with fishers (see Chapter 5). Interviewees were asked to identify fishing grounds used by members of their *barangay*, and to estimate the number of fishers using each site. I found that fishers operating inshore only identified fishing grounds in areas adjacent to their own community, adhering to informal boundaries between *barangays* (fishers using hook and line in open water travelled beyond their *barangay*, but rarely identified fishing grounds outside their municipality). I thus identified 67 stakeholder groups of small-scale fishers around Siquijor: one for each coastal *barangay*. I mapped marine tenure units by applying guidelines for delineating municipal waters (DENR 2001) to boundaries between *barangays*. Thus, the tenure boundaries used are conceptual, and should not be considered definitive. I did not consider commercial fishers as stakeholders, as following the Local Government Code of 1991 and Philippine Fisheries Code of 1998 (Republic Act 8550) commercial fishers are excluded from operating within 15 km of the coastline.

6.2.3. MPA network design

I used the conservation planning software Marxan with Zones (Watts et al. 2009; available online at http://www.uq.edu.au/marxan/) to identify MPA networks that achieved specified biodiversity objectives whilst minimising impacts on small-scale fishers. In contrast to other conservation planning software (including earlier versions of Marxan), Marxan with Zones allows users to allocate sites to a range of different zones that offer different levels of protection (e.g. no-take, habitat protection, open access). This functionality allows users to address multiple objectives simultaneously (Watts et al. 2009). For example, conservation planners typically require the presence of biodiversity features (e.g. habitat types, species) within protected zones. Using Marxan with Zones, it is possible to target simultaneously socioeconomic activities (e.g. fishing, recreation) for inclusion in zones in which that activity is permissible. Klein et al (2009) demonstrate how this functionality can be used to ensure that impacts on different commercial fishery

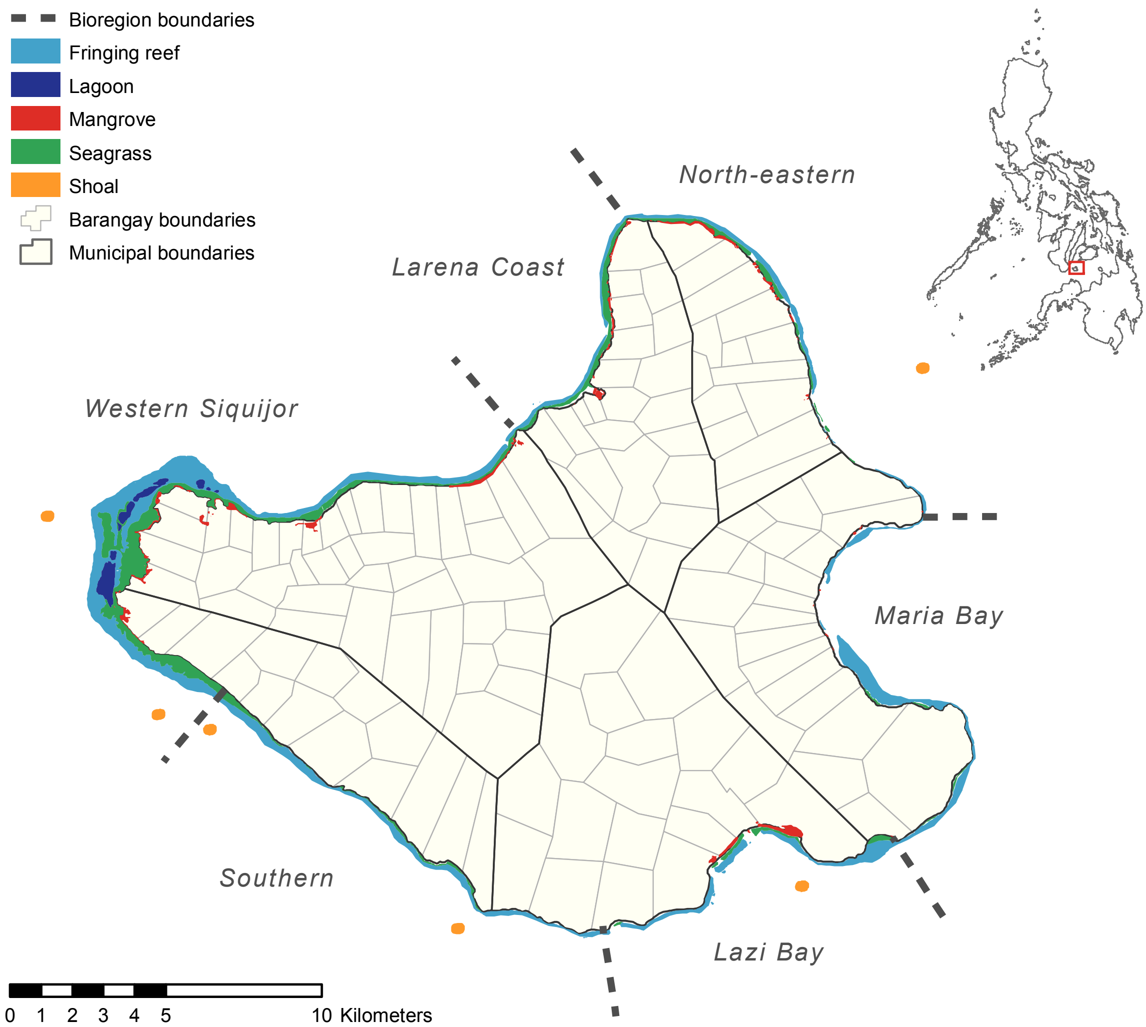

Figure 6.1. Benthic habitat types and bioregions targeted for inclusion in an MPA network for Siquijor Province, Philippines.

sectors are equitable. Here, I demonstrate the flexibility of this approach to incorporate the constraints of local marine tenure in the design of MPA networks.

I divided the planning region into regular hexagonal planning units of 0.05 km^2. This planning unit size was selected to be at a scale relevant to management: the median size of no-take MPAs in the Philippines is 0.12 km^2, and the minimum for Siquijor, 0.04 km^2 (Weeks et al. 2010a, Chapter 2). Thus, a single planning unit, or two contiguous units, would be a typical size range for a no-take MPA in the region. Each planning unit could be assigned to one of two zones: no-take or open to fishing.

To assess the effect of local marine tenure on MPA network design, I compared two different scenarios. In line with the Philippine Marine Sanctuary Strategy and the Coral Triangle Initiative (Arceo et al. 2004; Coral Triangle Initiative 2008), the biodiversity objective for both scenarios was to include 10 % of the area of each of five biodiversity features (reef-associated habitat types in each bioregion, Fig. 6.1) within MPAs. In scenario one, our objective was to represent 10 % of all biodiversity features within MPAs whilst minimising the cost of implementation to small-scale fishers as a single group. I assumed that minimising opportunity costs to fishers would increase the likelihood that they would support and comply with MPA implementation, resulting in more effective conservation. Thus, costs were assigned to each potential MPA site as the number of fishers that would be displaced if that site were protected, from interview data (Weeks et al. 2010b, Chapter 5). This scenario corresponds to a provincial-scale planning process, with no consideration of local tenure.

In scenario two, in addition to requiring representation of biodiversity features in no-take zones (as above), I required that a minimum percentage of the area of inshore fishing grounds in each *barangay* remain within the 'fished' zone. To examine trade-offs between achieving biodiversity and fishery targets, I incrementally increased the fishery targets until it was not possible to achieve all biodiversity and fishery targets simultaneously. I varied Marxan's 'feature penalty factor' to express a preference for meeting either biodiversity or fishery targets in order to determine where trade-offs occurred. For each scenario and target level I performed 100 replicate Marxan runs, and report results as the mean values of these replicates.

6.3. Results

In scenario one, which did not include fishery targets, the percentage of fishing grounds lost was highly variable among *barangays* (Fig 6.2). Whilst many communities retained all of their fishing area, others lost as much as 60 % (Fig. 6.3a). In contrast, the fishery targets

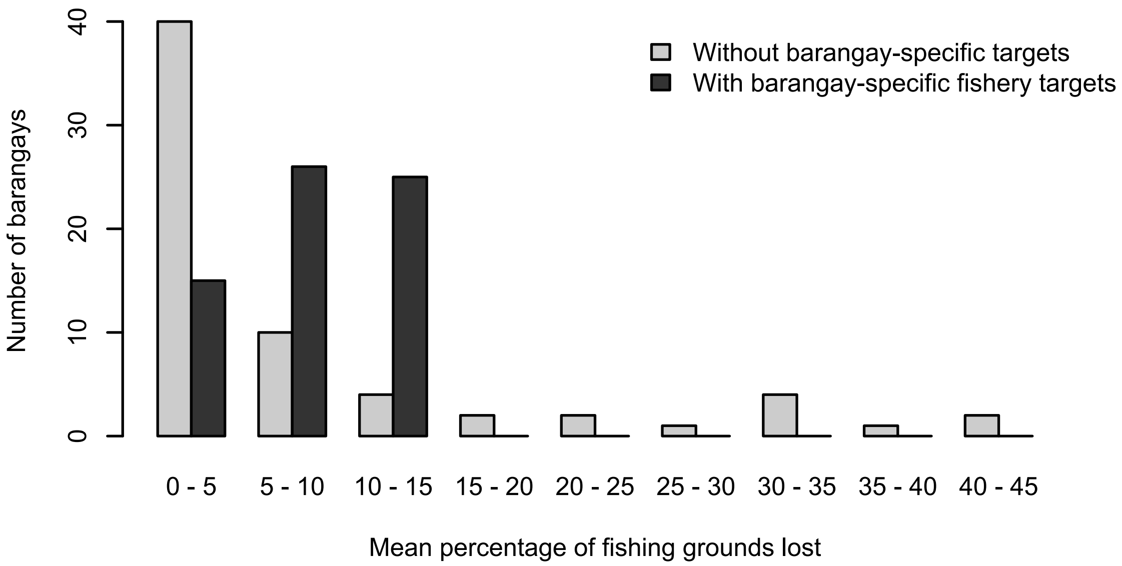

Figure 6.2. The mean percentage of fishing grounds lost (from 1000 Marxan solutions) by different *barangays* under two MPA network design scenarios: an MPA network designed to minimise overall cost to small-scale fishers without *barangay*-specific fishery targets; and an MPA network designed to minimise cost to small-scale fishers with the additional constraint that a minimum of 87 % of the fished area in each *barangay* remains open to fishing.

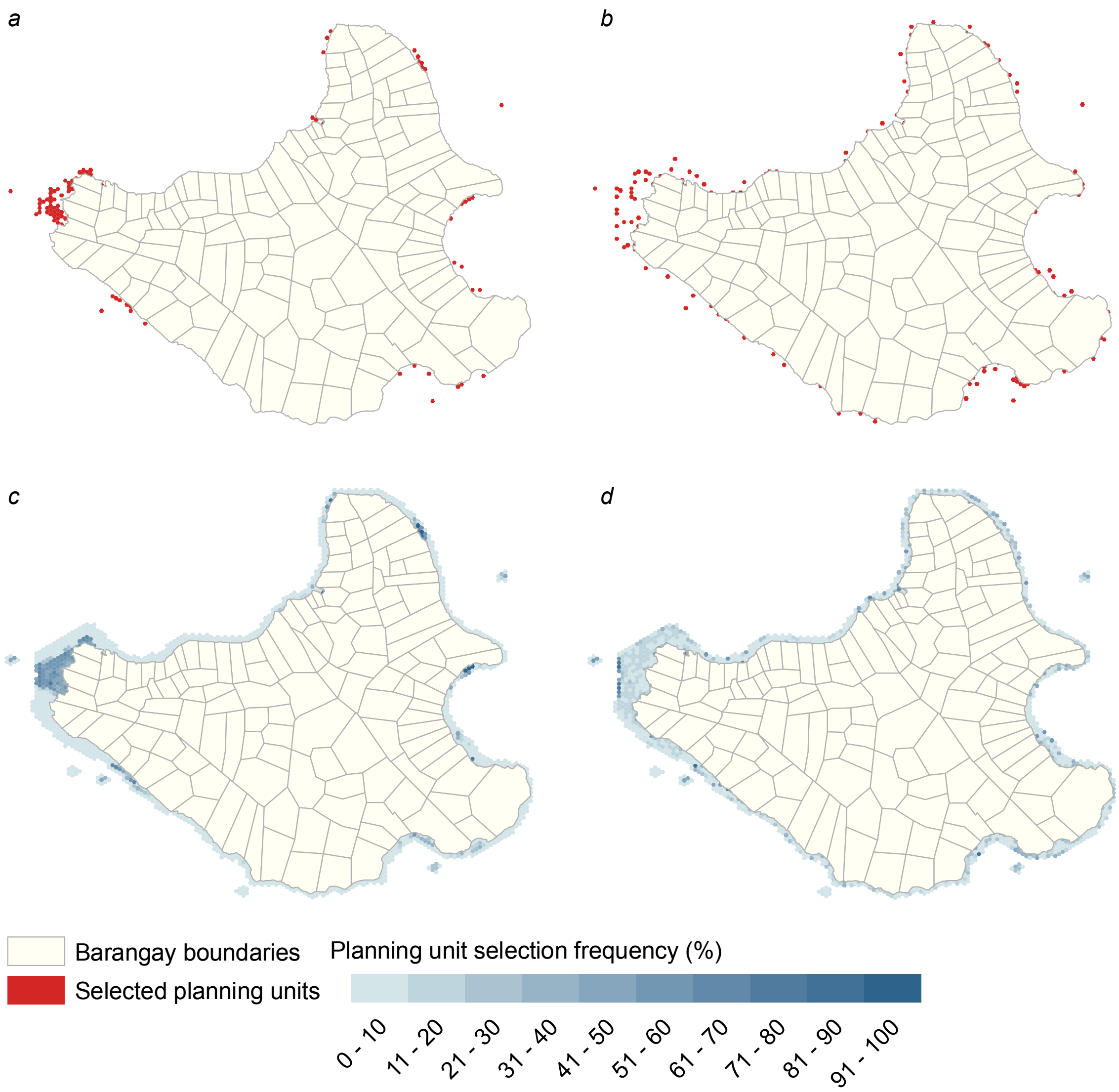

Figure 6.3. 'Best' Marxan results and planning unit selection frequencies under two MPA network design scenarios: (a and c) an MPA network designed to minimise overall cost to small-scale fishers, without *barangay*-specific targets; (b and d) an MPA network designed to minimise cost to small-scale fishers with the additional constraint that a minimum of 87 % of the fished area in each *barangay* remains open to fishing.

specified in scenario two ensured that the costs of MPA network implementation were distributed more equitably across local fishing communities (Fig. 6.3b).

Biodiversity and fishery targets could be achieved simultaneously with fishery targets up to 87 % (i.e. at least 87 % of the fishing grounds in each *barangay* remained open to fishing). When fishery targets were increased above this amount, not all biodiversity targets could be met (Fig. 6.4). Some biodiversity targets were achieved more easily than others: whilst the percentage area of seagrass, fringing reef and lagoons within MPA networks declined steadily as fishery targets increased above 87 %, representation targets for mangroves and shoals were still achievable with up to 95% of the fished area in each *barangay* open to fishing (Fig. 6.4).

MPA networks designed to achieve *barangay*-specific fishery targets (scenario 2) had greater total area (Fig. 6.5a) and cost (Fig. 6.5b) than those that sought to minimise costs to small-scale fishers as a single stakeholder group (scenario 1). MPA networks with fishery targets of 87 % were 40 % larger and almost twice as costly overall than those with no *barangay*-specific targets (Fig. 6.5).

