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A. Paulina Guarderas for the degree of Master of Science in Environmental Science

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Title: Marine Conservation in Latin America and the Caribbean: An Analysis of Marine Protected Areas (MPAs).

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Coastal and marine ecosystems in Latin America and the Caribbean are undergoing a rapid and drastic transformation. Dense human populations are concentrated in coastal areas, leading to increased coastal development, destruction of near-shore habitats, pollution, and overexploitation of marine resources. For most Latin American and Caribbean countries, the deterioration of coastal ecosystems is particularly critical due to the strong dependency of their economies on the quality of natural resources and ecosystems. Thus, the necessity of effectively conserving and managing marine ecosystems with a more integrative, ecosystem–based approach is urgent. Marine reserves constitute a powerful conservation tool for mitigating ocean

degradation. Because they provide spatial refuges for fished populations, and protect important habitats and their associated ecological interactions, they are particularly beneficial for counteracting the harmful effects of overfishing.

In Chapter 2 of this thesis, I present a comprehensive analysis of the status and progress of marine protected areas (MPAs), particularly no-take marine reserves in Latin America and the Caribbean. I also show that the number and area protected have increased through time, particularly since the 1980s; but the system of MPAs is still deficient in fully representing the whole array of marine biogeographic provinces. In addition, I demonstrate that no-take marine reserves are poorly utilized for conservation of marine biodiversity in this region. Finally, I highlight the need for strengthening the marine conservation initiative in Latin America and the Caribbean under a regional approach.

In Chapter 3 using meta-analytic methods, I quantitatively estimate the magnitude of the conservation effects of marine reserves in Latin America and the Caribbean. I examine the species and reserve characteristics that contribute to explain the variation in responses to protection. These analyses demonstrate positive outcomes of reserve protection at assemblage and species levels, and confirm the effectiveness of marine reserves as a conservation tool to rebuild exploited populations. Less clear is the relationship between density responses to protection and species-specific characteristics. Species with different trophic levels, adult mobility, body size and resilience can benefit from protection. Nevertheless, when I examine the effects of protection on one habitat type (coral reefs) using biomass as the response variable different trophic groups show differential responses. Predators demonstrated higher positive responses compared to herbivores or producers. In addition some indirect effects were disclosed.

Findings from this research have direct implications for the advancement of marine conservation in Latin America and the Caribbean. Chapter 2 provides an important tool for planning marine conservation strategies at a regional scale. Areas that need more protection are highlighted, especially networks of no-take marine reserves in the Eastern Pacific and Southern Atlantic. Additionally, this assessment can be used as a baseline to make future comparisons of the progress of marine biodiversity conservation in this region. Chapter 3 demonstrates the powerful effect of no-take marine reserves in restoring depleted populations and in some cases recovering ecological functions that have been lost due to overfishing in Latin American and the Caribbean.

Marine Conservation in Latin America and the Caribbean: An Analysis of Marine Protected Areas (MPAs)

by

A. Paulina Guarderas

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I understand that my thesis will become part of the permanent collection of Oregon State University libraries. My signature below authorizes release of my thesis to any reader upon request

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A. Paulina Guarderas, Author

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Marine Conservation in Latin America and the Caribbean: An Analysis of Marine Protected Areas (MPAs).

Chapter 1: General Introduction

Latin America and the Caribbean encompass a unique array of coastal ecosystems including mangroves, coral and rocky reefs, salt marshes, sea grasses, coastal lagoons, sandy beaches, kelp beds, and even fiords in the Southern region of Chile. Distinctive oceanic islands such as The Galapagos, Coco, Malpelo and Eastern Islands occur in the Pacific, while the Malvinas (Falkland) and Antillas Islands arise in the South Atlantic and Caribbean Sea, respectively (Kelleher et al. 1995).

The ocean masses that surround Latin America add significantly to the region's diversity and productivity, providing the background for economic development along the coast. Several of the world's largest and most productive estuaries occur in the region, such as those found at the mouth of the Amazon and Rio de la Plata Rivers in the Atlantic or the Guayas Gulf and the Gulf of Fonseca in the Pacific. The second largest coral system, the Mesoamerican Barrier Reef System, is located in the Caribbean Sea. The highly productive waters off Peru and Chile support one of the top five commercial fisheries in the world (IADB 1998).

However, the decline of marine environments documented worldwide, primarily as a result of overfishing, pollution and the direct and indirect impacts of climate change (Jackson et al. 2001, Myers and Worm 2003, Kappel 2005, MEA 2005, 2006), is also evident for this region. The coastal and marine areas of Latin

America are undergoing a rapid and drastic transformation (Sheppard 2000). Residential and commercial coastal development has modified the shoreline and caused degradation (Cohen et al. 1997). Sixty percent of the population is concentrated within 100 km of the coast. As a result, over-exploitation of marine resources is extensive. More than 80 percent of the commercially exploitable stocks in the South Western Atlantic and 40 percent in the South Eastern Pacific are fully fished, over-fished, or depleted (FAO 1995). In addition, there has been a proliferation of tourist resorts, uncontrolled discharge of wastes into the oceans, expansion of aquaculture, and a general lack of effective coastal regulations and enforcement (UNEP 2001).

For most Latin American and Caribbean countries, the deterioration of coastal ecosystems is particularly critical due to the strong dependency of their economies on the quality of natural resources and ecosystems. Therefore the necessity of effectively conserving and managing marine ecosystems with a more integrative, ecosystem–based approach is urgent (SCBD 2004). This approach recommends establishing a system of marine protected areas (MPAs) that include the following elements: (1) a representative network of highly protected areas, where all human extractive and destructive activities are prevented within their boundaries (no-take marine reserves), and (2) a complementary system of interconnected areas with different degrees of protection that support the biodiversity objectives of the highly protected network, and (3) sustainable resource extraction and control on environmental degradation outside MPA borders (SCBD 2004).

Despite these and other similar recommendations, there are few analyses of the existing MPAs and marine reserves in the region. In Chapter 2, I present a comprehensive analysis of the status and trends of MPAs, particularly no-take marine reserves in Latin America and the Caribbean. I show that the number and the area protected have increased through time, particularly since the 1980s. I also show that the existing MPAs are concentrated in some biogeographic regions and largely absent from others. In addition, I demonstrate that no-take marine reserves are poorly utilized for conservation of marine biodiversity in this region. Finally, I highlight the need for strengthening the marine conservation initiative in Latin America and the Caribbean under a regional approach.

Marine reserves constitute a powerful conservation tool (Lubchenco et al. 2003, Worm et al. 2006). They are particularly beneficial for counteracting the harmful effects of overfishing by providing spatial refuges for fished populations as well as protecting important habitats and their associated ecological interactions (NRC 2001, Sobel and Dahlgren 2003, Sladek Nowlis and Friedlander 2005). Recent syntheses of data from many reserve sites have shown that overall species richness, abundance (Côté et al. 2001), biomass, and size of marine organisms are consistently higher inside marine reserves than in nearby fished sites (Côté et al. 2001, Halpern 2003, Palumbi 2003a). However, the magnitude of responses to protection may vary based on the latitude and type of communities (Blyth-Skyrme et al. 2005). Furthermore, it is likely that not all species in a community will be equally affected within marine reserves. Species may respond differently to protection depending on the intensity of exploitation that each species experiences outside their boundaries, their trophic level, and their life history characteristics (Jennings et al. 1999, Mosquera 2000).

In Chapter 3 using meta-analytic methods, I quantitatively estimate the magnitude of the conservation effects of marine reserves in Latin America and the Caribbean. I compare the effects of protection in marine reserves for intertidal and subtidal communities. In addition, I examine the species and reserve characteristics that contribute to the variation in responses to protection. These analyses demonstrate positive outcomes of reserve protection at assemblage and species levels. However, high variation in responses to protection was observed at the level of species, and this variation is better explained by the intensity of exploitation.

Finally, in Chapter 4, I emphasize the conservation implications of my findings and discuss the need for advancing marine conservation efforts in Latin America and the Caribbean.

Both Chapters 2 and 3 are in preparation for submission for publication as multi-author papers, with my advisors as co-authors. In each case, my co-authors provided comments on the text and valuable advice.