To examine which *barangays* were causing trade-offs between biodiversity and fishery objectives, I changed Marxan's feature penalty factor to give preference to meeting biodiversity targets. When fishery targets were set at 90 %, on average 10 *barangays* (out of 67) did not achieve this target (i.e. less than 90 % of their inshore fishing grounds remained open to fishing). The identity of these *barangays* was relatively consistent, with 11 *barangays* accounting for two thirds of all missed targets. Of these, only two retained less than 87 % of their fishing grounds (mean values from 100 Marxan solutions).

6.4. Discussion

In regions where resource use patterns are defined by local tenure, it might be more important to minimise costs to each local community individually than to minimise the overall cost or area of an MPA network. For example, Ban et al (2009b) found that when presented with a choice of MPA networks, indigenous communities in Canada did not necessarily prefer the most efficient solution; other factors, such as the locations of individual MPAs, were more important. This is in contrast to previously stated concepts of 'efficiency' in conservation planning that emphasise minimising the overall area or cost of a protected area network, under the assumption that costs are borne by a single group (e.g. a non-governmental organisation purchasing land) (Stewart & Possingham 2005; Naidoo et al. 2006).

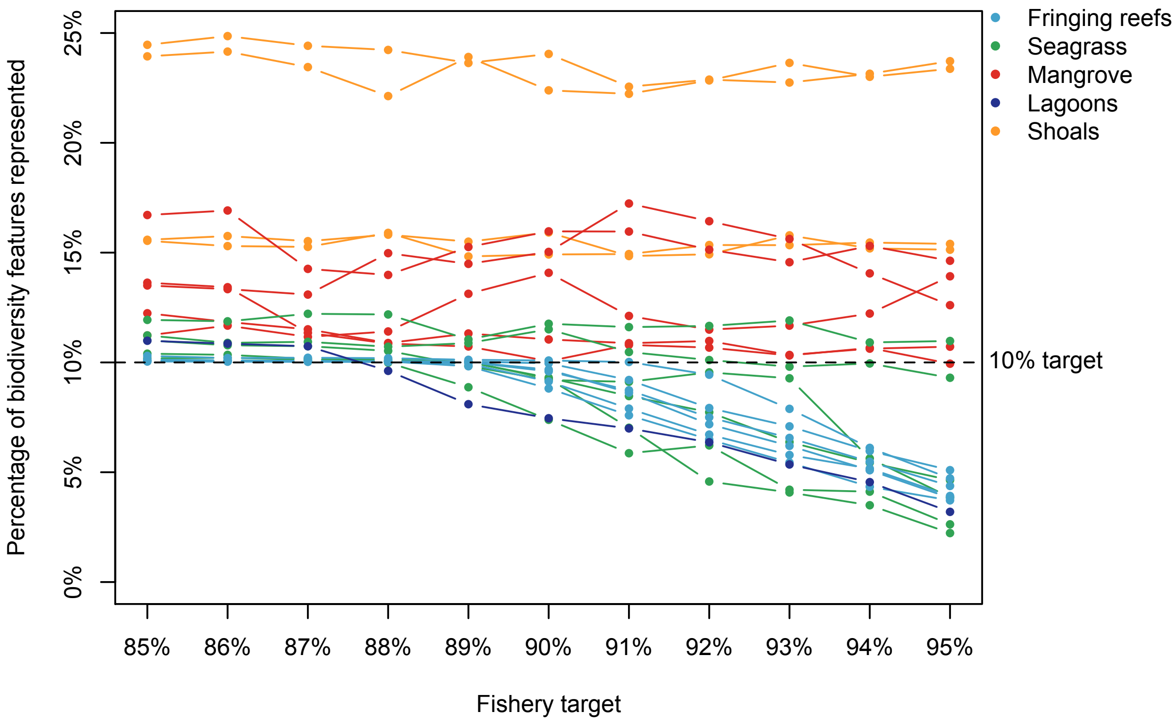

Figure 6.4. Trade-offs between achieving biodiversity and fishery targets. Fishery targets equate to the minimum percentage of inshore fishing grounds of each *barangay* that must remain open to fishing. Values shown are means for 100 Marxan solutions. Replicates for each habitat type show the percentage of habitat within MPAs for each bioregion in which that habitat occurs.

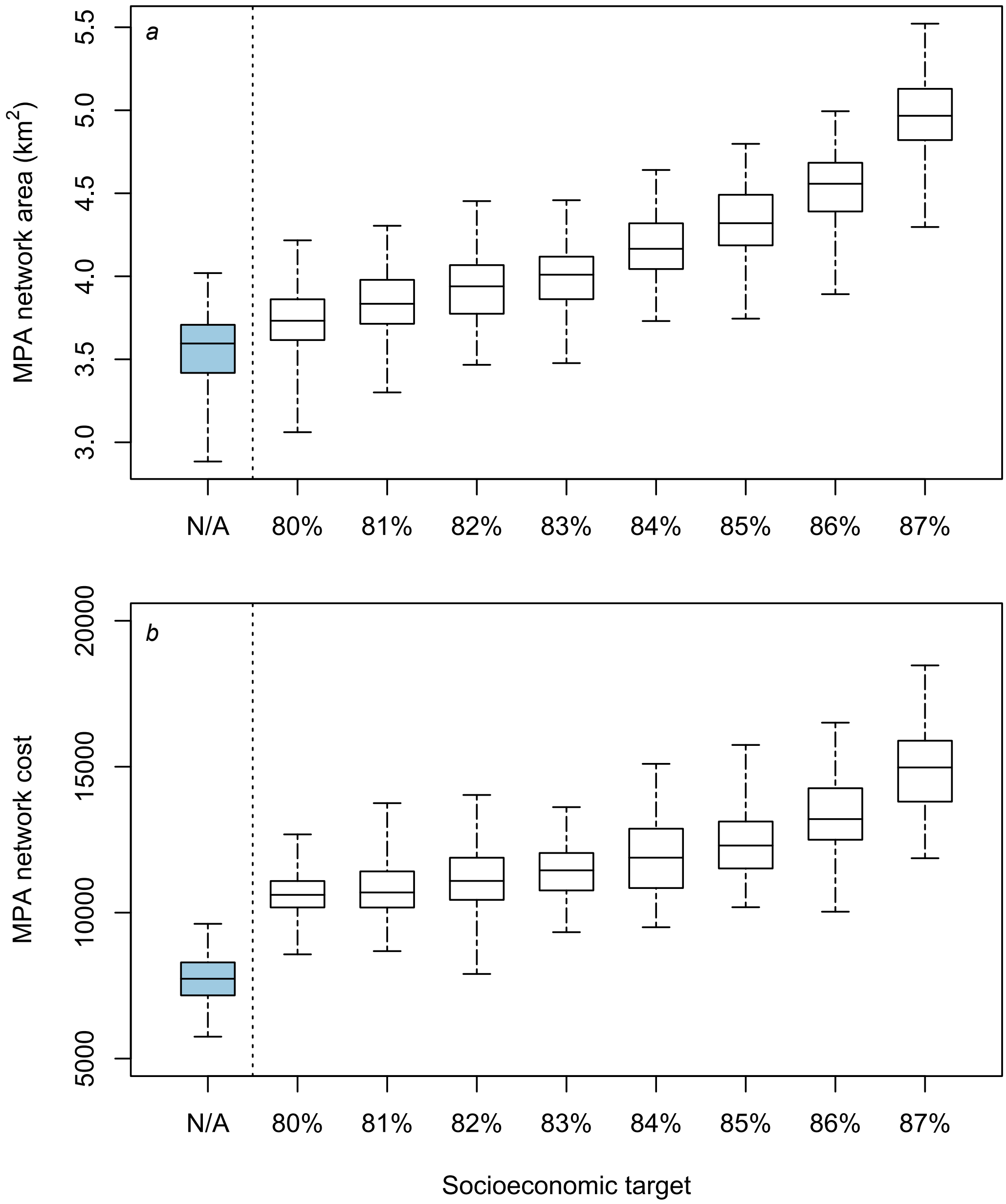

Figure 6.5. The total area (a) and cost (b) of MPA networks with different fishery targets. MPA network cost equates to the number of fishers that would be displaced by MPA implementation. Boxes show the mean and interquartile range of 100 replicate Marxan solutions, whiskers show the full range of the data. All scenarios that incorporate *barangay*-specific fishery targets (open boxes) have greater area and cost than those that aim to minimise costs to small-scale fishers as a single stakeholder group (i.e. do not include *barangay*-specific targets) (blue boxes).

I found that by specifying zone-specific fishery targets, I was able to distribute the cost of MPA network implementation more equitably across coastal communities. Klein et al (2009) achieved similar results for commercial fisheries. This approach has the additional benefit of presenting stakeholder interests as targets to be achieved, rather than obstacles to be overcome: a concept likely to be well-received in participatory planning.

My results indicate that to achieve biodiversity conservation objectives for Siquijor, each *barangay* would have to close up to a maximum of 13 % of their inshore fishing grounds. To put this figure into context, the no-take MPA at nearby Apo Island covers approximately 10 % of the coral reef area (Alcala & Russ 2006). This MPA is widely considered to be both an ecological and socioeconomic success (Russ et al. 2004; Alcala & Russ 2006). Another MPA at Sumilon Island covers 25 % of coral reef area; however, this island does not have a resident community (Alcala & Russ 2006). The inshore fishing grounds considered here represent a surprisingly small proportion of the total fishing grounds used by coastal communities: in many *barangays* the majority of fishers operate in open water or on offshore shoals, beyond the boundaries of our planning region. Thus, a 13 % reduction in inshore fishing grounds equates to a much smaller loss of fished area overall. Nevertheless, inshore habitats are used by gleaners and fishers with non-motorised boats; these resource users have the least spatial mobility, and are therefore most vulnerable to spatial closures (Johannes 2002; Fabinyi 2010).

MPA networks that incorporated local tenure boundaries through the use of zone-specific socioeconomic targets had greater total cost and area than those that did not. Mathematically, it is not surprising that a more constrained optimisation algorithm is less efficient (McDonald 2009). Several studies have demonstrated that planning across larger spatial extents results in protected area networks that are more cost and area efficient than if planning processes are conducted for subregions separately (Erasmus et al. 1999; Vazquez et al. 2008; Kark et al. 2009). Greater efficiencies have led some to advocate 'scaling-up' conservation planning efforts through multinational coordination (Kark et al. 2009). However, McDonald (2009) points to the increased transaction costs (time, money, politics) in coordinating conservation efforts across borders. Such transaction costs can be observed even at the village scale (Cinner & Aswani 2007), and can be especially acute if they require changing traditional patterns of resource use: for example, negotiating access for fishers from a different community to compensate for the loss of their traditional fishing grounds to establish an MPA.

MPA networks that incorporate local tenure at fine spatial scales are necessarily more fragmented than those that do not, as large, contiguous closures will only be viable if fishers have sufficient spatial mobility to retain access to fishing grounds (Aswani & Hamilton 2004b). However, MPA networks that comprise many small no-take areas may not be ecologically desirable: guidelines for MPA network design recommend 'bigger rather than smaller' no-take areas, to ensure that they are adequate to protect biodiversity features and ecological processes (McCook et al. 2009; McLeod et al. 2009). Thus, there may be a trade-off between the socioeconomic acceptability and ecological viability of an MPA network. Nevertheless, a network of small MPAs that are supported by local communities (and thus have good levels of compliance) is likely to be more effective than a system of large MPAs that exists on paper only.

It has also been suggested that the cost of implementing and managing many small reserves will be greater than that for fewer, larger closures (Roberts et al. 2003). This argument is most relevant to MPA systems in developed countries, with centralised management. In regions such as the Philippines, where local communities take responsibility for MPA management, the cost of managing many small reserves may not be significantly greater than that of a few large no-take areas, especially if the smaller closures are supported by local fishers, reducing the need for enforcement.

An alternative approach to establishing many small MPAs within tenure units would be to compensate communities that sacrificed a large proportion of their fishing grounds to establish an MPA. The results for our case study suggest that to achieve biodiversity objectives, relatively few communities would require compensation. However, attempts to develop sustainable alternative livelihoods are often unsuccessful (Johannes 2002), and might not reduce dependence on coral reef ecosystems (Cruz-Trinidad et al. 2009). In regions with highly exclusive marine tenure, it may be possible to designate larger no-take areas by placing them across boundaries, as fishers will only be impacted by the loss of fishing area due to an MPA that falls within their tenure unit.

Conservation planning software such as Marxan with Zones provides a tool for evaluating trade-offs between competing objectives, which is critical for informed decision-making. Setting zone-specific socioeconomic targets allows conservation planners to be explicit about trade-offs not only between biodiversity and socioeconomic objectives, but also between different stakeholder groups. For simplicity, in my analysis I applied the same fishery target for all *barangays*. An alternative approach would be to set variable targets, for example to reflect the number of fishers in each community, indices of occupational mobility or dependence on marine resources. This could be achieved either by specifying different percentage area targets for each fishing community, or by assigning variable penalty factors (which inform Marxan how important it is that a target is met).

By setting the minimum area of fishing grounds to be retained for each community, I was able to design MPA networks that impact resource users more equitably, and are therefore more likely to achieve support from local communities. This support is essential in regions where effective management is reliant upon voluntary compliance (e.g. Alcala & Russ 2006). Although it appears counter-intuitive that larger MPA networks may be more socially acceptable, in this context, minimising costs to each stakeholder group individually is likely to be more important than overall 'efficiency'.

Conservation planning has been criticised for creating 'grand designs' (Sayer et al. 2008) that fail to consider adequately the socioeconomic context in which they are to be applied (Knight et al. 2008; Polasky 2008). Whilst it is true that many early applications focused exclusively on biological aspects of conservation, more recent studies have demonstrated consideration for the socioeconomic viability of plans, for example, by incorporating opportunity costs to resource users (e.g. Cameron et al. 2008; Klein et al. 2008). We contribute to these efforts by demonstrating that spatial zoning software can be used to design MPA networks that achieve biodiversity conservation targets within the constraints of local marine tenure. The general approach that we describe here could be applied in any region in which spatial resource use is subject to local-scale constraints. It has particular relevance to the Coral Triangle Initiative, a key objective of which is to develop MPA networks throughout the Philippines, Indonesia, Malaysia, Papua New Guinea, Solomon Islands and Timor Leste (Coral Triangle Initiative 2008), all countries that are characterised by local marine tenure.

7. Proposed framework for systematic MPA network design in the Philippines

7.1. Introduction

Through the Coral Triangle Initiative (CTI), and complementary national legislation, the Philippines has committed to establishing a national system of comprehensive, ecologically representative, connected and resilient MPA networks (Coral Triangle Initiative 2008). There are two apparent approaches by which this goal might be achieved. First, continue the present 'community-based' approach to MPA establishment (e.g. Alcala 1998), directing funds towards supporting and extending the capacity of Philippine-based non-governmental organisations (NGOs) and local governments nationwide. Alternatively, take advantage of the injection of financial support and expertise offered by the CTI to undertake broad scale systematic marine conservation planning (Margules & Pressey 2000), as has been applied successfully on the Great Barrier Reef (Fernandes et al. 2005), and elsewhere.

Here, I contend that neither of these approaches will be effective to achieve the objectives of the CTI on their own. I propose a new framework for MPA network development in the Philippines that merges elements from regional-scale conservation planning with local-scale implementation. This approach is most likely to achieve a satisfactory trade-off between achieving objectives for biodiversity conservation and addressing the concerns of local fishing communities.