Chapter 2:

Current status of marine protected areas in Latin America and the Caribbean

A. Paulina Guarderas, Sally D. Hacker, and Jane Lubchenco

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Abstract:

In Latin America and the Caribbean, the deterioration of coastal ecosystems and the exploitation pressure for marine resources are increasing dramatically. A system of marine protected areas, in particular, no-take reserves, is a promising management tool to ameliorate the degradation of marine ecosystems. However, little is known about the scope of marine protection efforts in the entire region. In this paper, we document the status of MPAs and no-take reserves in Latin America and the Caribbean. Our comprehensive review of protected area databases, published and unpublished literature and internet searches showed that the number of MPAs and the area protected has increased through time, in particular since the 1980s. At present, more than 800 MPAs have been established, covering more than 300,000 km^2. Of these, the majority allow extractive activities (658 sites covering 53,221 km^2). Multiple use MPAs (those zoned to include both take and no-take areas) are the fewest in number (87), but they cover the largest area (238,600 km^2). No-take marine reserves represent one-tenth the number of MPAs (87 sites, covering 14,181 km^2). Despite the recent increase in number and area of MPAs, many biogeographic provinces still have few MPAs. Large coastal regions remain unprotected, in particular the southern Pacific and southern Atlantic coasts of South America. Our synthesis reveals multiple opportunities to strengthen marine conservation in Latin America and the Caribbean by improving implementation, management, and enforcement of existing MPAs.

Introduction:

Marine systems are increasingly affected by human activities and their indirect impacts (Crowder 2005; MEA 2006). Climate change, habitat destruction, pollution, introduced species, overexploitation, and destructive impacts of some fishing gear severely threaten marine biodiversity worldwide (Jackson et al. 2001; Bellwood et al. 2004; Wilson et al. 2006; MEA 2006), all of which can have important effects on ecosystem services (Balmford & Bond 2005; MEA 2006; Worm et al. 2006).

On a regional scale, marine ecosystems of Latin America and the Caribbean are undergoing drastic transformations. Sixty percent of the human population is concentrated within 100 km of the coast (Cohen et al. 1997), leading to increased coastal development and destruction of nearshore habitats, pollution, and overexploitation of marine resources (UNEP 2001). Although the quality of data available about the status of marine species varies, there is an overall consensus that fish stocks are heavily depleted and in some areas severely overfished (Cortés 2003). More than 80 percent of the commercially exploitable stocks in the southwestern Atlantic and 40 percent in the southeastern Pacific are fully fished, over-fished, or depleted (FAO 1995). In addition, there has been a proliferation of tourist resorts, an expansion of aquaculture, and a general lack of effective coastal regulations and enforcement (UNEP 2001).

The loss of marine biodiversity and the failure of some traditional management methods such us single species aproaches (Crowder 2005) indicates the need for a more integrative, ecosystem-based approach (SCBD 2004). Marine protected areas

(MPAs) are one effective tool for conserving biodiversity. MPAs are areas of the ocean where some or all activities are limited or prohibited to protect natural and cultural resources (Araime et al. 2003; Lubchenco et al 2003). They can halt some of the key threats to marine ecosystems: overexploitation, habitat degradation, and to a lesser extent pollution and invasion of allien species (Possingham et al. 2005; Roberts 2005). However, single MPAs will rarely be of adequate size to conserve a representative sample of regional biodiversity, or provide adequate protection for species or populations with complex life histories and large area requirements, therefore entire reserve systems are critical for the conservation of biodiversity (Gerber et al 2005; Possingham et al. 2005).

Marine protected areas should include: (1) a representative network of highly protected areas, where all human extractive and destructive activities are prevented within their boundaries (marine reserves), and (2) a complementary system of interconnected areas with different degrees of protection that supports the biodiversity objectives of the highly protected network.

The highly protected areas, also known as no-take marine reserves, are an important component of biodiversity conservation (Sobel and Dahlgren 2003). By excluding extractive activities as well as other human impacts, no-take marine reserves can protect both habitat and species. The maintenance and recovery of marine ecosystem integrity, structure, and functioning provides sources of resilience to maximize persistence even if natural and human disturbances occur (Sobel & Dahlgren 2003; Hughes et al. 2005). Used in combination with other management tools, marine reserves can enhance fish stocks and catches by providing a refuge for critical life history stages and by improving the reproductive potential of depleted species (Roberts & Hawkings 2000; NRC 2001). Studies have shown that most no-take marine reserves demonstrate significant increases in size, abundance, and diversity of fishes and invertebrates inside their borders (Roberts & Hawkings 2000; Halpern 2003).

Recently, “multiple use” MPAs have been developed to incorporate a larger variety of human activities within their boundaries. These MPAs (which go by varying names) often encompass core no-take zones surrounded by areas with different extractive uses to be managed in a sustainable manner (Bishop et al. 2004).

The international community has recognized the need to establish a global representative network of MPAs, including no-take marine reserves, as a mechanism to conserve biodiversity in the oceans (IUCN 2003; CBD 2004). One of the marine conservation targets proposed by the World Wildlife Fund (WWF) states that by 2012 at least 10% of the world’s oceans should be protected within a comprehensive, adequate, and representative system of MPA networks. The World Park Congress of 2003 proposed that by the year 2012, 20-30% of marine habitats should be strictly protected as reserves (IUCN 2003). Currently less than 1% of the world’s oceans are covered by MPAs with some form of protective regulation (Chape et al. 2005, Wood unpublished), and only 0.01 % is protected from all extractive activities (Roberts & Hawkins 2000). A recent global assessment showed that 18.7 % of the world’s coral reefs lie inside MPAs, and only 1.4 % inside no-take MPAs. Unfortunately, the majority are not adequately managed (Mora et al 2006).

One of the critical problems in conservation biology for both terrestrial and marine systems is that protected areas are too small, too isolated from each other, and represent too few types of ecosystem to sustain native populations (Newmark 1987). Thus, evaluating MPA size, their connectivity patterns and the degree to which represent biodiversity will improve the design of protected areas, and therefore biodiversity conservation.

In this paper, we document the status and trends of marine reserves and other MPAs in Latin America and the Caribbean. Despite marine conservation proposals and prioritization for this region (Sullivan-Sealey and Bustamante 1999; TNC 2007), a comprehensive survey of the current marine reserves in this part of the world has been lacking. Without this information, reasonable assessments of the progress and future planning is not possible. Documentation of MPAs in this region has occurred in the past (Silva & Desilvestre 1986; Kelleher et al. 1995, Appeldoom & Lindeman 2003, Mora et al. 2006) but these studies are either outdated, qualitative, habitat specific, and/or only include the Caribbean.

In this paper, we describe the current representation of MPAs and no-take reserves in Latin America and the Caribbean by examining 1) changes in their number through time, 2) their level of protection, 3) their size and connectivity, 3) the degree to which they fit IUCN management categories (IUCN 1994), and 4) by comparing the size and level of protection of these MPAs with regard to country and major biogeographic province. Adittionally, we assess the representation of MPAs in Latin America and the Caribbean in the context of regional and global conservation initiatives for both terrestrial and marine environments.

Methods:

To document the extent and level of marine protection in Latin America and the Caribbean, we conducted an extensive database review and created a map of the existing protected areas using data from Mexico, Central and South America, and the Caribbean. For the purposes of this review, MPAs are defined as any marine intertidal and subtidal area that has been set aside primarily for biodiversity purposes and includes biological, physical, and cultural attributes (IUCN 1994). Some protected coastal areas, including so-called "fishery management areas", that do not include biodiversity conservation goals were not included in our analysis. However, it should be noted that fishery management areas accomplish complementary goals and are becoming more common; for example, they have been widely established in Chile and Brazil (Fernández & Castilla 2005; IBAMA 2007).

Our primary sources of data were two freely available databases: World Database on Protected Areas (WDPA; 2006 version) and MPA Global (Wood 2007). Additional data that were not registered in the WDPA or MPA Global databases were added by conducting literature and internet searches and by communicating directly with managers and scientists.

Using these data, we created an up-to-date, comprehensive GIS-datalayer using ArcGIS 9.0 and included information about location, year of creation, size (marine portion only), level of protection, distance to the nearest MPA, and IUCN management category. To standardize the level of protection (in view of the multiplicity of names and designations), we categorized MPAs as either “no-take”, “limited take”, or “multiple use” (Mora et al. 2006). Multiple use MPAs are those zoned with different levels of protection inside their boundaries; they encompass no-take zones as well as areas where extractive activities are allowed with some restrictions (Mora et al. 2006). “No-take” areas are synonomous with marine reserves and are used interchangeably in this paper. “Limited take” areas allow some extractive activities with restrictions. Protection level information was obtained from the sources described above as well as Appeldoorn & Lindeman (2003) and Bustamante (2006, unpublished report) for the Caribbean.

IUCN management categories are assigned to every MPA by the conservation authority of each country based on the guidelines proposed by the IUCN (1994). They encompass seven categories classified by the degree of human intervention and protection goals. They include: I strict protection (Ia Strict Nature Reserve , Ib Wilderness Area), II ecosystem conservation and protection (National Parks), III conservation of natural features (Natural Monuments), IV conservation through active management (Habitat/Species Management Area), V landscape/seascape conservation and recreation (Protected Landscape/Seascape), VI sustainable use of natural resources (Managed Resource Protected Area).

We also assessed the status of MPAs in different countries and major marine ecosystems (bioregions). We conducted gap analyses (Jenings 2000) by overlaying our MPA datalayer on a country datalayer from the the Flanders Marine Institute Geodatabase (VLIZ 2007) and on a coastal biogegraphic regions and provinces of Latin America and the Caribbean datalayer developed from Sullivan-Sealey & Bustamante (1999). The country datalayer represents the maritime boundaries of the world countries, which are delineated by the 200 nautical miles of exclusive economic zone (EEZ) of each country. Using the coastal and marine biogeographic datalayer we estimated the percent area protected either as no-take and limited take in the entire region. Size and spacing patterns of MPAs were assessed for both the Pacific and Atlantic coasts of this region of the world.

Results:

MPAs are a recent management strategy in Latin America and the Caribbean (Fig. 1). Although there was a slow but steady increase in the number and size of MPAs through time, not was until the 1980s and 1990s when both number and coverage of MPAs largely increased (Fig. 1). To date, 832 marine protected areas (MPAs) covering more than 300,000 km^2 have been established in Latin America and the Caribbean (Table 1). The current extent of all MPAs represents 1.5% of the coastal and marine systems in Latin America and the Caribbean, whereas no-take marine reserves cover only 0.1%.

Tallies of numbers versus area provide different pictures and level of protection. Limited-take MPAs are by far the most common (658 or 79% of all protected aeas in Latin America and Caribbean) protection level when compared to no-take reserves (87 or 10.5%) or multiple-use MPAs (87 or 10.5%) (Fig. 2, Table 1). In contrast, multiple use MPAs cover the largest area (238,600 km^2 or 78%) compared to those MPAs allowing extraction (“take” MPAs; 53,221 km^2 or 17%) and no-take reserves (14,181 km^2 or 5 %) (Table 1).

For MPAs with available information, sizes ranged from 0.001 to 133,000 km^2 (Fig. 2, 3). The mean (± SE) size was 2,280 ± 1144 km^2. The median value was 65 km^2, and the most frequent size ranged between 100 and 1000 km^2 (Fig. 3A). The largest MPA in Latin America and the Caribbean is the Galapagos Marine Reserve (133,000 km^2, Ecuador), followed by the SeaFlower Biosphere Reserve (60,000 km^2, Colombia) and Banco de la Plata Whale Sanctuary (25,000 km^2, Dominican Republic). Conversely, the smallest MPAs are less than 1 km^2, including Quintay (0.002 km^2, Chile) and Montemar (0.01 km^2, Chile).

No-take reserves (either singly or as part of multiple use MPAs) varied in size from 0.006 to 8,575 km^2 (Fig. 2, 3B). The size most frequently ranged from 1 to 10 km^2, and the mean and median values were 455 ± 211 km^2 and 34 km^2, respectively (Fig. 3B). Malpelo is the largest no-take reserve (8,575 km^2) followed by the no-take zones within the SeaFlower Biosphere Reserve (2,230 km^2), both from Colombia and the Galapagos Marine Reserve (1,135 km^2).

Distance between MPAs and reserves are generally quite apart. MPAs located in the Atlantic coast of Latin America and the Caribbean Sea were commonly separated from one another by tens to thousands of kilometers, whereas fewer MPAs, separated by larger distances were identified in the Pacific coast (Fig. 4A). No-take reserves were generally more than 100 km apart, and only 33% were closer than 100 km in the Atlantic and Caribbean; while those established in the Pacific coast were fewer and more than 1000 km apart (Fig. 4B). Although mean distance values from the Pacific and Atlantic MPAs were not statistically different (t-test = -9.12, df = 811, p-value = 0).

Interestingly, there was no clear relationship between IUCN numbers and levels of protection (Table 2). For example, the most common IUCN management category for no-take and multiple use MPAs was II, followed by IV (Table 2).

There was considerable variation in MPA coverage among countries (Table 1). Bermuda had the highest number of no-take reserves; however, the total area protected was small (Table 1). For the 43 countries included in this review, 18 lacked any no-take reserves (Table 1). Colombia, Belize, Cuba, Ecuador (Galapagos region), Brazil, Mexico, and the Bahamas had the greatest total area of no-take marine reserve (Table 1). The greatest total area of limited-take MPAs occurred in Ecuador, Colombia, and the Dominican Republic (Table 1).

There was variation in both the areal extent and number of MPAs across different biogeographic provinces (Fig. 2, Table 3). The majority of MPAs are located in the Caribbean, which is situated in the Tropical Northwestern Atlantic biogeographic province. The MPA with the largest single area, however, is located in the Galapagos province, although this area was not exclusively no-take. The Tropical Eastern Pacific province has the largest areal extent of no-take MPAs (Fig. 2, Table 3). Large coastal areas remain unprotected, especially the south Pacific and south Atlantic coasts of Latin America (Fig. 2, Table 3), which are known as the Warm-temperate Southwestern Pacific and Cold Temperate South America provinces. A relatively small number of no-take reserves was observed across the entire region, although there were 48 no-take reserves (55 % of total) in the Caribbean (Fig. 2, Table 3).

Discussion

Our review demonstrates that the number and the area protected in Latin America and the Caribbean have increased through time, particularly since the 1980s. The current number of MPAs (832) in Latin America and the Caribbean and their coverage (300,000 km^2) represents 20% of the total global number and 15% of the total global area for MPAs (Wood 2005 unpublished, Chape et al. 2005), which constitute an important conservation effort, taken into account that this is a developing region. Nonetheless, only 1.5 percent of the coastal and marine systems in this region of the world are under some type of protection, which is biased by the presence of few very large MPAs (Galapagos Marine Reserve, SeaFlower Biosphere Reserve and Banco de la Plata Whale Sanctuary). Moreover, conservation efforts for this region follow the same global pattern (Chape et al. 2005), where extent of protected areas on land outweighs the marine protected coverage. The World Database on Protected Areas

(2006) records more than four million km^2 of land protected in Latin America and the Caribbean which is one order of magnitude higher compared to the extent of MPAs (300,000 km^2) identified in our review.

The majority of MPAs created in Latin America and the Caribbean allow extractive activities within their boundaries (Table 1, 3). In contrast, no-take marine reserves are poorly utilized in this region, representing 0.1% of the total marine biogeographic coverage for Latin America and the Caribbean. This low representation is especially evident in the Pacific coast of Central and South America and the South Atlantic. Multiple use MPAs are becoming a common management tool in this region and encompass large areas, particularly the most recently established MPAs, the Galapagos Marine Reserve and the SeaFlower Biosphere Reserve. If multiple use MPAs include a significant fraction of no-take reserves and are well designed and enforced, they can protect ecological processes, conserve linkages between habitats, and allow multiple uses by establishing zoning to achieve a wide range of conservation objectives (Agardy 2002). The best known example of a multiple use MPA is the Great Barrier Reef Marine Park of Australia where 33% of the total area protected is in no-take reserves (Fernandes et al. 2005). However, simply zoning an area without including networks of no-take reserves is unlikely to accomplish a high level of protection.

The current system of MPAs is a promising starting point towards achieving marine conservation goals in Latin American and Caribbean countries. However, it is insufficient to fully represent the entire array of marine biogeographic provinces and ecoregions, a prerequisite for protection of biodiversity (Roberts et al. 2003). Our results show that the existing MPAs are concentrated in some biogeographic regions and largely absent from others. The south Pacific and south Atlantic coasts of Latin America are largely unprotected (Fig. 2, Table 3). In contrast, the Caribbean Sea (Tropical Northwestern Atlantic biogeographic province) was the most represented region in terms of number of MPAs, particularly no-take MPAs. However, their distribution in relation to habitat type is uneven, they are covering mainly coral reefs and associated seagrass and mangrove habitats while other important habitats are not represented (Appeldoorn & Lindenman 2003). Our review demonstrates that from the high-priority conservation regions identified by Sullivan-Sealey and Bustamante (1999) along the shores of Latin America and the Caribbean only the Central Caribbean, Cortezian, and Panama Bight ecoregions (Table 3) have increased their MPA coverage. The other areas identified by Sullivan-Sealey and Bustamante (1999) remain largely unprotected.