7.1.1. Why regional-scale conservation planning is inappropriate, and community-based efforts insufficient, for MPA network development in the Philippines

Systematic conservation planning (Margules & Pressey 2000) is widely considered to be the most effective approach to design MPA networks and multiple-use zoning plans, and increasingly underpins the conservation strategies of NGOs and governments worldwide (e.g. Groves et al. 2002; Pressey & Bottrill 2009). In 2004, a systematic approach was successfully applied to extend no-take protection of the Great Barrier Reef (Australia) to 33%, including representation of all habitats and bioregions, with acceptable levels of negative impact on commercial and recreational fishers (Day et al. 2002; Fernandes et al. 2005; McCook et al. 2010). Similar approaches have been applied to design MPA networks in California (Gleason et al. 2010), and elsewhere.

Numerous frameworks for conservation planning have been proposed (e.g. Margules & Pressey 2000; Groves et al. 2002; Cowling & Pressey 2003; Knight et al. 2006a; Pressey & Bottrill 2009; Banks & Skilleter 2010; Foley et al. 2010). Whilst many of these focus on the technical aspects of conservation assessment (Knight et al. 2006a), others outline the entire planning process, from inception to implementation. The framework proposed by Pressey and Bottril (2009) (summarised in Table 7.1) contains eleven stages: (1) scoping and costing the planning process; (2) identifying and involving stakeholders; (3) describing the context for conservation areas; (4) identifying conservation goals; (5) collecting data on socioeconomic variables and threats; (6) collecting data on biodiversity and other natural features; (7) setting conservation objectives; (8) reviewing current achievement of objectives; (9) selecting additional conservation areas; (10) applying conservation actions to conservation areas; and (11) maintaining and monitoring areas.

There are two key reasons why a regional-scale systematic conservation planning approach will be inappropriate for MPA network development in the Philippines. First, in the Philippines, as is the case in many tropical developing countries, the resolution and extent of available biodiversity data are likely to be insufficient to inform regional-scale MPA network design adequately (Ban et al. 2009a). To avoid bias in site prioritisation, data must be consistent in extent and resolution across the planning region (Pressey 2004; Grand et al. 2007); yet fine-scale data are typically only available for small areas (Good et al. In review), requiring the use of broad-scale surrogates for regional-scale planning (Rouget 2003). Nevertheless, the spatial resolution of these data must be at least that of the size of MPAs to be established; otherwise there is a risk that, without extensive groundtruthing, MPAs will be placed in areas that do not in fact contain the features they were designed to protect. Thus, the need for accurate, fine-resolution data to guide MPA placement is especially acute in regions where the size of no-take areas is heavily constrained by socioeconomic factors, as is the case in the Philippines, where the median size of community-based MPAs is just 0.12 km^2 (Weeks et al. 2010a, Chapter 2). Acquiring such data across sufficient extents to undertake regional planning would require a significant amount of time and resources, which would be better invested in applying conservation actions (Grantham et al. 2008)

Second, and more importantly, previous attempts to establish MPAs in the Philippines through top-down planning approaches have not been successful (Christie & White 1997; Alcala 1998; Russ & Alcala 1999; Alcala & Russ 2006). Projects undertaken with little or no stakeholder involvement have disenfranchised local fishing communities from the process of managing marine resources (Alcala & Russ 2006), and centralised management systems mired in bureaucracy have failed to enforce regulations (Alcala 1998). Although some conservation planning frameworks (including that proposed by Pressey and Bottril, Table 7.1) emphasise the importance of stakeholder involvement, it is not logistically

Table 7.1. Framework for systematic conservation planning; adapted from Pressey and Bottril (2009)

Stage		Description
1	Scoping and costing the planning process	Decisions are necessary on the boundaries of the planning region, the composition and required skills of the planning team, the available budget, necessary funds in addition to those available and how each step in the process will be addressed, if at all.
2	Identifying and involving stakeholders	Important stakeholders include those who will influence or be affected by conservation actions arising from the planning process, or be responsible for implementing those actions. Different groups of stakeholders will need to be involved in different ways in specific stages of planning.
3	Describing the context for conservation areas	The planning team describes the socioeconomic and political setting for conservation planning, identifying the types of threats to natural features that can be mitigated by spatial planning and the broad constraints on, and opportunities for, conservation actions
4	Identifying conservation goals	May begin with agreement on a broad vision statement for the region that is then progressively refined into qualitative goals about biodiversity (e.g. representation, persistence), ecosystem services, livelihoods and other concerns. Goals help to identify the need for spatial data.
5	Collecting data on socioeconomic variables and threats	Relevant spatially explicit data will include variables such as tenure, extractive uses, costs of conservation, and constraints and opportunities to which planners can respond. Will also involve predictions about the expansion of threatening processes.
6	Collecting data on biodiversity and other natural features	The planning team will collect spatially explicit data on biodiversity that include representation units (e.g. vegetation types), focal species and ecological processes. This may extend to ecosystem services (e.g. maintenance of water flows, carbon sequestration).
7	Setting conservation objectives	Involves interpreting goals to define quantitative conservation objectives for each spatial feature (e.g. 2,000 ha of each vegetation type, 16,500 individuals of each species) and, where necessary, qualitative objectives related to configuration, past disturbance and other criteria.
8	Reviewing current achievement of objectives	Remote data, and perhaps also field surveys, are used in this stage to estimate the extent to which objectives have already been achieved in areas considered to be adequately managed for conservation
9	Selecting additional conservation areas	With stakeholders, this stage requires decisions about the location and configuration of conservation areas that complement the existing ones in achieving objectives. Factors influencing decisions will include costs, constraints on, and opportunities for, effective conservation.
10	Applying conservation actions to conservation areas	Application of conservation actions requires a variety of technical analyses and institutional arrangements to ensure that selected areas are given the most feasible and appropriate conservation management and that areas are prioritised for action when resources are limited
11	Maintaining and monitoring areas	Activities ensure that individual areas are managed to promote the long-term persistence of the values for which they were established. This involves explicit management objectives and monitoring to ensure that management actions are effective.

feasible to engage all coastal communities meaningfully in regional-scale planning (Green et al. 2009). For this reason, Christie et al (2009) suggest that in the Philippine context, large-scale MPA networks or centrally planned zoning schemes based primarily on national law and international targets, will almost certainly fail.

In contrast to the failure of top-down management approaches, in the Philippines, community-based (or 'bottom-up') approaches to no-take MPA implementation are well established and empirically proven. Following the successful establishment of no-take MPAs at Sumilon and Apo Islands (in 1974 and 1982 respectively, Alcala & Russ 2006), local NGOs and academic institutions have facilitated similar projects throughout the country, resulting in over 900 community-based MPAs nationwide (Weeks et al. 2010a, Chapter 2). The success of these initiatives is widely attributed to the involvement of local fishing communities in all stages of planning and management, resulting in strong stakeholder support for MPAs and good levels of compliance (Alcala 1988; Pomeroy et al. 1997; Alcala 1998; White & Vogt 2000; Pollnac et al. 2001; Christie et al. 2002; Walmsley & White 2003; Crawford 2004; Alcala & Russ 2006; Samoilys et al. 2007).

Although the planning process may vary, depending on the unique context of the community in which it is applied, community-based coastal resource management projects in the Philippines typically involve the following stages (Alcala 1998; White et al. 2002): (1) project initiation and planning; (2) community organisation; (3) environmental education and capacity building; (4) resource management planning and MPA establishment; (5) development of alternative livelihood activities; (6) research and monitoring; and (7) networking activities (Table 7.2).

Whilst it has been demonstrated that community-based MPAs in the Philippines can achieve a range of local-scale objectives for fisheries management, conservation and socioeconomic development (Chapter 4), at present it is clear that they do not constitute a comprehensive national MPA network (Weeks et al. 2010a, Chapter 2). Furthermore, even in regions where they have been applied extensively, such approaches have not resulted in ecologically functional MPA networks (Chapter 4). The majority of community-based initiatives lack the spatial context required to achieve regional-scale objectives for biodiversity conservation. Although efforts to develop social networks around existing MPAs have been initiated in some regions (Alino et al. 2006; Christie et al. 2009; Eisma-Osorio et al. 2009; Lowry et al. 2009), these efforts will not result in ecologically functional networks unless new MPAs are designated to address biases in representation, which

Table 7.2. Framework for community-based MPA establishment in the Philippines, adapted from Alcala 1998 and White et al 2002.

Stage		Description
1	Project Initiation	This stage involves conceptualising the project, selecting a focal community to work with, arranging the administrative setup and hiring staff required to undertake the project.
2	Social preparation and community organising	Community organisation is essential to the success of community-based coastal resource management, and may take a year or more. This stage involves formation of viable people's organisations that will plan and implement coastal resource management projects and collection of baseline socioeconomic and environmental data to inform planning.
3	Environmental education and capacity building	The community needs to be convinced of the need to protect and manage their own resources. Ecological relationships, e.g. the role of healthy environments in sustainable marine productivity, are demonstrated, and the economic values of coral reef and mangrove ecosystems are explained.
4	Resource management planning, including MPA establishment	Formation of a marine management committee comprising members of the community that will lead the MPA implementation process and be responsible for day to day MPA management. Decisions about the size and location of a no-take area are made with scientific advice from partner organisations and through extensive consultation with the community.
5	Development of alternative livelihood activities	After the reserve is established and some form of management is already developed, support is provided to develop alternative livelihood strategies, e.g. mat weaving, t-shirt selling, establishment of MPA entrance fees for tourists. Activities also refine management schemes and broaden the conservation strategy of the community.
6	Research and monitoring	Monitoring undertaken by researchers or community members trained in basic data collection and analysis provides information and feedback on management.
7	Networking activities	Links may be established with other MPA management committees in the region. Site visits by members of people's organisations in other communities are encouraged, to demonstrate the effects of no-take MPA implementation on fish stocks and livelihoods and accelerate the MPA implementation process in those communities.

exist as a result of previous opportunistic approaches to MPA establishment (Agardy 2005, Chapter 4).

It is clear then that neither regional-scale systematic conservation planning nor community-based MPA establishment can provide an effective approach to develop MPA networks in the Philippines on their own. Whilst community-based approaches have been successful in establishing MPAs to achieve local-scale objectives, they lack the scientific basis and spatial context required to develop ecologically functional MPA networks. Systematic conservation planning approaches provide this perspective, but are likely to be hindered by a lack of suitable data, and may ultimately fail if they are not accepted by local stakeholders. It is apparent that to successfully establish MPA networks in the Philippines, a planning framework that incorporates elements of both regional-scale conservation planning and community-based implementation will be required (Agardy 2005; Alino et al. 2006).

7.2. A new framework for MPA network design and implementation in the Philippines

Here, I propose a hierarchical framework for MPA network development in the Philippines that merges elements of a systematic conservation planning approach with the proven community-based approach to implementation. The framework, illustrated in Figure 7.1, comprises three phases: regional-scale planning and prioritisation, MPA network design, and local-scale implementation. The anticipated end result of this process is a regional-scale MPA system that is made up of several local-scale MPA networks, each comprising individual MPAs that are closely linked both ecologically and socially to other MPAs in their network, and are more loosely connected in both contexts to others in the region. As the planning process progresses from regional-scale prioritisation to implementation within individual communities, emphasis shifts from achieving biodiversity conservation objectives to realising local community objectives (which are more likely to prioritise socioeconomic factors). However, regional-scale priorities should be retained, ensuring that biodiversity targets are met at both regional and local scales.

The stages involved in each of the three phases are discussed below. Suggestions for how to define the spatial scale at which each phase is undertaken, and the roles that different organisations may play in the planning process follow. Many of the stages outlined here have been adopted from previously published frameworks for systematic conservation planning or community-based management. More detailed information on these stages is available elsewhere (see Tables 7.1 and 7.2, and associated references). I focus here on the

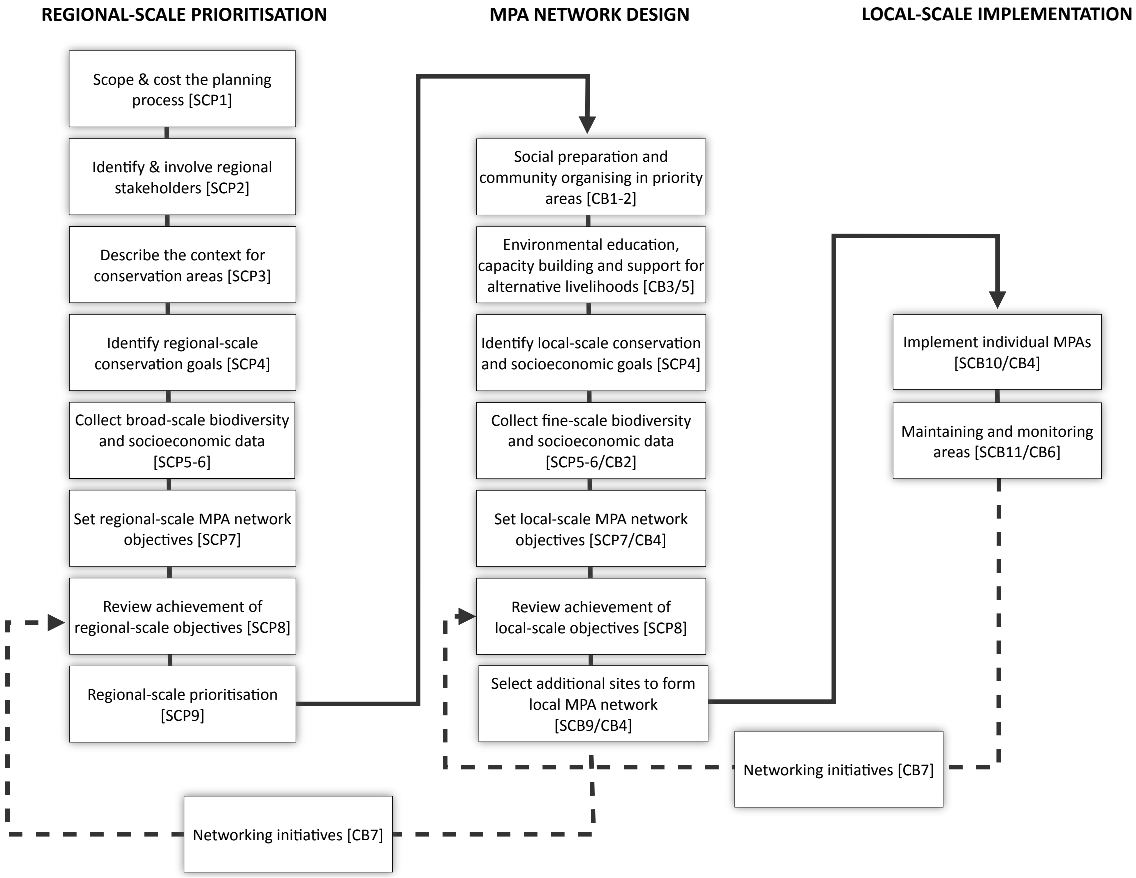

Figure 7.1. A proposed framework for the design and implementation of MPA networks in the Philippines. Although the process is depicted as a linear sequence, some stages may be undertaken simultaneously and there will be many feedbacks (dashed lines) from later to earlier stages. For example: once the process of designing a local MPA network has been completed, MPAs may be implemented in several communities concurrently; after each phase has been completed, the degree to which objectives have been achieved will be fed back to earlier phases so that targets for other regions can be adjusted as necessary. Parentheses indicate associated stages in frameworks for systematic conservation planning (SCP) and community-based MPA establishment (CB) outlined in Tables 7.1 and 7.2, e.g. 'SCP4' is the 4th stage in the systematic conservation planning framework.

key stages at which the two frameworks can effectively inform one another and thus act synergistically.