The uneven distribution of MPAs in Latin America and the Caribbean could be explained by differential knowledge of the status of marine systems across the entire region, different degrees of organization between countries, and special recognition of some areas as irreplaceable sites for conservation. For instance, Caribbean coral reefs are the most widely studied within the American coral reefs (Cortés 2003), and therefore their degradation has been well documented (Harvell et al. 1999; Jackson et al 2001; Gardner et al. 2003; 2005). Thus, conservation action may be the result of documented widespread degradation of marine ecosystems attributed to anthropogenic activities (Hughes et al. 2005). Additionally, many institutions working in the Caribbean scientific and conservation arena may lead to more conservation planning and implementation (Côté et al. 2005). In contrast, monitoring of marine biodiversity and conservation efforts in the rest of the region seems to be reduced or more localized (Cortés 2003). Nevertheless, the declaration of the marine conservation corridor of the Tropical Eastern Pacific is encouraging regional conservation efforts along the Pacific coasts of Latin America (UNESCO 2007). This effort may have facilitated the establishment of the largest no-take reserve in this region (Malpelo 8,575 km^2).

The international recognition of the importance of specific marine habitats and ecosystems could promote local conservation efforts and international funding resources. The creation of the Galapagos Marine Reserve (GMR), a conservation icon around the world, was highly supported by international conservation organizations in collaboration with the local community (WWF 2007). However, much remains to be done to achieve its full implementation. For instance, the no-take zones have not been implemented to date and illegal industrial and artisanal fishing still threaten marine systems of the GMR (WWF 2007).

Marine conservation initiatives should recognize the importance of connectivity between habitats and populations. Most marine organisms have complex life histories including ontogenetic shifts and migrations (Gerber et al 2005). Some have pelagic larvae that can travel distances from meters to hundreds of kilometers (Kinlan & Gaines 2003; Palumbi 2003b; Shanks et al. 2003). These distances have direct implications in the design of MPAs because they will greatly determine reserve effectiveness via connectivity and self-seeding processes (Lipcius et al. 2005). Considering these larval dispersal distances, recommendations of size and spacing of marine reserves have been proposed. Marine reserves from 10-20 km in size and spaced 20 km apart can be effectively self-seeding and allow connectivity between protected populations (Palumbi 2003b, Shanks et al 2003). Our data suggest that MPAs, particularly no-take MPAs were not design using biological principles such as connectivity, thus design issues needs to be improved to allow connectivity between populations across this region, particularly the number,size and spacing of no-take marine reserves in the Pacific coast of Latin America (Fig. 3, 4). No-take reserves were small in size (1 to 10 km^2) and frequently distant from the closest no-take reserves (Fig. 4B). It seems that efforts toward protected area establishment tended to focus on designation of single protected areas rather than entire reserve systems. Additionally, the concept of networks in marine protected areas has gained increasing recognition only after the result of the NCEAS science of marine reserves working group, initiated in 2000.

Moreover, our review revealed a deficiency in the application of the IUCN management category for MPAs. The IUCN management categories assigned to MPAs by the conservation authority of each country did not accurately reflect the level of protection (Table 2). In the context of MPAs, it is generally considered that only IUCN category Ia (Strict Nature Reserve) is a no-take marine reserve, where the management objectives include preserving biodiversity (Bishop et al. 2004). However, we observed that almost all IUCN categories were assigned for no-take marine reserves. For limited-take or multiple use MPAs, strict IUCN management categories (Ia and Ib) were also identified in our review, which are not consistent with the objectives of these designations. No-take zones within multiple use MPAs were not recognized with an IUCN category. Clarification by the World Commission on Protected Areas-Marine (WCPA) is needed to improve the use of IUCN management categories under the IUCN (1994) guidelines within each country (Bishop et al. 2004).

Although the number and extent of protected areas are indicators of the commitment of biodiversity conservation in Latin America and the Caribbean, they do not assess management effectiveness (Chape et al 2005). Management capacity, enforcement, and monitoring of many MPAs are limited by funding, expertise, and training (Silva & Desilvestre 1986). Appeldoorn and Lindeman (2003) suggest that effective compliance with no-take marine reserves in the Caribbean is a significant problem that needs to be addressed, with the highest level of compliance being reported for only 16% of the reserves. At a global scale, MPA management effectivenes is generally low (Kelleher et al 1995). Mora et al (2006) found that only 1.6 % of the world’s coral reefs are adequately managed, and from this no more than 0.1 % are within no-take reserves that are well enforced. McClanahan (1999) suggests that the recent proliferation of MPAs in developing countries does not necessarily result in conservation of marine resources, and that it is more important to improve management inside the existing MPAs and outside MPA borders than to designate more MPAs. In contrast, Roberts (2005) suggests that the present coverage, in particular no-take marine reserves, will do little to stop biodiversity loss in the sea, even if they are well managed.

Our review demonstrates that the number and extent of MPAs, including no-take marine reserves, has increased, but the existing protection is currently insufficient to represent marine biodiversity at a biogeographic scale, and to allow connectivity between populations that are protected. In addition, management effectiveness tends to be low for the majority of MPAs, then improvements in the implementation and management of existing MPAs is essential if marine conservation goals in Latin America and the Caribbean are to be fulfilled. A large-scale regional approach to planning networks of MPAs may be more effective for conserving marine biodiversity than piece-meal efforts (Agardy 2002). It is far easier to protect biodiversity and ecosystem functioning than it is to restore once it is compromised (Hughes et al. 2005).

This review provides a basis for which future comparisons of marine conservation in this region of the world can be compared. However, fine scale analysis (100 km or less) that represent habitat between and within biogeographic provinces, and that match the scale of community organization and national initiatives are also needed to provide a better appreciation of the marine conservation initiatives in this region (Kappel 2006).

Acknowledgements

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Table 1. The number, size, and percentage protected for limited take and no-take marine protected areas in different countries in Latin America and the Caribbean. Limited take refers to those MPAs where some limitations on fishing activities take place, no-take are MPAs that do not allow extractive activities inside their boundaries, and multiple use are those MPAs zoned with different levels of protection, they encompass no-take zones as well as areas where extractive activities are allowed with some restrictions. The percent area protected in relation to each country is delineated by the 200 nautical miles of exclusive economic zone of each country.

Country	Number of MPAs			MPA Coverage (km^2)				% Area of country protected	
	No-take	Limited Take	Multiple use	No-take	Limited Take	Multiple use Total	Multiple use No-take	No-take	Take
Anguila		5							
Antigua and Barbuda		8	3			20			0.02
Argentina		40			3848				0.35
Aruba		1							
Bahamas	5	30		408	83			0.058	0.03
Barbados	1			2				0.001	
Belize	10	5	6	880	525	1185	92	2.698	4.25
Bermuda	29	73		14		982			
Brazil	5	66	2	1465	4985	5117		0.046	0.32
Cayman Islands	10	36	4	2	147			0.002	
Chile	1	17	1	0.14	147	672	70	0.002	0.02
Colombia	2	8	3	8584	752	66104	2330	1.362	8.05
Costa Rica	2	32	2	970	1912	682			0.6
Cuba	5	17	3	1836	983	2045	1400	0.88	0.44
Dominica	1	2			3				0.01
Dominican Republic		13	2		25000				9.28
Ecuador		8	1		933	133000	1135	0.001	12
El Salvador		5			10				
French Guyana	1	4							
Grenada		1							
Guadeloupe		11	3			47			0.03
Guatemala		6							
Honduras		19	1						
Jamaica	3	9	2	3	1356	175		0.001	0.62
Martinique		17							
Mexico	1	19	13		9712	22838	688	0.021	0.97
Montserrat		6							
Netherland Antillas		7	3		23	40	32	0.046	0.04
Nicaragua		5							
Panama		21			2568				0.77
Perú		3							
Puerto Rico	1	28	3					0.008	0.03
Saint Kitts & Newis		1							
Saint Lucia	2	27	1						
Saint Vincent & Grenadines	1	18							
Suriname		7							
Trinidad & Tobago	1	12						0.007	
Turks & Caicos	3	11	15	17			4		
Uruguay		8							
Venezuela	1	11	12		216	5693	205	0.043	1.15
Virgin Islands (British)		31	2						
Virgin Islands (U.S.A.)	2	10	5		17		75		
Total	87	658	87	14181	53220	238600	6031		

Table 2. Distribution of marine protected areas (MPAs) by IUCN management category and level of protection (no-take, limited take, and multiple use) in Latin America and the Caribbean. IUCN management categories include: I strict protection (Ia, Ib), II ecosystem conservation and protection, III conservation of natural features, IV conservation through active management, V landscape/seascape conservation and recreation, VI sustainable use of natural resources.