Following the Philippine Local Government Code of 1991 (Republic Act 7160) and the Philippine Fisheries Code of 1998 (Republic Act 8550), 'municipal waters' within 15 km of the coastline are under the jurisdiction of Municipal governments and are reserved for small-scale fishers. This framework is intended to be applied for developing MPA networks within this area. Other strategies will be employed to manage commercial fisheries, which may only operate beyond 15 km offshore. The recommendations and guiding principles for MPA network development presented here refer primarily to the implementation of no-take MPAs. Nevertheless, these will undoubtedly be more effective within a wider coastal resource management framework. Whilst discussed within the specific context of the Philippines, this framework could easily be applied with minor adjustments in other countries with a similar socioeconomic and ecological environment.

7.2.1. Regional-scale planning and prioritisation

The stages undertaken during this phase closely follow a typical systematic conservation planning process, with the key difference being that the ultimate objective is to identify regional priority areas within which to undertake local-scale MPA network design, rather than to select sites within which to implement conservation actions (e.g. establishing an MPA).

There are two benefits to undertaking regional-scale planning prior to designing MPA networks at local scales. First, regional priority areas can be identified to schedule community engagement. Given the small size of no-take MPAs that can be realistically established in the Philippines, it is likely that to achieve regional and national targets for conservation, MPAs will need to be established in the majority of coastal communities (Weeks et al. 2010a, Chapter 2). However, limited resources available for conservation initiatives dictate that implementation will occur incrementally (Pressey et al. 2007; Stewart et al. 2007). Regional prioritisations based on broad-scale biodiversity and socioeconomic data can assist facilitating organisations to schedule community engagement in areas that complement their existing project portfolio.

Second, regional-scale analyses provide a broader spatial context for conservation efforts within priority areas. NGOs can use this information to direct environmental education programmes for local communities. Although community representatives will ultimately determine the size and location of MPAs, they may be influenced to provide better protection for biodiversity features with regional importance.

Where possible, regional-scale prioritisations should be undertaken using existing or easily obtainable data. Investigations into data sufficiency for conservation planning have demonstrated diminishing returns on investment (Grantham et al. 2008), and recommend investing in implementation rather than additional data collection (Ban 2009). Although inadequate to inform MPA network design (i.e. the selection of sites for MPA designation), broad-scale data on the spatial distribution of biodiversity features and resource use can be used to identify regional priority areas. Habitat maps derived from high-resolution satellite imagery may be informative at this scale (Wabnitz et al. 2010). However, although remote sensing data is generally considered to be cost-effective (Dalleau et al. 2010; Wabnitz et al. 2010), the cost of acquiring high-resolution imagery and expertise required to interpret such images may be prohibitive for local NGOs in the Philippines. The Millennium Coral Reef Mapping Project (MCRMP, Andrefouet et al. 2006) aims to produce globally consistent geomorphological maps of coral reefs worldwide. MCRMP products, together with other biophysical and biodiversity data, were used in the development of an MPA network plan for Kimbe Bay, Papua New Guinea (Andrefouet et al. 2006; Green et al. 2009). Unfortunately, as of June 2010, MCRMP outputs have not been made available for the Philippines (http://www.imars.usf.edu/MC/products.html). Ideally, habitat maps should be supplemented by biogeographic classifications (Chapter 3) based on existing survey data or expert knowledge. Nautical charts may provide an additional source of biophysical data in data-poor regions (Ban 2009).

Although the primary focus of this phase is to achieve objectives for biodiversity conservation, socioeconomic factors should not be disregarded completely during regional scale planning, as their inclusion may help to highlight priority areas (Ban et al. 2009a; Weeks et al. 2010b, Chapter 5). Population data are unlikely to be a suitable socioeconomic surrogate at this spatial scale (Weeks et al. 2010b, Chapter 5); however, the relative density of small-scale fishers, collected at municipal level (e.g. Green et al. 2004) could provide appropriate socioeconomic data for regional-scale planning. If data on other threats, such as coastal development or major shipping routes are available, these could also be incorporated during this phase.

To allow assessment of the progress made by existing MPAs, and to provide a means for identifying priority areas, regional conservation goals must be translated into quantitative, operational targets. Whilst the ultimate goals of MPA networks are widely agreed upon (i.e. comprehensive, representative, adequate, replicated, connected, and resilient, Chapter 4), specific operational principles, and in particular percentage area targets, are more contentious (Agardy et al. 2003; Rodrigues et al. 2004; Svancara et al. 2005; Carwardine et al. 2009). Many commonly stated operational principles for MPA network design are not achievable, or appropriate in the social, economic and ecological context of the Philippines (Chapter 4). For example, recommendations typically state that no-take areas should span a minimum of 5 – 10 km of the coastline (Fernandes et al. 2005; CDFG 2008). If this principle were followed in the Philippines, no-take areas would encompass the entire fishing grounds of numerous *barangays*, impairing the livelihoods of hundreds of fishers. If they are to be useful, operational principles must evidently be tailored to the region in which they are to be applied (Chapter 4).

Proposed ecological principles for MPA network design in the Philippines are presented in Table 7.3. Many differ significantly from previously published guidelines (e.g. Fernandes et al. 2005; CDFG 2008; Green et al. 2009; McCook et al. 2009; McLeod et al. 2009). Aspects of MPA network design frequently constitute trade-offs between biodiversity and socioeconomic objectives (Halpern & Warner 2003; Mills et al. 2010); for example, the optimal size of no-take areas to achieve ecological objectives may not be consistent with socioeconomic constraints, which argue for smaller MPAs (Aswani & Hamilton 2004b). Although ideally, quantitative targets for conservation should be dictated by the requirements of the biodiversity features to be conserved (Pressey et al. 2003; Harborne 2009; Rondinini & Chiozza 2010), in reality they must strike a balance between aspirational and achievable. Thus, the design principles presented in Table 7.3 are not purely ecological, in that they incorporate considerations of feasibility.

It may be argued that, where conservation targets fall short of scientific recommendations, there is a risk that they will be insufficient to achieve the ultimate goal of conservation: ensuring the persistence of biodiversity and underlying ecological processes. However, whilst it is dangerous to create the impression that very small no-take areas will be sufficient, clearly unattainable targets may be counterproductive (Agardy et al. 2003; Carwardine et al. 2009). Thus, rather than implying that anything less than no-take protection for 30% of all habitat types will constitute failure, it may be more productive to set lower targets for most habitat types, whilst emphasising the need for greater protection in areas that have both ecological importance and in which it may be easier to establish MPAs, such as small island reefs (White & Vogt 2000; Beger et al. 2005). Recognising inevitable socioeconomic constraints, such as the maximum size of no-take areas, from the outset, allows planners to compensate in other areas, for example by increasing targets for feature replication within the MPA network (Table 7.3).

Table 7.3. Proposed ecological principles for MPA network design in the Philippines. Design principles that differ considerably from previous recommendations are italicised.

Principle	Comments
Include at least 10% of the area of each targeted habitat type in no-take MPAs; aim to include 20-30% where possible.	*Although recommendations from the scientific literature typically recommend that between 20-30 % of habitat should be protected to ensure the persistence of species and ecosystem functions (e.g. Fernandes et al. 2005), this will be difficult to achieve in the Philippines. The target set by the Philippine Marine Sanctuary Strategy (Arceo et al. 2004) and adopted in the Philippines National Plan of Action for the CTI is 10% of coral reef associated habitats.*
Include at least 10% of the area containing targeted habitat in each bioregion in no-take MPAs.	This principle should be applied at the regional scale, but may not be relevant to local scale MPA network design if the focal area occurs within a single bioregion.
Within bioregions, ensure protection for areas that are representative of the range of current, river exposure and slope gradients present in the planning region.	This principle should be applied for both regional and local scale planning.
Where information is available, protect ecologically important sites such as spawning aggregations, turtle nesting areas and nursery grounds within no-take MPAs.	*Migration corridors and sites used by cetaceans and elasmobranchs (often targeted for inclusion in MPA networks (e.g. Green et al. 2009)) are unlikely to be afforded protection by MPAs in the Philippines due to the small size of the MPAs (Chapter 2). These features and species should be protected by alternative legislation.*
Establish the largest no-take areas that can be effectively managed with good levels of compliance; aim for minimum area of 0.3 km^2, with an absolute minimum of 0.1 km^2. Where possible, seek to enlarge existing no-take MPAs.	*Recommendations that MPAs span coastlines of 5-20 km are not socioeconomically viable in the Philippines. Principles for the minimum area of no-take MPAs are based on the current median size of community-based MPAs in the Philippines (0.12 km^2; no new MPAs should be established below this size) and the size required to achieve national targets for coral reef habitat representation (0.3 km^2; Weeks et al 2010a, chapter 2).*
Aim to protect the entire reef area of offshore reefs and uninhabited islands.	*Guidelines commonly recommend that where a reef is incorporated into no-take areas, the whole reef should be included (e.g. McCook et al. 2009). However, given that much of the Philippine coastal environment comprises continuous fringing coral reefs, and coastal population density is consistently high, this will only be feasible in offshore, uninhabited areas (Chapter 4).*
Protect at least ten examples of coral reef habitat in each bioregion; include five examples of other habitat types.	*This represents the upper level of replication recommended in previously published guidelines for MPA network design (McLeod et al. 2009), in recognition of the typically small size of no-take MPAs in the Philippines. A higher target is specified for coral reef habitats, due to their relatively high biodiversity and importance for fisheries.*
Provide for a wide range of dispersal distances between no-take MPAs, with a maximum distance of 15 km between no-take areas.	The principle for maximum distance between MPAs can be assured by ensuring that regional priority areas are no more than 15 km apart (e.g. McCook et al. 2009). MPAs within local-scale networks will provide for shorter dispersal distances.

Table 7.3. (continued)

Principle	**Comments**
No-take MPAs should extend from the coastline to the bottom of the reef slope (with a small buffer zone to allow beach access where necessary).	*This strategy may offer protection for local-scale connectivity by protecting adjacent areas of coral reef, seagrass, and mangrove habitat (Mumby & Hastings 2008); it is preferable to targeting areas with high habitat diversity explicitly, which, given the small size of no-take areas, may result in the selection of marginal and fragmented habitat.*
Include sites that are likely to be resistant or resilient to coral bleaching.	E.g. areas in proximity to deeper, cooler water; those shaded by steep-sided islands or suspended sediments in the water column; and reef flats where corals are adapted to stress (West & Salm 2003). Where data are available, this principle can be applied at both regional and local scales.

Prior to identifying regional priority areas for MPA network design, the extent to which objectives are achieved in existing MPAs should be assessed using a gap analysis approach (Weeks et al. 2010a, Chapter 2). In addition to quantitative targets for biodiversity representation, such analyses require information on the location, sizes and features protected within existing MPAs to be collated or collected. Some progress has been made in this respect, including the development of a national MPA database (Arceo et al. 2008; Weeks et al. 2010a, Chapter 2), an MPA management effectiveness rating system (White et al. 2006b), and a comprehensive directory of marine reserves in the Central Visayas region (Alcala et al. 2008). Further efforts may be required to document MPAs in regions of the Philippines that have to date received less NGO support.

Identification of regional priority areas within which to pursue MPA network design and implementation may be facilitated by the use of conservation planning software, such as Marxan (Ball & Possingham 2000; Watts et al. 2009). These software packages assist conservation planners to identify potential protected area networks that achieve quantitative targets for biodiversity representation for a minimal cost, where 'cost' may relate to the monetary cost of implementation and management, foregone opportunity costs to resource users, or a combination of the two (Naidoo et al. 2006; Ban & Klein 2009; Weeks et al. 2010b, Chapter 5).

The region is first divided into 'planning units', which can be assigned to different use zones, most simply either 'fished' or 'MPA'. Given information about the amount of each biodiversity feature occurring within each planning unit, and the relative cost of designating each unit as an MPA, Marxan selects complementary sets of sites that achieve conservation targets for a minimal cost. Repeated runs of the software produce slightly different solutions, providing flexibility. Planners can also calculate the 'selection frequency' of planning units as the number of times that each unit appears in (for example) 1000 runs of the software. Selection frequency scores indicate the relative importance of different sites to achieve the stated conservation objective, assisting planners to identify and schedule priority areas (Figure 7.2). By analysing selection frequency scores for each targeted conservation feature separately, planners can identify why priority areas are important, providing a broader spatial context for conservation efforts in those areas. For example, Figure 7.3 demonstrates that whilst the Western Siquijor region is an overall priority area, it is especially important to achieve representation targets for seagrass habitat.

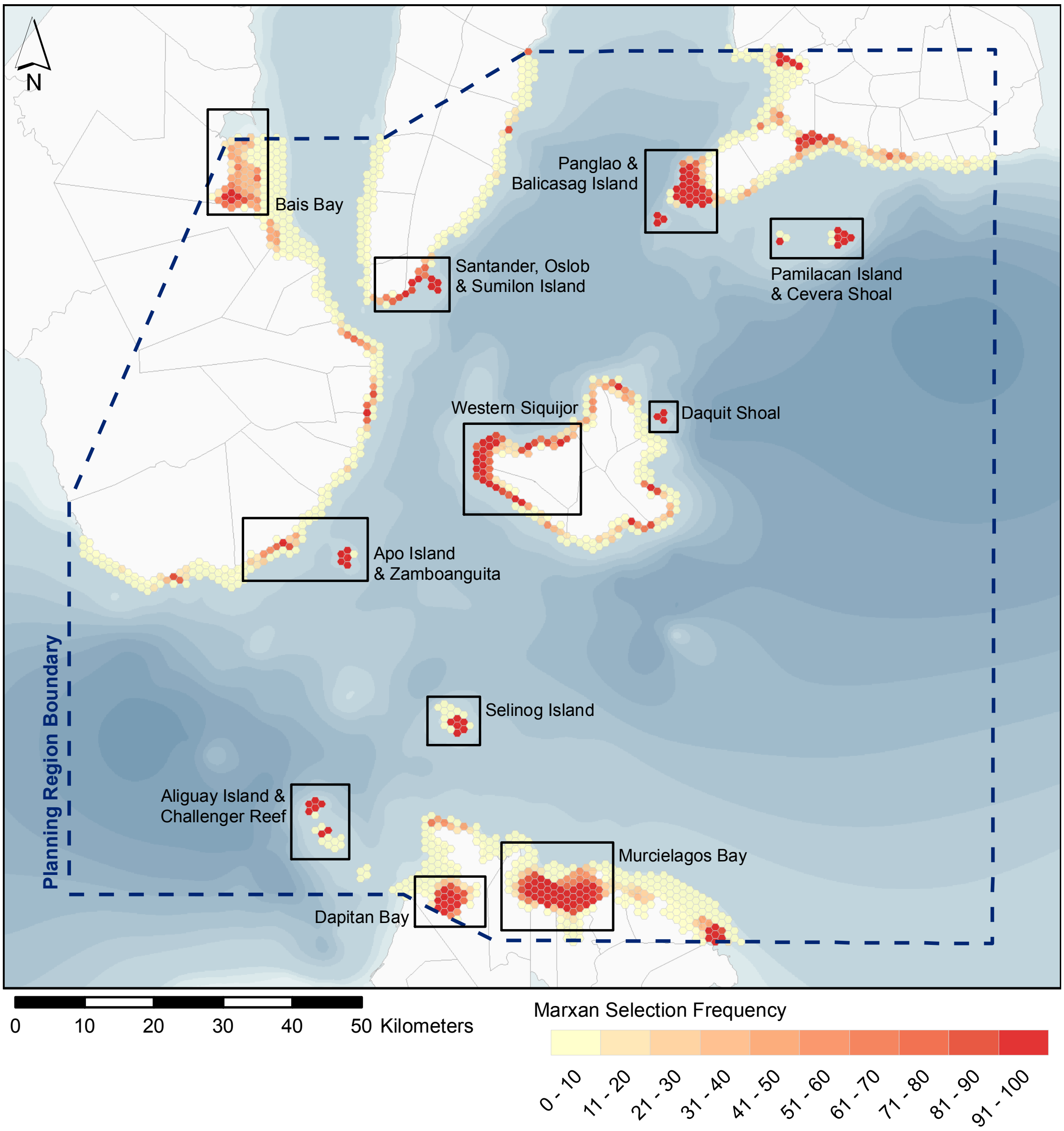

Figure 7.2. Regional priority areas identified for the Western Bohol Sea using the conservation planning software Marxan. Highlighted regions would be prioritised for community engagement and MPA network development (Fig. 7.1). Selection frequency scores indicate the percentage of times that each planning unit was selected in 1000 repeat runs of the software; planning units with higher selection frequency scores have greater conservation importance relative to the stated conservation objective. In this example, conservation targets correspond to those outlined in Table 7.3: e.g. 10% of the extent of fringing reef, seagrass, lagoon and mangrove habitats; 20% of the extent of island fringing reefs and shoals; a minimum of 5 occurrences of reefs classified into low, medium and high current, slope and river influence. Cost values were assigned to planning units as the relative density of small-scale fishers, calculated as the number of fishers per kilometre of coastline, modelled to 10 km offshore using linear distance decay (see Chapter 5).

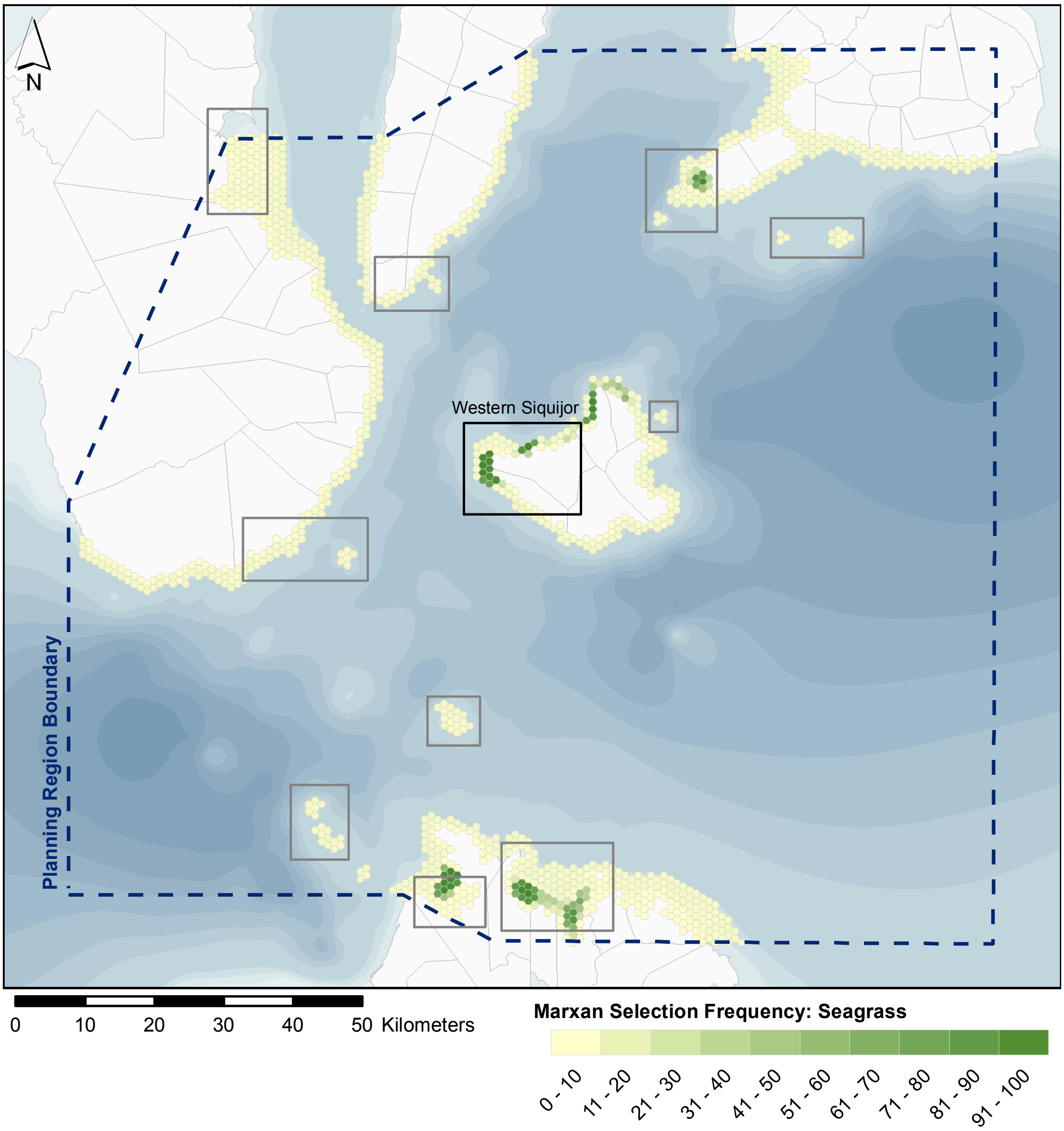

Figure 7.3. Areas of importance for seagrass conservation in the Western Bohol Sea. The conservation target was 20% of the extent of seagrass habitats in each bioregion. Performing regional-scale prioritisation analyses on each targeted conservation feature separately provides a broader spatial context for conservation efforts in priority areas. The regional priority areas highlighted in boxes here are the same as those shown in Figure 7.2; NGOs working with communities in the Western Siquijor region may choose to tailor community education programs to emphasise the ecological importance of seagrass habitats and set higher representation targets for these features in local MPA network design.

Marxan is programmed to identify sets of planning units 'efficiently', that is, for the minimum total cost. However, this approach can often result in highly fragmented MPA systems, which may make it difficult to distinguish regional priorities. Marxan's 'boundary length modifier' should therefore be used to improve the clustering and compactness of solutions (Ardron et al. 2008), allowing priority areas to be identified more clearly.

The size of planning units should match the spatial resolution and accuracy of the data available (Mills et al. 2010): thus, regional-scale prioritisations based on broad-scale data will require larger planning units than those undertaken at local scales with fine-resolution data. However, a problem arises if planning units are larger than the size at which MPAs are likely to be implemented. The software assumes that each planning unit selected for inclusion in an MPA system will be 100% protected, and selects sets of planning units to achieve conservation targets accordingly. Yet this assumption is clearly invalid if planning units are 1 km^2, whereas the typical size of MPAs in the region is 0.1 – 0.3 km^2, resulting in a significant underestimate of the number of planning units required to achieve targets. This problem can be overcome using a feature of Marxan with Zones (an extension to the original Marxan software, Watts et al. 2009) that allows users to specify how much each planning unit contributes towards the overall target. For example, applying a contribution fraction of 0.2 indicates that only 20% of the area of each selected planning unit will contribute towards conservation targets.

7.2.2. MPA network design

The process of designing local MPA networks within regional priority areas most closely integrates systematic conservation planning with community-based implementation. MPA network design should be scientifically rigorous but socially flexible (Agardy 2005), involving local communities in each stage, including the definition of MPA network goals and objectives, collection of fine-scale data, reviewing the effectiveness of existing MPAs and selecting sites in which to designate new MPAs (Figure 7.1).

Designing and implementing MPA networks at local, as opposed to regional, scales has several advantages. Fewer communities are involved in each network, reducing transaction costs (McDonald 2009), and increasing the likelihood of effective cooperative governance (Christie et al. 2009). Strong social networks linking 'MPA communities' can be developed and enforced through regular meetings, which may be more costly or difficult to organise over larger spatial scales. The cost of managing MPAs may be reduced (Alino et al. 2006), for example if infrastructure such as enforcement patrol boats can be shared between two or more MPAs. Finally, revenue-sharing mechanisms can be developed to distribute income from user fees across MPA communities (Emerton et al.

2006), providing incentives for some communities to protect habitats that are less likely to attract tourists (which will be necessary to achieve a fully representative MPA network (Chapter 4)). Such mechanisms are more likely to be effective if funds are managed at local scales; for example, community members criticised the collection and subsequent re-allocation of user fees collected from Apo Island Marine Sanctuary by the (national) Department of Environment and Natural Resources for corruption and a lack of financial transparency (Hind et al. 2010).

In designing MPA networks at local scales, planners can take advantage of fine-scale biodiversity and socioeconomic data. Collection of such data would not be feasible at regional scales. In many cases, new data will need to be collected, or existing datasets verified and refined. Participatory coastal resource assessment, which involves community members in mapping the spatial distribution of conservation features and resource use, simultaneously fulfils data requirements for planning and provides opportunities to engage community members in conservation (Olson et al. 2005; Aswani & Lauer 2006). Local ecological knowledge, for example of the location and timing of spawning aggregations, may also be an effective substitute in regions with few scientific survey data (Aswani & Hamilton 2004b; Granek & Brown 2005; Aswani & Lauer 2006; Ban et al. 2009b; Lavides et al. 2010). Collection of fine-scale data may result in modifications to regional datasets and objectives; for example, interviews conducted to map the spatial distribution of fishing effort in Siquijor Province (Weeks et al. 2010b, Chapter 5) identified several deep shoals that do not appear on nautical charts and were not previously known to local researchers. Examples of fine-scale socioeconomic data that may be considered in the design of local MPA networks are illustrated in Figure 7.4; these include the spatial distribution of resource use, local marine tenure boundaries, the locations of dive sites and existing MPAs.

Local-scale MPA network objectives should be informed by scientific guidelines and the regional conservation context of the area, identified during phase one. These considerations will be supplemented or adjusted by local community objectives for biodiversity, fisheries management and socioeconomic development. Adopting a participatory approach to establishing MPA network objectives and measuring progress made towards them in existing MPAs is most likely to result in community awareness of the need for increased protection and subsequent support for the designation of additional no-take areas (White et al. 2002; Christie et al. 2005; Fernandes et al. 2005; Alcala & Russ 2006).

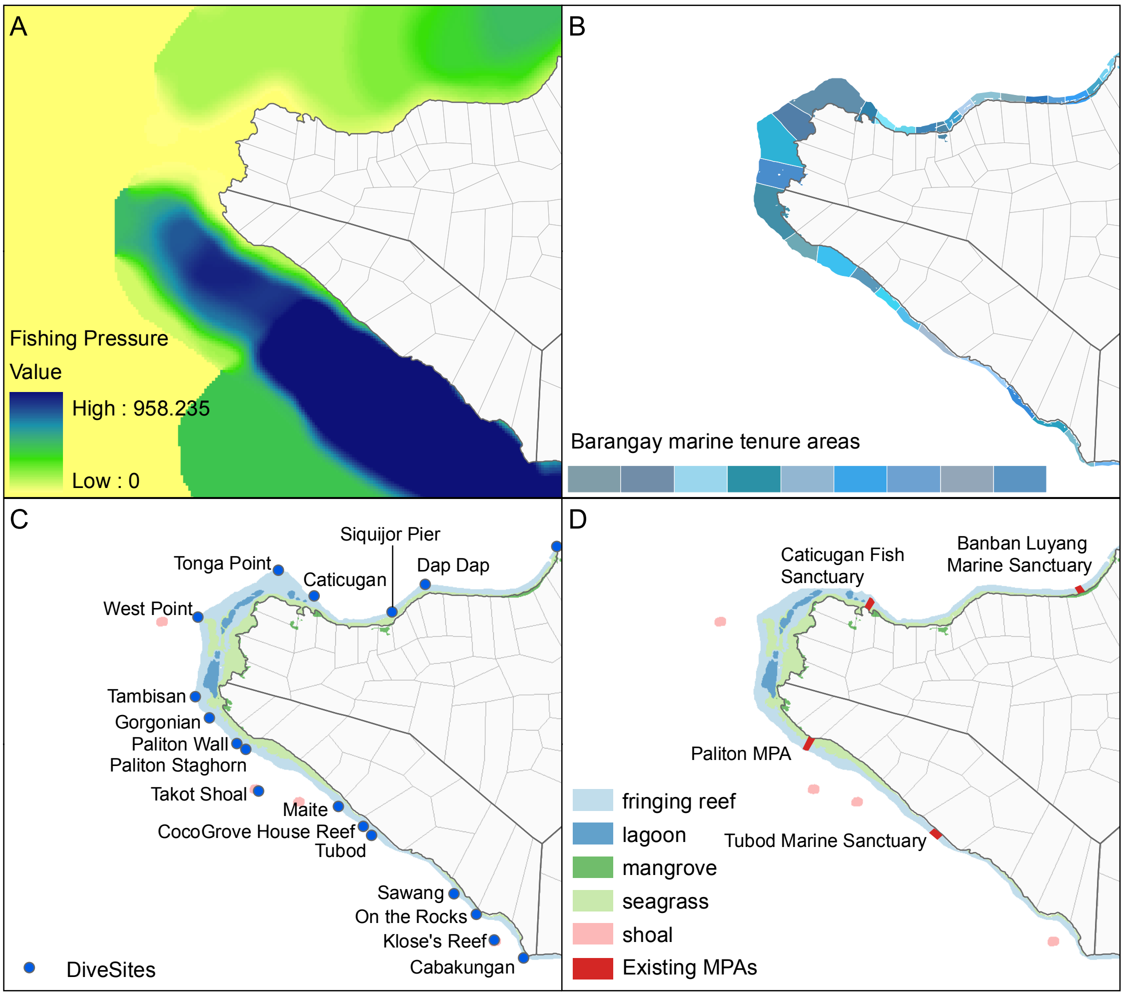

Figure 7.4. Examples of fine-scale socioeconomic data that can be considered in the design of local MPA networks (here, in the Western Siquijor priority area shown in Fig. 7.3): (A) the spatial distribution of small-scale fishing effort, collected and mapped through interviews (Weeks et al. 2010b, Chapter 5); (B) the locations of marine tenure boundaries between *barangays* (Weeks et al 2010c, Chapter 6); (C) the location of dive sites frequented by tourists; (D) the location of existing MPAs (Weeks et al. 2010a, Chapter 2).

Socioeconomic operating principles for MPA network design have frequently been defined less precisely than their ecological counterparts. For example: "*ensure that final selection of no-take areas recognises social costs and benefits*" (Fernandes et al. 2005); "*minimise negative impacts on existing livelihood strategies*" (Green et al. 2009). However, given that socioeconomic objectives are likely to be equally important to their ecological counterparts in the design of local-scale MPA networks, they should also be defined explicitly. For example, quantitative targets can be established for the minimum area of fishing grounds that must remain open to fishing (Weeks et al. 2010c, Chapter 6). Suggested socioeconomic operating principles for MPA network design in the Philippines are outlined in Table 7.4. Nevertheless, emphasis should be placed on encouraging local communities to define (and justify) their own principles, rather than dictating what these should be. Again, socioeconomic operating principles in the Philippines are likely to differ markedly from those applied in other regions. For example, strategies to maximise the manageability and public recognition of zoning schemes in Australia recommend "fewer and larger" no-take zones (Fernandes et al. 2005), and to implement simple "coordinate-based boundaries" (Lewis et al. 2003). In the Philippines, compliance is facilitated by placing MPAs within sight of a community (Pollnac et al. 2001; McClanahan et al. 2006; Ban et al. 2009a) and physically marking boundaries with buoys (Pollnac et al. 2001; Walmsley & White 2003).

When selecting sites to form a local MPA network, existing MPAs should be 'locked in' to the design. Given their small size, existing MPAs that are well supported and effectively managed are unlikely to represent opportunity costs to MPA network development (Stewart et al. 2003; Ban et al. 2009a). Attempts to change their location may undermine local support (Lowry et al. 2009). However, if existing MPAs are found to be socially and ecologically ineffective, it may be better to 'start from scratch' rather than to pursue enforcement of an MPA that is perceived to have failed.