Type of MPA	IUCN Management Category									
	Ia	Ib	II	III	IV	V	VI	Unset	Unknown	Total
No-take	2		27	2	21	3		7	25	87
Limited Take	39	9	123	24	249	46	47	104	18	659
Multiple use			34	5	24	3	10	11		87
Total	41	9	182	31	294	52	57	122	43	831

Table 3. The number, size, and percentage protected for limited take and no-take marine protected areas in different biogeographic provinces and ecoregions in Latin America and the Caribbean. Biogeographic provinces are classified based on Sullivan-Sealey & Bustamante (1999). Limited take refers to those MPAs where some limitations on fishing activities take place, no-take are MPAs that do not allow extractive activities inside their boundaries, and multiple use are those MPAs zoned with different levels of protection, they encompass no-take zones as well as areas where extractive activities are allowed with some restrictions.

Biogeographic Province and Ecoregion	**Number of MPAs**			**MPA Coverage (km^2)**				**% of Province/Ecoregion**	
	No-take	**Limited take**	**Multiple use**	**No-take**	**Limited take**	**Multiple use Take**	**Multiple use No-take**	**No-take**	**Limited Take**
Warm Temperate Northeastern Pacific									
Cortezian	1	3	4		5454	5308	688	0.25	3.64
Mexican Temperate Pacific		1	1			3171			0.6
Magdalena Transition									
Total Province	**1**	**4**	**5**		**5454**	**8479**	**688**	**0.07**	**1.58**
Tropical Eastern Pacific									
Chiapas Nicaragua		25			1082				0.27
Clipperton and Revillagigedo			1			6209			1.18
Cocos Island	1			970					0.32
Guayaquil		8			933				<0.01
Mexican Tropical Pacific		5			47				0.01
Nicoya	1	28			3030			<0.01	0.92
Panama Bight	1	15		8575	718			1.69	0.22
Total Province	**3**	**81**	**1**	**9545**	**5810**	**6209**		**0.27**	**0.38**
Galapagos*									
Northern							14	0.01	19.57
Western							340	0.14	18.4
Eastern							781	0.19	10.86
Total Province			**1**			**133000**	**1135**	**0.14**	**17.22**
Warm Temperate Southeastern Pacific									
Central Per□									
Araucanian	1	4		0.14	42			<0.01	0.01
Central Chile		5			65				0.02
Humboldtian		4			3				<0.01
Total Province	**1**	**13**		**0.14**	**110**			**<0.01**	**<0.01**
Cold Temperate South America									
Channels and Fiords of Southern Chile		3	1			672	70	<0.01	0.07
Chiloense		3			36				0.01
North Patagonian Gulfs		10			2583				1.3
Patagonian Shelf		10			200				0.05
Malvinas-Falklands									
Total Province		**26**	**1**		**2819**	**672**	**70**	**<0.01**	**0.15**
Tropical Nothwestern Atlantic									
Bahamian	8	43	16	426	25083		4	0.04	3.05
Central Caribbean	33	170	42	2730	4337	79144	4027	0.25	3
Guaianan	1	11							
Gulf of Mexico		6	2		3537	4918			0.71
Lesser Antillas	6	136	17	2	21	80	107	0.01	0.01
Total Province	**48**	**366**	**77**	**3158**	**32978**	**84142**	**4138**	**0.12**	**2.03**
Tropical Southwestern Atlantic									
Amazonia		3							
Eastern Brazil	3	5	1	992	3150	982		0.2	0.83
Northeastern Brazil	1	25	1	423	95.8	4134			0.4
Sao Pedro and Sao Pablo Islands									
Total Province	**4**	**33**	**2**	**1414**	**3245.8**	**5116**		**0.03**	**0.32**
Warm Temperate South Western Atlantic									
R'o de la Plata		11							
R'o Grande		2							
South East Brazil	1	31		50	1739			0.01	0.46
Uruguay Buenos Aires Shelf		11			1065				0.28
Total Province	**1**	**55**		**50**	**2804**			**0.02**	**0.23**
Juan Fern‡ndez and Desventuradas									
Bermuda	**29**	**73**		**14**		**982**			
Undertermined	2	4							
Total	87	658	87	14181	53221	238600	6031		

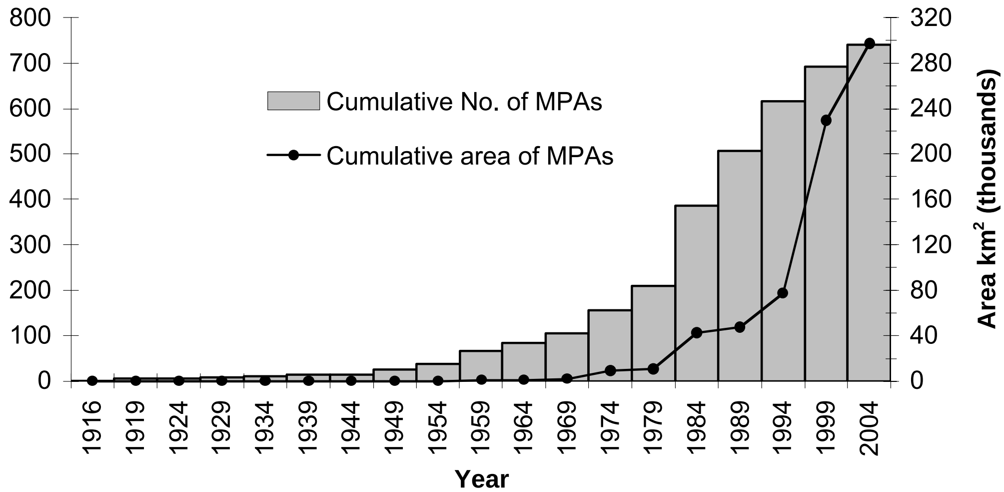


Figure 1. The number and size of marine protected areas (MPAs) established over time in Latin America and the Caribbean.