Present approaches to community-based MPA implementation in the Philippines have not always resulted in the equitable distribution of costs and benefits among and throughout coastal communities (Walmsley & White 2003; Christie 2004; Leisher et al. 2007, Chapter 4). Decision support tools such as Marxan with Zones may assist planning teams to achieve this goal, by facilitating explicit consideration of the costs and benefits of alternative MPA configurations to different resource users, who may be defined by community identity, gear usage, or target species (Weeks et al. 2010c, Chapter 6). Whilst conservation planning software does not provide a perfect or final design, it can be a

Table 7.4. Proposed socioeconomic feasibility principles for MPA network design in the Philippines.

Principle	Comments
Minimise displacement of municipal (small-scale and subsistence) fishers.	Municipal fishers, defined as those fishing within municipal waters using fishing vessels of three gross tons or less, or fishing not requiring the use of fishing vessels, are the primary stakeholders for conservation planning in coastal waters.
Designate no more than 15% of the inshore fishing grounds used by fishers from each *barangay* as no-take areas, so that a minimum of 85% of fishing area per *barangay* remains open to fishing	Setting explicit targets for the area of fishing grounds that must remain open to fishing is likely to engage fishers in participatory planning. In most instances it should be possible to achieve conservation targets without designating more than 20% of inshore fishing grounds as no-take areas (Chapter 6).
Ensure that all fishers maintain access to beaches and landing sites.	Where MPAs are designated 'no-entry' zones, it is important that fishers retain access to beaches were they may land catch or keep their vessel.
Where information on the spatial distribution of fishing effort by gear type is available, ensure that different groups are impacted equitably.	MPA networks that are designed to impact resource users equitably are less likely to result in social conflict, and are more likely to be socioeconomically acceptable.
Implement buffer zones, in which gleaning is permitted but no extraction of finfish or use of vessels is allowed, around no-take areas.	Gleaners are less able to redistribute effort following MPA implementation than other groups of fishers, and may as a result be the most vocal opponents to MPA designation (personal communication, Municipal Agricultural Office San Juan, Siquijor). Buffer zones in which gleaning is allowed may reduce this conflict, and allow for larger MPAs to be established.
Incorporate areas that the local community perceives to be important, e.g. places of biological, cultural, aesthetic, historic, physical, social, or tourism value.	Sites that have tourism value (e.g. established dive sites) may be preferred locations for MPAs, as entry fees may contribute towards management costs and fisher compensation.
Exclude commercial fishers from operating in municipal waters (within 15 km of the coastline).	The Philippine Fisheries Code of 1998 (RA8550) provides municipal governments with the option to authorise small and medium scale commercial fishing vessels to operate between 10.1 km and 15 km from the shoreline. Authorising this usage may cause conflict between municipal and commercial fishers, and is not recommended.
Maintain no-take protection for all existing MPAs, except where they have been demonstrated to be ecologically and socially ineffective.	MPA networks should build upon and scale-up existing MPAs. Although they may not be optimally located, existing MPAs are unlikely to represent opportunity costs for future conservation efforts. However, if previously designated MPAs are not achieving management objectives and have poor levels of compliance, it may be better to identify a new site that will be more effective.

Table 7.4. (continued)

Principle	Comments
MPA boundaries should consider the sea boundaries between *barangays*.	Although it has been claimed that local marine tenure institutions have disappeared from the Philippines, evidence suggests that the spatial distribution of small-scale fishing effort does adhere to informal boundaries between *barangays*. Planners must therefore consider local-scale patterns of tenure, and ensure that implementation occurs through consultation at the *barangay* level, even though MPAs are designated through municipal legislation (Chapter 6).
Where possible, MPA boundaries should align with easily recognisable shore-based markers (e.g. a prominent rock, headland etc).	The concept that 'fewer and larger rather than more and smaller' MPAs facilitate compliance is not relevant in this context. Compliance is most effective where boundaries are clearly marked by shore-based markers and permanent buoys and MPAs are situated within sight of a village, guardhouse or resort that can provide surveillance.

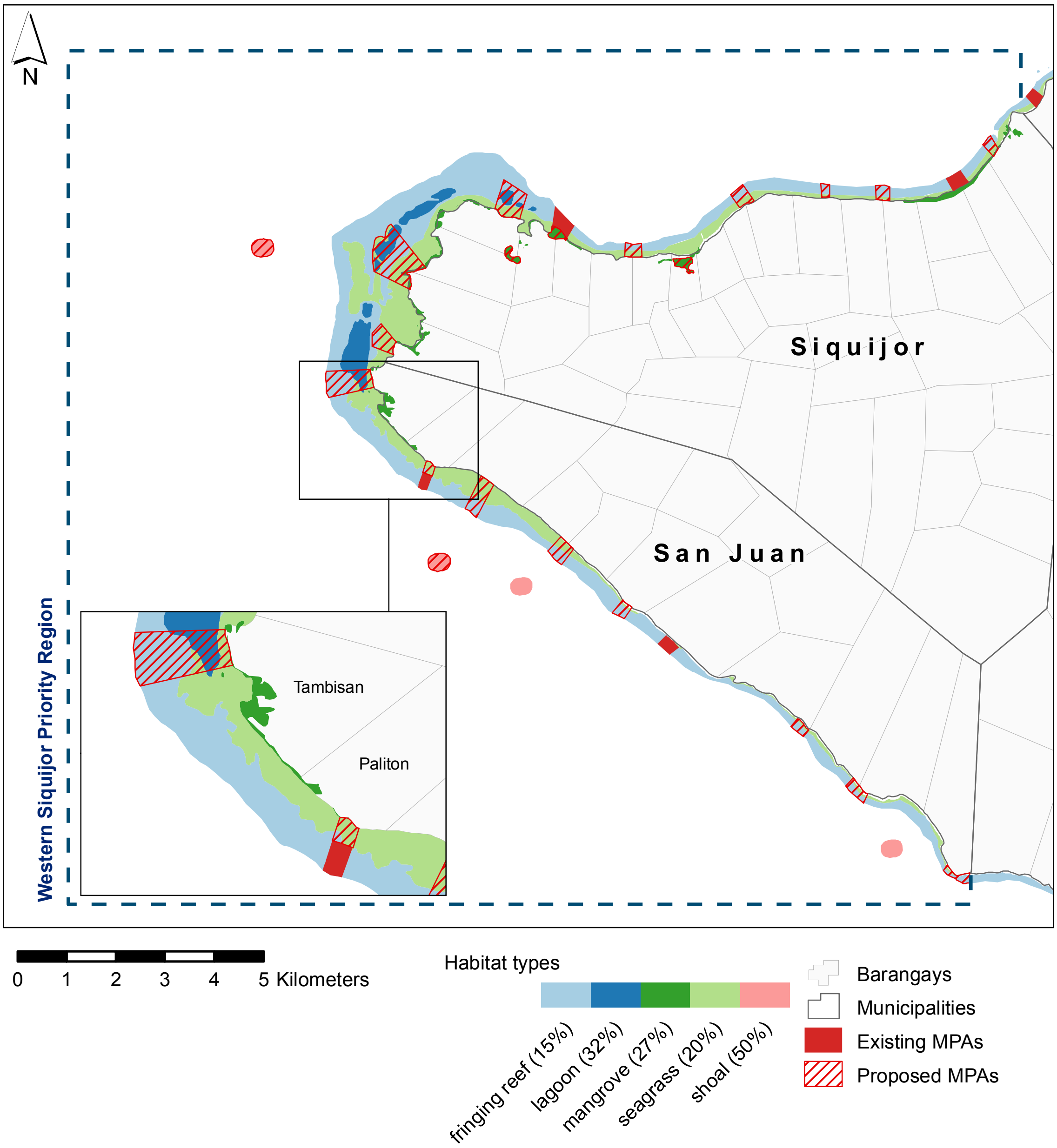

Figure 7.5. An example of a hypothetical local MPA network for the Western Siquijor priority region (see Figures 7.2 and 7.3 for regional context). Parentheses in the legend indicate the level of representation in the MPA network. This MPA network design illustrates the complex trade-offs encountered when trying to follow both ecological and socioeconomic design principles. Whilst targets for habitat representation are met and in some cases exceeded, most of the proposed MPAs are smaller than the absolute minimum size of 0.1 km^2, and only three MPAs exceed the suggested size of 0.3 km^2 (Table 7.3). Nevertheless, six coastal *barangays* (of 29) lose more than 20% of their inshore fishing grounds (Table 7.4). The inset highlights how design principles for local-scale connectivity have been followed, by creating new MPAs (Tambisan) and extending exiting MPAs (Paliton) so that they extend from the coastline to the bottom of the reef slope, to protect adjacent areas of seagrass, lagoon and coral reef habitat.

powerful tool to demonstrate the nature of such trade-offs and enable interactive exploration of different options regarding MPA size, placement and spacing.

It may not always be possible to adhere to both ecological and socioeconomic principles simultaneously. For example, ecological design principles may state that at least 10% of each targeted habitat type should be protected in no-take areas (Table 7.3), whereas socioeconomic feasibility principles suggest that at least 85% of fishing grounds remain open to fishing (Table 7.4). Some habitat configurations may make achieving these two objectives impossible, for example if a high percentage of a rare habitat type occurs solely within the fishing grounds of a single fishing community. In this instance, decisions will need to be made as to whether ecological or socioeconomic factors take priority (e.g. Fig. 7.5). In regions where resources are insufficient for active enforcement, and compliance with no-take area designation must therefore be voluntary, better conservation outcomes may be achieved through prioritising socioeconomic objectives. However, the relative importance of achieving ecological objectives and ensuring high levels of compliance, in terms of overall conservation effectiveness, remains uncertain, and requires further research.

7.2.3. Local-scale implementation

Following the selection of sites for the designation of new MPAs, this final phase involves delineation and implementation within individual *barangays*. Delineation of MPA boundaries should follow extensive consultations with all stakeholders within the community. It may also be necessary to consult with members of adjacent *barangays*, who may be affected by the displacement of fishers post-implementation. Following implementation, if community negotiations have resulted in representation of biodiversity that differs from that expected during the MPA network design phase (e.g. if greater or less area is protected, or if different features are represented), this information should be used to adjust targets elsewhere in the local MPA network.

Where possible, baseline data should be collected prior to MPA designation, to enable a scientifically rigorous before-after control-impact pair (BACIP) monitoring design (Russ 2002). The MPA Rating and Database System provides a standard framework for monitoring the ecological, social and management effectiveness of MPAs in the Philippines (White et al. 2006b). Direct participation of resource users in collecting and interpreting monitoring data provides a venue for increasing ecological awareness and knowledge (Pietri et al. 2009), and improves management effectiveness, as communities see the effects of MPA designation first hand (Samoilys et al. 2007; Almany et al. 2010). Local NGOs in the Central Visayas have been supporting community MPA monitoring through the provision of training in biophysical monitoring techniques, such as estimating fish abundances and substrate composition, and assistance with data analysis (Eisma-Osorio et al. 2009).

7.2.4. Decisions relating to spatial scale

Decisions need to be made regarding the spatial scale at which MPA networks are designed and implemented (Mills et al. 2010). These decisions will be influenced by ecological, socioeconomic and governance considerations. Here, I outline the decision-making process to identify an appropriate spatial scale for each phase in the planning framework.

Regional-scale planning and prioritisation

The majority of MPAs in the Philippines are legislated under Municipal Ordinances (Weeks et al. 2010a, Chapter 2). Therefore this is the basic unit and scale of governance that should be considered for MPA network planning (Eisma-Osorio et al. 2009). Christie et al (2009) suggest that an upper limit to collaborative MPA network management would be 10 to 15 municipalities. However, this extent is likely to be insufficient to realise the benefits of regional-scale planning (i.e. complementarity and connectivity) (Mills et al. 2010). In most cases, larger political divisions (i.e. provinces or regions) are inappropriate for marine conservation planning, as the archipelagic nature of the Philippines means that seascapes must often be co-managed by municipalities from different provinces. For example, the Tañon Strait, between the provinces of Negros Oriental and Cebu, is less than 10 km wide at its narrowest point. Planning regions for this first phase should therefore be defined on an ecological basis.

Six marine biogeographic regions have been identified for the Philippines (Ong et al. 2002, Fig. 2.1). However, these regions (in the order of 100,000 km^2 in size) may prove to be too large to allow oversight by a single planning organisation that can also engage with local stakeholders throughout the region. Ideally, the six existing bioregions would form the starting point for biogeographic classifications at finer spatial scales (Chapter 3). If insufficient data are available to enable this, seascapes at scales of 5,000 – 10,000 km^2 (e.g. the Tañon Strait, Camotes Sea, Leyte Gulf etc.) would make intuitive planning regions. Although Pressey and Bottril (2009) contend that seascapes are sufficiently small that there is a "more or less direct connection between the areas selected from maps as candidates for conservation and the areas in which conservation actions are eventually applied", this is unlikely to be true for MPA designation in the Philippines. However, implementation is not an objective of this phase, which aims to identify regional priority areas within which to undertake local scale MPA network design.

Local-scale MPA network design

Local-scale MPA network design is undertaken within priority areas identified during regional-scale planning. The extent of these priority areas will depend upon the capacity of the facilitating NGO, and the regional-scale conservation context for prioritising an area.

This planning phase requires environmental education programmes, capacity building and fine-scale data collection to be undertaken in most, if not all, the communities within the priority region. Thus, the number and size of communities, and the capacity of the organisation leading the MPA network design process (i.e. how many communities they can work with simultaneously) may restrict the spatial extent of MPA networks. The scale of local MPA networks should also consider the regional conservation context for the area. For example, if an area is identified as a regional priority for mangrove ecosystems, it may be appropriate to incorporate the full local extent of mangroves. Where offshore reefs or islands are prioritised for MPA implementation, they may form a local network with MPAs on nearby islands (e.g. Apo Island and the fringing reefs in Zamboanguita, Pamilacan Island and Cevera Shoal, Figure 7.2), or may only be connected with other MPAs at the regional scale (e.g. Selinog Island, Figure 7.2). Ideally, the scale of MPA networks should be large enough that they are substantially self-replenishing (Almany et al. 2009), to ensure that the communities who establish MPAs benefit from them (Foale & Manele 2004). Depending on the coastline length of the municipalities, MPA networks may include between two and six municipalities, well within the limit of effective governance suggested by Christie et al. (2009).

Local-scale implementation

Whilst the scale at which planning phases are undertaken may vary, implementation will always occur at the *barangay* scale (expect in the case of larger offshore MPAs). In most cases, community-based MPAs in the Philippines have been established within a single *barangay*; however, there are examples of collaboratively managed MPAs that span the boundary between two *barangays*. Such 'cross-boundary' MPAs may offer a way to increase the size of no-take areas: if two adjacent *barangays* both agree to give up 0.2 km^2 of their fishing grounds, an MPA that spans the boundary between them could be twice this size.

7.2.5. Roles and responsibilities in the planning process

As planning progresses from inception to implementation, the organisations involved in and responsible for leading the process will change. With this transfer of responsibility comes the need to build capacity at each subsequent level.

National governments set policies for natural resource management and may specify conservation priorities that determine the distribution of conservation funds throughout the country. In the Philippines, responsibility for coastal resource management has been devolved to municipal governments, so the role that the national government plays is likely to be reduced. Nevertheless, national government agencies such as the Department of Environment and Natural Resources, and the Bureau of Fisheries and Aquatic Resources will be partners in regional scale planning and MPA network design (Lowry et al. 2005).

International NGOs and funding bodies finance conservation initiatives, provide scientific guidance, and often direct regional conservation priorities. The Nature Conservancy (TNC), Conservation International (CI), the World Wide Fund for nature (WWF), the Global Environment Fund (GEF), United States Agency for International Development (USAID) and the Asian Development Bank (ADB) are all partner organisations in the Coral Triangle Initiative, and may be involved in MPA network development in the Philippines in some capacity. Although international NGOs have led conservation planning projects in other countries in the region (for example, TNC's work in Indonesia and Papua New Guinea), in the Philippines, local NGOs and research institutions (see below) are better placed to perform this role. International NGOs should therefore work in partnership with local organisations in the regional-scale planning phase to build their capacity and expertise.

Local NGOs and research institutions play a key role in facilitating MPA implementation and network development in the Philippines, providing initiative, direction, technical advice and funding to local governments and coastal communities, and often acting as co-managers of community-based coastal resource management projects (Rivera & Newkirk 1997; White et al. 2002; White et al. 2006a; Lowry et al. 2009). These organisations have a thorough understanding of the local context in which conservation plans need to be realised, and are therefore best positioned to integrate regional- and national-scale priorities with local-scale implementation. Unlike many big international NGOs, local NGOs frequently maintain long-term involvement with individual communities, providing ongoing support and training for effective management and monitoring post-implementation (Alcala 1988; Alcala & Russ 2006). Local NGOs will play a key role in identifying regional priority areas, within which they will lead the process of designing MPA networks.

Ideally, a single group will oversee the planning process within each region. This may be one NGO, or a group comprising representatives from several different facilitating organisations working in the region. If the latter, collaboration and cooperation between these groups will be essential to ensure that individual efforts complement one another. The Southeast Cebu Coastal Resource Management Council provides an example of how such groups may operate: comprising Mayors and Vice-Mayors of eight municipalities, supported by DENR, BFAR, the Philippine National Police and local NGOs, the Environmental Legal Assistance Center and CCEF, this group acts as a coordinating body for MPAs in eight municipalities (Eisma-Osorio et al. 2009).

Local government units, particularly at the municipal level, are key partners for MPA establishment (Lowry et al. 2005). Municipal (and to a lesser extent provincial) governments provide funding for MPA management and assist with enforcement (Lowry et al. 2009). Planning teams must also coordinate with government agencies responsible for managing other activities that may impact on the MPA network, such as commercial fisheries, coastal development, mangrove logging and oil prospecting (Lowry et al. 2005).

Finally, individual fishing communities are the ultimate determinants of the success or failure of MPA networks (Alcala & Russ 2006). Whilst it is not typically feasible to involve individual communities in the regional-scale planning and prioritisation phase, representatives from stakeholder communities should be involved throughout the process of MPA network design, including the collection of ecological and socioeconomic data, setting goals and objectives, and selection of sites to be designated as MPAs. Marine management committees comprising members of the local community, with support from local NGOs and governments, will take responsibility for the implementation and ongoing management of each MPA.

7.3. Discussion

Authors familiar with developing country contexts have been quick to warn against attempts to duplicate approaches to conservation planning that have been successful in countries with a very different economic, social, political and ecological environment (Christie & White 2007; Cinner 2007; Ban et al. 2009a; Christie et al. 2009; Clifton 2009). Increasingly, conservation practitioners are recognising that to achieve effective marine conservation in tropical developing countries there is a need to reconcile regional-scale planning with local-scale implementation (Agardy 2005; Alino et al. 2006; Mills et al. 2010). The framework that I present here provides one option for how regional-scale systematic conservation planning may be integrated with community-based approaches to MPA network development in the Philippines. It is unlikely to be a perfect solution for any individual situation, let alone all. Nevertheless, I hope that it provides an insight to local practitioners and international NGOs involved in the CTI, and other similar endeavours, as to how these apparently opposing approaches may be reconciled.

A hierarchical approach to MPA network development, whereby different objectives are emphasised at different spatial scales, is most likely to be effective where regional-scale conservation priorities set by international NGOs and funding organisations do not coincide with local community objectives for implementation (Agardy 2005; Gilliland & Laffoley 2008). However, this approach is not without limitations. Spatial patterns of conservation importance differ when identified using broad- versus fine-scale data (Rouget 2003; Banks & Skilleter 2007). This has implications for MPA network design, as regional prioritisations based on broad-scale data may overlook important areas where features exist, but have not been mapped. Furthermore, broad-scale surrogates may not be sufficient to ensure representation of fine-scale measures of biodiversity (Banks & Skilleter 2007). Nevertheless, a systematic conservation planning initiative on Réunion Island found that a hierarchical approach to conservation assessment that first used broad-scale habitats and processes at the regional scale, to guide fine-scale mapping of threatened species and degraded habitats in smaller areas was effective (Payet et al. 2010). To achieve national targets for MPA coverage in the Philippines, it is likely that MPAs will eventually need to be established in most, if not all, coastal municipalities (Weeks et al. 2010a, Chapter 2); thus, there is less risk that areas of high conservation value will be overlooked due to poor-resolution data, although they may be scheduled for implementation later than if fine-resolution data were available.

The use of broad-scale data in regional planning may also result in uncertain estimates of progress towards conservation targets, as there may be significant error in knowledge of the extent of marine habitats (Wabnitz et al. 2010). For this reason, feedback from local- to regional-scale planning, both in terms of revised spatial data and achievement of objectives (i.e. the amount of biodiversity features actually protected within MPAs, post-implementation) is essential.

Another weakness of a hierarchical framework is that it is dependent upon regional-scale priorities and objectives being retained effectively throughout local-scale planning and implementation, i.e. that areas identified as regional priorities for seagrass conservation end up with MPA networks that adequately protect seagrass habitats. Although education campaigns may seek to influence the goals and objectives of MPA networks, these must ultimately be determined by local communities, who may choose to disregard regional perspectives. Thus, until this strategy is empirically tested, its conservation effectiveness remains uncertain. Nevertheless, it seems probable that this approach will be more effective at achieving regional-scale objectives for biodiversity conservation than present opportunistic approaches (Chapter 4).

Whilst there is an evident need to increase the rate of MPA implementation (Weeks et al. 2010a, Chapter 2), there are also advantages to implementing MPAs and MPA networks sequentially. The extent and identity of biodiversity features eventually protected within each MPA may differ considerably from targets established during local and regional planning, both negatively, if socioeconomic constraints reduce the extent of habitats protected, and positively, if unforeseen opportunities arise (Mills et al. 2010). Sequential implementation allows conservation targets to be adapted in response to the outcomes achieved elsewhere in the planning region.

Many will think that the ecological principles for MPA network design suggested here (Table 7.3), particularly those that specify representation targets and minimum sizes for no-take areas, will be insufficient to achieve effective conservation of ecological features and processes . Nevertheless, given current levels of protection in the Philippines (presently c. 3% of coral reefs are protected within no-take areas, and the median size of no-take areas is 0.12 km^2 (Weeks et al. 2010a, Chapter 2)), these targets still represent a considerable challenge. Conservation planners should take care not to disenfranchise local communities by demanding that they give up more fishing area than they are happy to support; if an MPA is effective at achieving local objectives, such as improving catch per unit effort in adjacent areas (Russ et al. 2004) or providing alternative sources of income through tourism (Russ & Alcala 1999), support may grow through time, allowing no-take areas to be enlarged or supplemented at a later date.

A key element of many systematic conservation planning projects is the use of decision-support software. In contrast, these tools have rarely been applied to community-based conservation projects. Their use remains largely the domain of academic researchers, big international NGOs (e.g. TNC and WWF), and planning authorities in developed countries (e.g. the Great Barrier Reef Marine Park Authority). Notable examples where conservation planning software (Marxan) has been applied to MPA network planning in developing counties include Berau Marine Conservation Area and Wakatobi Marine National Park in Indonesia (TNC et al. 2008), and Kimbe Bay in Papua New Guinea (TNC et al. 2008; Green et al. 2009). Implementation is ongoing in all of these projects, which were led by TNC.

There are several reasons why local NGOs may be hesitant to use conservation planning software in the design of community-based MPA networks. These include: uncertainty regarding the technological and data requirements of software packages (Didier et al. 2009); the perception that such software is difficult to use (Smith et al. 2006; Didier et al. 2009); that software is difficult to communicate with stakeholders (TNC et al. 2008); and the misconception that decision-support tools are only relevant to top-down, centralised planning (Christie & White 2007), and as such are unable to account for the socioeconomic complexities encountered in many tropical developing countries (Clifton 2009). Nevertheless, I recognise two areas where conservation planning software can assist in the development of MPA networks in the Philippines: first, in the identification of regional-scale priority areas, and second, in the design of local-scale MPA networks.

Unfortunately, the bias of previous applications towards large-scale planning in developed countries has perpetuated the belief that conservation planning software is only useful in such contexts. In reality, Marxan is a flexible tool that can be applied to answer a wide variety of questions in many different situations (Ardron et al. 2008). The idea that there is an "absence of sociopolitical variables in Marxan" (Clifton 2009) is completely false, as demonstrated by an increasing body of literature which applies this software specifically to address trade-offs between competing socioeconomic uses (Adams et al. 2009; Ban & Vincent 2009; Klein et al. 2009). For example, Weeks et al (2010c, Chapter 6) demonstrate that Marxan can account for local-scale socioeconomic constraints, such as the location of marine tenure boundaries, and the need to distribute the costs of MPA network implementation equitably across different stakeholder groups. More case studies demonstrating the application of conservation planning software in tropical developing country contexts (e.g. Ban et al. 2009a; Green et al. 2009; Weeks et al. 2010b; c, Chapter 5, Chapter 6) will help to dispel this misconception.

Nevertheless, to use Marxan effectively does require an in-depth conceptual and methodological understanding of both Marxan and GIS (Ardron et al. 2008), which local NGOs might not have. One option to overcome this constraint would be to employ international consultants to undertake spatial prioritisation analyses. However, funding might be better directed towards providing training to build the capacity of local NGOs to use conservation planning software confidently and effectively. Researchers involved in software development may also benefit from such exchanges, by better understanding the problems that practitioners wish to solve and the data limitations they have to work within. It is important to note however, that the use of decision-support software is not a requirement for systematic conservation planning. The same approach to prioritisation, whereby potential sites are assigned biodiversity and cost values and are selected on the basis of explicit targets for representation and complementarity, can be achieved manually, e.g. using paper overlays (Smith et al. 2009). Although undoubtedly more time-consuming, designing MPA networks 'by hand' may better engage communities in exploration of alternative solutions where computer literacy is low.

Many different frameworks for systematic conservation planning have been proposed (e.g. Margules & Pressey 2000; Groves et al. 2002; Cowling & Pressey 2003; Knight et al. 2006a; Pressey & Bottrill 2009; Banks & Skilleter 2010; Foley et al. 2010). In fact, it is likely that a greater number of frameworks exist for this approach than there have been faithful applications. Community-based approaches to conservation planning are, on the other hand, less well documented (but see Alcala 1998; White et al. 2002; TNC et al. 2008). The absence of a strategic framework for community-based conservation does not necessarily imply that the stages outlined here have not been undertaken in the development of MPA networks in the Philippines (or elsewhere). Indeed, many would be familiar to organisations facilitating MPA establishment. Nevertheless, there are important advantages to an explicitly stated framework. Documentation of case studies against a standard framework facilitates sharing of successes and failures, and allows the relative advantages and limitations of alternative approaches at each stage to be better understood (Margules & Pressey 2000; Pressey & Bottrill 2009). Furthermore, an explicit framework to guide the process of MPA network development may be vital in regions that do not have a history of NGO involvement.

The framework I propose here bears similarities to the approach undertaken by TNC to design a network of MPAs for Kimbe Bay, Papua New Guinea (Green et al. 2009). TNC used a systematic conservation planning process to identify 14 'areas of interest' for Kimbe Bay, within which they are currently working with local communities to implement locally managed marine areas. However, they do not offer a framework for how regional-scale planning will inform conservation efforts within areas of interest, and it is unclear whether explicit targets will be established for local-scale planning. An alternative approach to integrating systematic conservation planning with community-based management was explored by Ban et al (2009b), who incorporated priority areas identified through interviews with indigenous resource users in two regions of British

Columbia, Canada, into science-based MPA networks identified using Marxan. They found that community members preferred this integrated solution, which augmented the areas they identified for protection with additional sites to achieve conservation objectives. Others have advocated hybrids between customary and 'western' management systems, for example establishing MPAs within customary governance structures (Cinner & Aswani 2007). In the Roviana and Vonavona Lagoons, Solomon Islands, a system of 23 MPAs was established under customary sea tenure following integration of indigenous ecological knowledge and scientific rationale (Aswani & Hamilton 2004b).

Many of the preconditions for applying the proposed framework are already in place in the Philippines, which leads the way amongst the Coral Triangle countries (and other tropical developing countries worldwide) in terms of both the number of existing MPAs and capacity for MPA network development. Decentralised governance of marine resources (Christie & White 1997), a strong momentum for MPA establishment (Weeks et al. 2010a, Chapter 2), the existence of successful community-based MPAs to use as a template (Alcala & Russ 2006), local NGOs that have established relationships with both local communities and international funding organisations (Armada et al. 2009; Eisma-Osorio et al. 2009), and a national MPA database and support network (Campos & Alino 2008) all provide a strong foundation for the development of a national system of comprehensive, ecologically representative, connected and resilient MPA networks. Nevertheless, progress needs to be made in several respects to enable a systematic framework for MPA network development, as outlined here, to be adopted nationwide.

The bias in the distribution of MPAs towards the Central Visayas (Fig. 2.1) reflects the location of local NGOs and academic institutions that facilitate MPA establishment in this region (including Silliman University, CCEF, FISH, Project Seahorse, LMMA and others) (Weeks et al. 2010a, Chapter 2). These NGOs have a wealth of experience in community-based MPA establishment, but may benefit from training in the use of conservation planning software, GIS and spatial data management, and how to interpret scientific guidance for resilient MPA network design in the local context. Other regions of the Philippines have fewer existing MPAs and less NGO presence; facilitating organisations will need to be established in these regions, ideally with assistance from NGOs currently operating in other parts of the country. Investment arising from the CTI should focus on building capacity at this level.

Sustainable financing represents a significant obstacle for the successful implementation of MPAs and MPA networks in the Philippines. Whilst some MPAs generate a significant portion of management costs through visitor fees (Vogt 1997; Cadiz & Calumpong 2000), this model cannot be expanded indefinitely, and will not be effective in regions with poor tourist infrastructure or to which many countries advise their citizens against travelling to (e.g. Mindanao, in the Southern Philippines). Identifying sustainable financing mechanisms, through revenue sharing at the appropriate spatial scale or otherwise, represents a significant challenge to conservation planners.

In summary, whilst community-based approaches applied in the Philippines have been successful in establishing MPAs to achieve local-scale objectives, they lack the scientific basis and spatial context required to develop ecologically functional MPA networks. Systematic conservation planning approaches provide this perspective, but are likely to be hindered by a lack of appropriate data, and may ultimately fail if they are not accepted by local stakeholders. In this chapter, I have proposed a new planning framework that incorporates elements of both regional-scale conservation planning and community-based implementation. Although yet to be empirically tested, this approach is most likely to achieve a satisfactory trade-off between achieving regional-scale objectives for biodiversity conservation and addressing the concerns of local fishing communities.