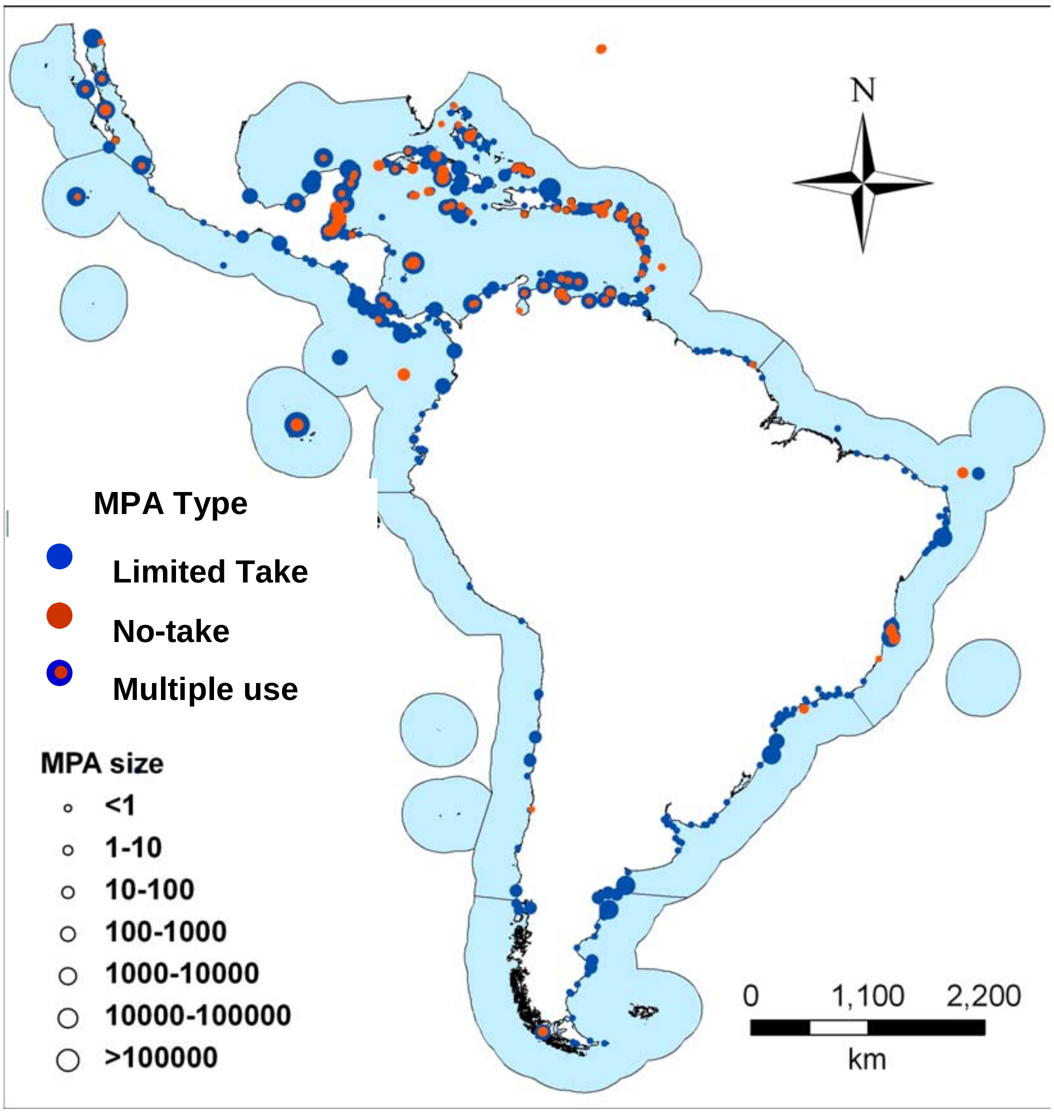
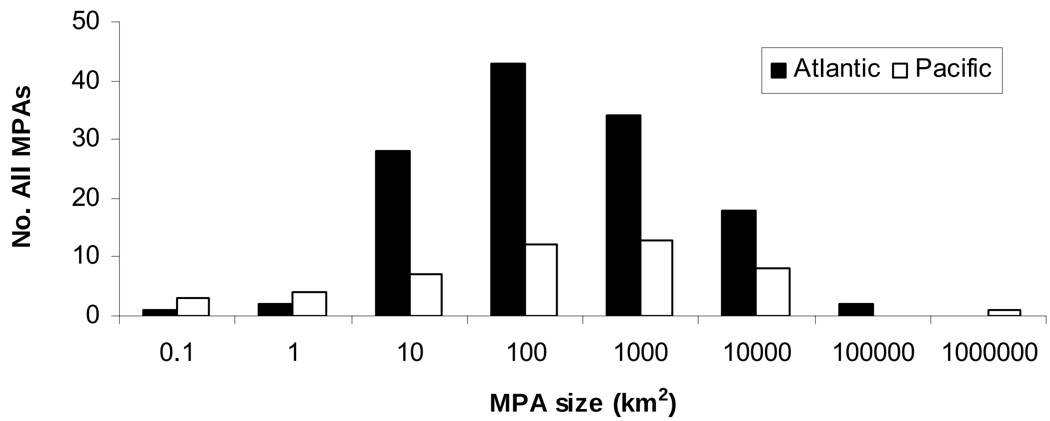


Figure 2. Distribution of marine protected areas (MPAs) in Latin America and the Caribbean, classified according to levels of protection from extractive activities.

A



B

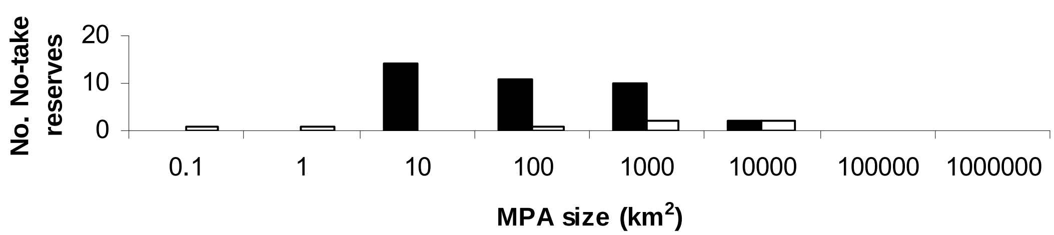


Figure 3. Frequency of different sizes of a) all marine protected areas (MPAs), and b) no-take marine reserves established in Latin America and the Caribbean. Dark bars represent MPAs located in the Atlantic and white bars represent MPAs located in the Pacific coast.

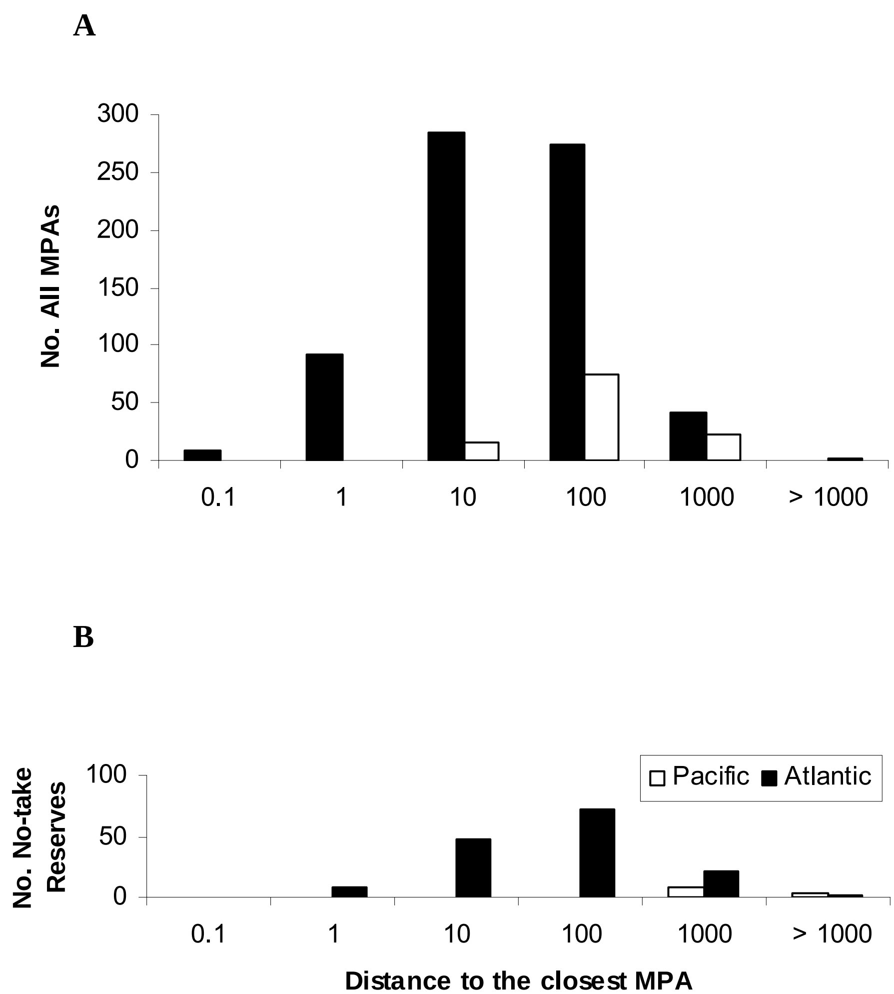


Figure 4. Frequency of distance to the nearest a) marine protected area (MPA), and b) no-take marine reserve established in Latin America and the Caribbean. Dark bars represent MPAs located in the Atlantic and white bars represent MPAs located in the Pacific coast.

Chapter 3:

A meta-analysis of the ecological effects of marine reserves in Latin America and the Caribbean

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Target Journal: *Ecological Applications*

ESA Publications, Ithaca, NY, USA

Abstract:

Marine reserves constitute a powerful conservation tool for mitigating ocean degradation, particularly the harmful effects of overfishing. Given the potential benefits of marine reserves for both fisheries and conservation, it is crucial to understand the patterns of change following the establishment of a reserve. In this paper we use meta-analytical methods to synthesize the performance of marine reserves established in Latin America and the Caribbean. Our analysis shows overall positive outcomes of reserve protection at assemblage and species levels. Biomass and density nearly doubled inside marine reserves whereas body size and species richness increased only slightly and did not significantly differ. We demonstrated that marine reserves are an effective conservation tool to rebuild exploited populations, especially invertebrate species. When density was analyzed, the only consistent pattern across reserves was a positive correlation between species responses to protection and their individual level of extraction outside reserve boundaries. In contrast to other findings, our analysis did not show a clear relationship between responses to protection and species-specific characteristics. However, when biomass was analyzed across trophic groups, the effect of protection on predator groups was more evident, and in some cases indirect effects were disclosed. Our review also revealed that the number of publications of marine reserve performance in Latin America and the Caribbean is still limited. There is a need to increase the number of studies that document the ecological effects of marine reserves and make such findings available in peer-reviewed journals.

Introduction:

Anthropogenic activities are increasingly affecting the world's oceans (Crowder 2005, Harley et al. 2006). Overfishing is one of the major threats to marine systems (Jackson et al. 2001, Breitburg and Riedel 2005, Kappel 2005), and direct or indirect consequences of fishing include population decline of numerous marine organisms (Myers and Worm 2003), degradation of marine habitats (MEA 2006), and modification of food webs (Pauly et al. 1998, 2001).

Marine reserves, which are areas set aside from extractive activities, constitute a powerful conservation tool for mitigating ocean degradation (Lubchenco et al. 2003, Worm et al. 2006). They are particularly beneficial for counteracting the harmful effects of overfishing by providing spatial refuges for fished populations as well as protecting important habitats and their associated ecological interactions (NRC 2001, Sobel and Dahlgren 2003, Sladek Nowlis and Friedlander 2005). Recent syntheses of data from many reserve sites have shown that overall species richness, density, biomass, and size of marine organisms are consistently higher inside marine reserves than in nearby fished sites (Côté et al. 2001, Halpern 2003, Palumbi 2003a).

Positive effects after reserve establishment have been described in a variety of marine communities including kelp and seagrass beds and rocky and coral reefs located at different latitudes around the world (Durán and Castilla 1989, Polunin and Roberts 1993, McClanahan and Kaunda-Arara 1996, Stoner and Ray 1996, Russ and Alcala 1999, Roberts and Hawkins 2000, Gell and Roberts 2003b). In contrast to what was previously thought (Blyth-Skyrme et al. 2005, Laurel and Bradbury 2006), marine reserves established in temperate latitudes appear to produce equal if not greater overall benefits on biomass and density than tropical marine reserves (Lester et al. in review). However the overall importance of community type in explaining variation in reserve response is largely unknown.

In addition, it is likely that not all species in a reserve will be equally affected by protection. Species may respond differently depending on whether they are targeted for fisheries outside the reserve and their history of exploitation prior to reserve establishment (Côté et al. 2001). For instance, because fishing mainly targets species from higher trophic levels (Pauly et al. 1998), these large predators are likely to benefit from protection but their recovery rates may be slow (Jennings et al. 1999).

Responses to protection may also be influenced by life history characteristics of individual species (Jennings et al. 1999, 2001, Gerber et al. 2003, 2005). For example, short-lived, fast-growing, and prolifically recruiting species may respond rapidly to protection while large and slow growing species may gradually increase in population size over time (Roberts 2005). Dispersal ability is another characteristic that may differentially affect the response of individual species to reserve protection. Studies have shown that reserves work most effectively for sedentary species (Jennings 2001), compared to highly mobile species, which need a reserve that encompasses their entire range (Sladek Nowlis and Bollerman 2002, Gerber et al. 2003). Although some highly mobile species can show positive responses to protection if a proportion of the population show high levels of site fidelity (Gell and Roberts 2003a)

In addition to the direct effects of reserves, protection can have an influence on the community through indirect means (Pace et al. 1999, Pinnegar et al. 2000). Change in community composition can occur by predator interactions cascading down food webs after long periods of protection (McClanahan and Graham 2005, Mumby et al. 2006). For example, benthic communities in the Leigh Marine Reserve (New Zealand) shifted from being dominated by sea urchins to being dominated by macroalgae; this was a result of a trophic cascade thought to be an indirect effect of increased predator abundance (Shears and Babcock 2003).

Given the potential benefits of marine reserves for both fisheries and conservation, it is crucial to understand the patterns of change following the establishment of a reserve. In this paper we use meta-analysis to synthesize the performance of marine reserves at a regional scale. We focus our study on Latin America and the Caribbean where a synthesis of the impact of marine reserves has not been assessed. This is true despite the widespread degradation of many coastal ecosystems in the region (Cortés 2003). Additionally, Latin American and Caribbean countries experience similar levels of development, socio-economic structure and sources of production, making them a natural group to consider as one region (UNEP 2001).

We ask the following questions in our study: (1) Do marine reserves in this region result in increased critical population rates (biomass and density), growth rates (individual size) and community rates (species richness) inside their boundaries? Is the effect of protection in this region similar to the overall effects reported in other reviews? (2) Are there different responses across habitats (subtidal vs. intertidal)? (3) Are there differences in responses among species and how do they vary between fish versus invertebrate species? (3) What reserve and species characteristics play important roles in determining the magnitude of responses?

Methods:

A literature survey was performed to collect studies on the ecological effects of marine reserves in Latin America and the Caribbean. The keywords used in the search were: marine protected areas, marine reserves, and marine parks. We searched four databases: the Aquatic Sciences and Fisheries Abstracts (ASFA), Oceanic Abstracts, Environmental Sciences, and the Zoological Record Plus Database from 1970 to 2007. References cited in these publications were also evaluated for use in the meta-analysis. Peer-reviewed studies and gray literature that quantitatively evaluated biological effects of total protection from extractive activities were included. We focused on marine reserves in Latin American and Caribbean countries that (1) were established at least one year before their effects were evaluated, (2) were reasonably well enforced, and (3) compared the effects of marine reserves to control sites (fished areas) at one or multiple points in time [i.e. Before After Control Impact (BACI) designs].

When data were provided for a number of sites within and outside each reserve, they were converted to a single mean value for the reserve and control areas (Côté et al. 2001, Halpern 2003). In addition, if different studies evaluated the performance of the same reserve, averages were taken across studies to obtain an overall reserve response variable. We chose an average value because different species or assemblages were generally evaluated in different studies. However, when data for the same species were evaluated in different publications from the same reserve, but at distinct points in time, only the latest publication date was selected. If studies with BACI designs evaluated the performance of marine reserves at distinct points in time, the most recent data were compared with the baseline in order to include the longest duration of protection.

In order to quantify the effect of protection inside marine reserves compared to fished areas we calculated response ratios (RR) (Hedges et al. 1999). In this review, RR was defined as the ratio of the biological variable of interest inside the reserve to outside the reserve (Mosquera et al. 2000, Côté et al. 2001, Halpern 2003). In addition, when studies included a BACI design, the biological variable quantified inside the reserve after its establishment was divided by the biological variable obtained before the reserve establishment. This ratio was then compared to temporal changes outside the reserve ($After_{Inside}/Before_{Inside}$) / ($After_{Outside}/Before_{Outside}$). In addition to the response variables, independent variables were recorded that included total reserve area, years of protection, number of species surveyed, census area, habitat type (subtidal or intertidal) and biogeographic region (Sullivan Sealey and Bustamante 1999).

Some studies detailed the effects of protection at the species level, while others reported the effects of functional groups, taxonomic categories, or overall effects of protection of all the species studied inside a marine reserve. Thus, our research questions addressed three levels: (1) overall reserve effects, (2) trophic effects, and (3) species effects.

Data from 21 reserves and 36 publications were identified and used in the analyses (Table 4). To evaluate the overall reserve effect we performed separate meta-analysis on population biomass (n = 11 reserves), organism size (n = 11 reserves), species richness (n = 5 reserves), and density (n = 19 reserves). Because density was the most frequent response variable recorded in the literature, it was used for the majority of the analyses. For statistical analysis, the natural logarithm of the response ratio was estimated. Confidence intervals around the mean were generated by bootstrapping (1000 iterations with replacement were conducted) and were corrected for bias in distribution around the observed mean. Mean response ratios were considered to be statistically significant if their confidence intervals did not overlap zero. Analyses were conducted using the software package MetaWin (Rosenberg et al. 2000)

To understand if any reserve characteristic or methodological factors influence the responses to protection, a multiple regression analysis was conducted. The reserve response ratios were included as the dependent variable and the independent variables were reserve size, years of protection, number of species sampled, and area surveyed. We also compared the effects of protection between intertidal and subtidal reserves.

For trophic level analysis, abundance estimates were included either as biomass, density or percent cover. We focused our review on coral reef studies because they were most abundant. Data from 8 reserves were included (Table 4). Effects of protection were explored for four trophic groups including corals, macroalgae, herbivores and predators.