8. General Discussion

The aim of this thesis was to investigate the potential for systematic conservation planning to support the development of ecologically functional MPA networks in the Philippines. I first determined the need for a new approach to MPA establishment. I demonstrated that the existing MPA system in the Philippines falls a long way short of achieving national conservation targets (Chapter 2), and that even in regions where they have been applied extensively, community-based approaches to MPA implementation have not resulted in comprehensive, representative and adequate MPA networks (Chapter 4). I then addressed the problem of how to make systematic conservation planning relevant to the socioeconomic context of the Philippines. I examined the effectiveness of different data surrogates for small-scale fishing effort (Chapter 5), highlighting the need to collect data that are relevant to the spatial scale at which MPA networks will be designed and implemented. I then demonstrated, for the first time, how conservation planning software can be used to distribute the negative impacts of MPA network implementation equitably across different fishing communities in regions with local marine tenure (Chapter 6). Finally, I proposed a hierarchical framework for MPA network design and implementation that borrows from both systematic conservation planning, to achieve regional-scale objectives for biodiversity conservation, and community-based implementation, to ensure that the interests and needs of local communities are adequately represented (Chapter 7).

8.1. Key results and ideas arising from the thesis

The field of conservation planning is evolving rapidly (Pressey & Bottrill 2008), and planners are increasingly developing innovative ways to incorporate ecological, social, economic and political considerations into the design of MPA networks (Aswani & Hamilton 2004a; Klein et al. 2008; Ban et al. 2009b; Ban & Vincent 2009; Green et al. 2009; Klein et al. 2009; Edwards et al. 2010). This thesis adds to these efforts, and provides several novel contributions. The thesis moves beyond stating the obvious conclusion, that to achieve effective conservation no-take MPAs in the Philippines need to be larger and more numerous, to identify pragmatic recommendations to design MPA networks that will be supported by local stakeholders, and that will also contribute effectively towards conservation objectives at broader spatial scales.

The result that the current MPA system in the Philippines fails to achieve national targets for MPA coverage (Chapter 2) is intuitive, but important to quantify nonetheless. What may be more surprising to those unfamiliar with the region is the extraordinary number of community-based MPAs that have been established in the Philippines. This thesis thus not only highlights deficiencies in the existing MPA system, but also draws attention to the substantial momentum for MPA establishment and the capacity of local NGOs to achieve real conservation progress in a challenging environment.

I found that community-based approaches to MPA establishment in the Western Bohol Sea, although effective at achieving a wide range of local-scale objectives, have not resulted in an ecologically functional MPA network (Chapter 4). This result is fundamental to the development of an MPA network strategy for the Coral Triangle region (Coral Triangle Initiative 2008). The Western Bohol Sea has the highest concentration of MPAs in the Philippines, which itself leads the way amongst the Coral Triangle countries in terms of both the number of MPAs designated and local capacity for MPA network development. If the community-based MPAs in this region do not form a network capable of achieving regional-scale conservation objectives, it is unlikely that similar approaches will successfully achieve this goal anywhere in the region. This result provides a clear indication that a new approach to MPA network development in the Coral Triangle region is needed.

This thesis highlights the need to tailor conservation planning strategies to the socioeconomic, political (i.e. governance) and ecological context of the region in which they are to be applied. Whilst the concept of a universal framework for conservation planning is attractive (Pressey & Bottrill 2009), any such framework must be sufficiently flexible to allow interpretation in a range of different contexts. An important consideration is the mismatch between the relatively broad spatial scale at which conservation planning has previously been undertaken, and the very fine spatial scale at which it must be applied in the Philippines (Mills et al. 2010). The need to match the spatial resolution of biodiversity data surrogates to the scale at which MPA networks are designed and implemented has been established (e.g. Rouget 2003); research presented in this thesis demonstrates that similar considerations are necessary when selecting socioeconomic datasets to represent the interests of stakeholders (Chapter 5). Likewise, ecological principles for MPA network design (see Tables 4.3 and 7.3) must be relevant to the region in which they are to be applied (Chapters 4 and 7). Placing emphasis on unattainable targets, for example for the size of individual MPAs or the total area of habitat to be protected, is likely to be counterproductive (Agardy et al. 2003; Carwardine et al. 2009). Nevertheless, results from Chapter 2 emphasise the need to establish large (i.e. >100 km^2) no-take MPAs wherever the opportunity arises, for example on offshore reefs or in areas with low fishing effort.

In Chapter 6 I demonstrate how spatial zoning software (Marxan with Zones) can be used to incorporate local marine tenure into the design of MPA networks in the Philippines. To my knowledge, this is the first time that social, as opposed to economic factors, have been explicitly considered in the systematic design of MPA networks. This work thus constitutes a significant and novel contribution to the field of conservation planning. My results suggest that local marine tenure can provide both constraints and opportunities for MPA network design. Where tenure operates at a fine spatial scale, it is likely to constrain the size of individual MPAs. However, if MPA networks are carefully designed to distribute the negative impacts of implementation equitably, local communities might in fact support larger no-take closures overall. This counter-intuitive result is in contrast to previously stated concepts of 'efficiency' in conservation planning, which emphasise minimising the overall area or cost of protected area networks (e.g. Stewart & Possingham 2005). The approach I demonstrate is not only relevant to regions with local marine tenure, but could be applied anywhere where spatial resource use is subject to local-scale constraints; these might include habitat-specific gear types or the different spatial mobility of fishers using motorised versus non-motorised vessels (Fabinyi 2010).

8.2. Limitations of the thesis and unresolved challenges

In exploring systematic approaches to designing MPA networks, the biodiversity data that informed my conservation targets were limited to broad-scale habitat types identified from satellite imagery, and bioregional classifications developed from reef fish survey data (Chapter 3). In Chapter 7, I also included environmental data, setting objectives for representation of coral reef habitats in areas with high, medium and low current, slope, and exposure to river runoff. The resolution at which biodiversity features are mapped (e.g. broad-scale versus fine-scale habitat classes, species- versus family-level distribution data) affects the relative conservation value of planning units, and thus which sites are prioritised for inclusion in MPA networks (Rouget 2003; Gladstone & Alexander 2005; Banks & Skilleter 2007; Dalleau et al. 2010). In Chapters 5 and 6, I found that MPA network design was strongly influenced by socioeconomic objectives. If fine-scale biodiversity data were available, it is likely that there would have been less flexibility in the spatial configuration of sites to achieve biodiversity objectives, and it might have been more difficult to achieve both conservation and socioeconomic targets simultaneously. Nevertheless, the data that I used are representative of the types of data that will be available for regional-scale conservation planning in the Philippines. Given that the final locations of MPAs will be determined primarily by socioeconomic factors, investing in further biodiversity data collection might not improve the effectiveness of conservation plans. However, where MPAs are necessarily small, accurate data on the locations of spawning aggregations for example, could greatly increase the conservation impact of MPAs (Green et al. 2009; McLeod et al. 2009).

In Chapter 5, I sought to identify appropriate socioeconomic data surrogates for small-scale fishing effort. Empirical data on the distribution of fishing effort were collected through interviews with fishers in each of the coastal communities around the island of Siquijor. Time constraints allowed only one interview per community, and interviewees were asked to summarise patterns of fishing effort for all fishers in their community. Thus, whilst the data collected were adequate for the analyses performed, more accurate information might be obtained from a greater number of interviews, and the location of fishing grounds could be verified by accompanying fishers with a GPS. A more complete understanding of the social and cultural factors driving the spatial distribution of fishing effort may also be informative (Johannes 2002; Aswani & Hamilton 2004a; Foale & Manele 2004). Finally, fishers' spatial preferences were typically not described at a sufficiently fine scale to guide the placement of individual MPAs. This indicates that planners will need to undertake more intensive surveys when identifying locations for MPA implementation.

I assumed that minimising opportunity costs to small-scale fishers, by placing MPAs in areas with lower fishing effort where possible, would increase the likelihood that local communities would support and comply with MPA network implementation, resulting in more effective conservation. This assumption is supported by the locations of existing MPAs, which are in areas with relatively low fishing pressure (Weeks et al. 2010b, Chapter 5). Yet many other factors will also influence community support for conservation initiatives. For example, a perception of fishery decline, the presence of a clear MPA leader, community participation in cross visits and environmental education programs are all associated with strong community support for MPAs (White & Vogt 2000; Pollnac et al. 2001; Pietri et al. 2009). A better understanding of how these factors interact will enable conservation planners to better identify, create and respond to conservation opportunities (Knight & Cowling 2007; Cowling et al. 2010).

At first glance, the recommendations arising from Chapters 5 and 6 appear to be contradictory. In Chapter 5, I highlight the importance of collecting empirical data on spatial patterns of resource use to minimise the negative impacts of MPA network implementation on small-scale fishers; MPA network designs are thus biased towards areas with lower fishing pressure. In Chapter 6, I emphasise the importance of distributing the costs of implementation equitably across local fishing communities; resulting in fragmented MPA network designs. If the no-take area designated must be distributed equitably around the island, it appears that information on the relative distribution of fishing effort becomes redundant. However, despite apparent contradictions, both approaches are informative. The first, identifying conservation priorities in areas with relatively low fishing effort, will assist planners to identify opportunities to implement MPAs more easily. Limited resources available for conservation initiatives dictate that implementation will occur incrementally (Pressey et al. 2007; Stewart et al. 2007). These communities can thus be scheduled for implementation earlier than those in areas with higher fishing effort, where MPA implementation might be more contentious. The second approach to MPA network design, that incorporates the constraints of local tenure, provides a more pragmatic indication of how conservation objectives can be achieved and a valuable tool to explore trade-offs between competing objectives.

How small-scale fishers in the Philippines redistribute fishing effort in response to MPA implementation is unknown. The capacity of these fishers to transfer effort to new locations is much reduced compared to commercial or recreational fishers in developed countries (Valcic 2009). Anecdotal evidence from fisher interviews (Chapter 5) suggests that in the absence of formal tenure agreements, these tenure systems may break down, with fishers redistributing effort into fishing areas associated with adjacent communities. This may be a source of conflict, as fishers from adjacent communities perceive that they are 'losing' fishing area, but do not have access to the immediate benefits of the MPA, such as income from user fees. Thus, a better understanding of how small-scale fishers redistribute effort in response to MPA implementation is required both to ensure that displacement of fishing activity does not negate the ecological benefits gained from MPA establishment (Salas & Gaertner 2004; Powers & Abeare 2009), and to reduce the potential for social conflict.

8.3. What can developed countries, such as Australia, learn from the Philippines?

The focus of much of this thesis has been on how MPA network development in the Philippines may benefit from science-based approaches, developed and tested in countries such as Australia. However, there are also areas in which conservation planners working in developed countries might learn from their counterparts in the Philippines.

The rezoning of the Great Barrier Reef Marine Park (GBRMP) in 2004 (Fernandes et al. 2005) created the world's largest network of no-take MPAs, and is the best known example of a no-take MPA network that has been implemented following a systematic conservation planning approach. Whilst this initiative is generally considered to have been very successful (Russ et al. 2008; McCook et al. 2010), there remains room for improvement. Despite extensive public consultation during the planning process (Fernandes et al. 2005), surveys conducted three years after implementation revealed that the majority of commercial, charter and recreational fishers on the GBR did not believe that they were adequately consulted about the zoning changes (Sutton & Tobin 2009; McCook et al. 2010). This is notable, given that fishers who did believe that consultation was adequate were significantly more likely to express support for the plan (Sutton & Tobin 2009). In addition, compensation provided to commercial fishers who perceived that they had been negatively impacted by the rezoning was an order of magnitude greater than initial estimates (Macintosh et al. 2010). Thus, even though countries such as Australia have the capacity to plan and implement MPA networks across large spatial scales (the GBRMP covers 344,400 km^2) (Mills et al. 2010), it appears that the need to meaningfully involve stakeholders at the local scale is no less important than it is in the Philippines. Conservation planners should be mindful of this when pursuing the benefits in efficiency that may result from coordinating conservation efforts across national or even multinational scales (Kark et al. 2009).

Following the rezoning of the GBR, and other similar initiatives such as that undertaken in Moreton Bay, Queensland, Australia (McPhee et al. 2007), a guidance statement for MPA network design in Australia was published in 2009 (The Ecology Centre UQ 2009). This statement includes recommendations to "*devolve management of MPAs to the lowest practical level to assist with integration of management, development of incentives across spatio-temporal scales and levels, and improve learning, monitoring and compliance systems*" and to "*encourage community acceptance and ownership*" of MPAs. This statement comes 18 years after responsibility for managing marine resources in the Philippines was legally devolved from national to municipal governments (through the Philippine Local Government Code of 1991), and four decades after the first no-take MPAs were established following an approach that strongly emphasised community involvement and ownership (Russ & Alcala 1999; White et al. 2002; Alcala & Russ 2006). Thus, it appears that conservation planners in Australia are now recognising that factors that have underpinned successful MPA establishment in the Philippines are also relevant in their region.

8.4. Future research and applications

Conservation biology is an applied discipline, in which facilitating conservation action is as important, if not more so, than conducting theoretical research (Knight et al. 2008). The scope and demands of a doctoral thesis provide limited opportunity to make practical contributions in addition to theoretical ones. Nevertheless, this thesis has created opportunities to influence conservation action in the Philippines.

The MPA database created for Chapter 2 has been included in the Indigenous and Community Conserved Areas (ICCA) Registry (http://www.iccaregistry.org), for which the Philippines is one of four pilot countries. The ICCA Registry is being developed following the same structure as the World Database on Protected Areas, and is designed to raise awareness of the contribution that community-based protected areas make to conservation efforts worldwide. This project, initiated by the United Nations Environment Program World Conservation Monitoring Centre (UNEP-WCMC), addresses the underrepresentation of community-based MPAs in existing databases and previous gap analyses, a problem that I raised in Section 2.4.

The study region for Chapter 4 was the Western Bohol Sea. In this thesis I concluded that the existing MPA system, although effective in terms of local-scale objectives, does not form an ecologically functional MPA network. A sophisticated larval dispersal model has since been developed for this region (R. Abesamis, personal communication) as part of a parallel PhD project (supervised by G. R. Russ and A. C. Alcala). Future research undertaken by JCU and SUAKCREM will integrate the results from both of these theses, to identify optimal locations for new MPAs to develop an ecologically connected MPA network in the Western Bohol Sea.

The study region for Chapters 5 and 6 was the island of Siquijor. Since conducting this research, I have been awarded funds from the NOAA International Coral Grant program to undertake a collaborative project between the ARC Centre of Excellence for Coral Reef Studies at JCU and the Coastal Conservation and Education Foundation (CCEF), a Philippine NGO. The dual aims of this project are to develop a Provincial MPA Network Management Plan for Siquijor, and to provide a case study that interprets the IUCN Guidelines for Conservation Planning (Table 7.1) in the context of the Philippines. This project will provide the opportunity to empirically test many of the ideas developed in this thesis, in particular the framework proposed in Chapter 7, and to utilise additional ecological and socioeconomic data that I collected, but have not yet fully explored, such as the distribution of fishing effort by gear type around the island. Outcomes from this future work, for which this thesis provided both concept and foundation, will include a greater scientific basis for coastal resource management in Siquijor and the implementation of at least one new MPA.

Finally, both the research presented in this thesis, and the work that will follow on from it, have the potential to inform the Coral Triangle Initiative Regional Plan of Action in a real and meaningful way.

8.5. Conclusion

A system of comprehensive, ecologically representative, connected and resilient MPA networks is required not only to conserve the Philippines' rich marine biodiversity, but also to ensure the sustainability of coastal fisheries and the human communities that are dependent upon them. In identifying how best to build upon existing MPAs to develop ecologically functional MPA networks, conservation planners do not face a choice between systematic and community-based approaches, but the challenge of identifying ways to successfully integrate the two. This thesis provides an important step towards achieving this goal, by providing pragmatic recommendations to local practitioners and international NGOs working in the region as to how this might be possible.

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