For individual species analyses, we collected data from 19 reserves including 166 species (111 fish species and 55 invertebrate species). To test if species responses to protection differ between invertebrates and fishes, we conducted a categorical meta-analysis. We first analyzed the following potential sources of non-independence (Gurevitch and Hedges 1999, Mosquera et al. 2000): (1) the effect of species richness within a reserve on the response, and (2) the effect of more than one estimate for a particular species within a reserve. To do this we conducted a meta-analysis with partial datasets and compared them with the outcome from the whole dataset. Two partial datasets were used by: (1) omitting the study that included the largest number of species (Sáenz and Torre 2005) and (2) randomly selecting only one density estimate per species.

Using the information provided in Froese's and Pauly's (2007) database "FishBase" and in regional specific literature for marine invertebrates (Humann and DeLoach 2002, Gotshall 2005, Briones-Fourzán et al. 2006), we recorded the trophic level of individual species. This is a unitless and continues variable that expresses where fish and other organisms tend to operate in their respective food webs. Herbivores may have values between 2-2.19, secondary and tertiary consumers, which consume mainly animals, may have trophic levels equal to or greater than 2.8; and omnivores may have trophic levels between 2.2 and 2.79 (Froese and Pauly 2007).

Other species charactaristics were also recorded: exploitation intensity (1-4), ranging in a gradient from no exploitation (1), minor commercial (2), commercial (3), to high commercial exploitation (4); maximum body size (7-300 cm); taxonomic group (invertebrate or fish); adult mobility (1-4), varying gradually from highly mobile to less mobile species as follow: oceanodromous (1), reef associated (2), benthic (3), and sessile (4); and resilience (1-4) ranging from very high (1), medium (2), low (3), and, very low (4). The resilience variable was only recorded for fish species and it is the minimum time required for doubling of the population size. Very low resilience equates to a minimum population doubling of more than 14 years, low resilience to 4.5-14 years, medium to 1.4-4.4 years, and high resilience to less than 15 months (Froese and Pauly 2007). We also recorded the intensity of exploitation and years of protection for each species in each reserve.

To explore whether species-specific characteristics affect responses to protection, we tested for the correlation between individual species response ratios and the independent variables of body size, mobility, trophic level, population resilience, and intensity of exploitation in each reserve. These correlations are based only on data from reserves that evaluated the effects of protection of at least three species in order to have sufficient degrees of freedom (Harpole and Tilman 2006). Moreover, we included studies that evaluated the responses to protection from different trophic levels to test if this variable is important in explaining responses to protection. Data from 7 reserves (162 species in total) fulfilled our criteria. For statistical purposes, all data were natural log transformed.

Additionally, to test whether the magnitude of the correlation between the species responses to protection and all the independent variables vary through time or change when the reserve size is accounted, we conducted a MANOVA. The statistics package, S-Plus 9.0, was used in all the analyses. The average of the correlation coeficients and their bootstrapped confidence intervals were calculated across reserves using Metawin v.2 (Rosenberg et al. 2000).

Results:

The overall effect of protection was higher inside reserves than outside reserves (Fig. 5). Biomass and density doubled and these differences were significantly different from zero. Body size and species richness increased slightly but were not significantly different from zero (Fig. 5). The response to protection was not affected by reserve size or years of protection (Fig. 6A, B), but it was associated with the number of species sampled in each study and the area surveyed (Fig. 6C, D). When more species were included in the study, the response ratio significantly declined (y = 0.25 -0.14 x; r^2 = 0.38; p = 0.006; Fig. 6C). In addition, the larger the area surveyed, the higher the response ratios (y = 0.25 + 0.08 x; r^2 = 0.38; p = 0.014; Fig. 6D), however, this relationship appeared to be driven by one study (Stoner and Ray 1996); when it was removed, the relationship was not statistically significant. Intertidal reserves showed greater responses (mean = 0.91, CI = 0.39-1.45, median = 1.03) than subtidal reserves (mean = 0.52, CI = 0.33-0.70, median = 0.62), but they were not statistically different.

Marine reserves established in coral reef habitats showed overall positive responses for predators and herbivores (Fig. 7), with the exception of two outliers, Hol Chan marine reserve and Punta Francés, which exhibited extremely low values for both trophic groups (Fig. 7). In contrast, the effect of protection on corals and macroalgae was less evident. It is important to highlight that a decrease in macroalgal cover inside marine reserves is a desirable conservation outcome, because such reduction would decrease competition, allowing survivorship and recruitment of corals (Belliveau and Paul 2002). A strong effect in controlling macroalgae cover was only evident in the Exuma Cays Land and Sea (Fig. 7).

At the species level, density increased inside reserves compared to outside reserves (mean = 0.24; CI = 0.16-0.31) and this was significantly different from zero (Fig. 8). Although species responses to protection varied (Fig. 8), the majority of responses were positive (66%) and a small number were strongly positive (14 %) (defined as an increase of density inside the reserve of 100 % or more; Edgar and Barrett 1997, Willis et al. 2003). To check for the non-independence of multiple response ratios for one species (from different studies), one response ratio per species was randomly selected and the data were re-analyzed. The response ratio decreased slightly (mean =0.20, CI = 0.13-0.30, median = 0.19) compared to the whole dataset but did not significantly differ. In the same way, when the study that included the largest number of species surveyed (Sáenz and Torre 2005) was excluded from the analysis, the response ratio (mean = 0.30 CI = 0.21-0.41, median = 0.11) did not significantly differ. The response ratios were also partitioned into fish versus invertebrate responses. The density of fish was significantly higher inside than outside marine reserves (Fig. 8), and the magnitude of this effect, although not significantly different from zero, was greater than the value for invertebrate species (Fig. 8), which also was not different from zero.

The correlation between species responses to protection and the species-specific characteristics exhibited high variation across reserves (Fig. 9) and the overall mean for each variable was not statistically different from zero (Fig. 9). Only the correlation between species responses to protection and exploitation intensity was significant and positive across reserves (Fig. 9). Correlations of response ratios and species-specific characteristics showed no significant relationship with years of protection or reserve size (Fig. 9). Some linear trends were apparent when reserve size was accounted, however, they were not statistically significant (MANOVA based on Wilks Lambda $F_{4,3} = 19.01$, $p = 0.53$).

Discussion:

We found overall positive responses of marine reserves at the reserve, trophic, and species level. Demographic rates (biomass and density) were greater inside marine reserves than in fished areas (Fig. 5). Although biological (body size) and community rates (species richness) had positive values, they were not statistically significant increases (Fig. 5). Our results corroborate the findings of other studies but the magnitude of the responses are much greater than those presented here (Halpern 2003, Lester et al. in review). We suggest that the lack of statistical significance in some responses may be due to a small number of studies available for this region of the world (n= 11 body size, and n = 5 species richness). Additionally, the differences in magnitude of the effects of protection may be related to latitudinal trends. A recent global review of the impacts of marine reserves demonstrated that temperate reserves show equal and in some cases greater effects compared to tropical reserves (Lester et al. in review). Because we found more studies that evaluated the performance of marine reserves in tropical areas (mostly the Caribbean), the estimation of the overall effects from protection was highly influenced by the performance of those reserves.

Intertidal marine reserves tended to provide greater responses than subtidal ones, although a low sample size may have led to a lack of significance. It makes sense that intertidal reserves will respond greater and faster than subtidal reserves because of their (1) high levels of human extraction (Moreno 2001, Thompson et al. 2002), and (2) distinct assemblages of species, which generally consist of invertebrate and macroalgal species that tend to recover faster after disturbance events (Hawkins 1999). However, the difference in response observed for intertidal reserves could also be explained by an interaction between habitat type and latitude (i.e., a majority of the intertidal reserves are in temperate regions).

We found a strong relationship between the magnitude of responses to protection and intensity of exploitation, which supports previous findings (Mosquera et al. 2000, Côté et al. 2001, Micheli et al. 2004). The only consistent pattern across reserves was a positive correlation between species responses to protection and intensity of exploitation (Fig. 9, 10). Marine reserves are an effective conservation tool to rebuild exploited populations, particularly for species such as invertebrates with high reproductive rates and short lifespans that can respond rapidly to protection. However, invertebrates also showed high variation in responses to protection (Fig. 8). Some of the highest positive responses shown in our review are from a spiny lobster (Acosta and Robertson 2003), the queen conch (Stoner and Ray 1996), a carnivorous gastropod, and grunts (McClanahan et al. 2001).

In contrast, fish species that are highly targeted by fisheries did not exhibit extremely positive outcomes from reserve establishment (Fig. 8). There are some potential explanations for this result. First, because the biological variable that we examined at the species level of analysis was density, studies that recorded the effect of protection only as biomass were not included. Second, the recovery of severely depleted species can be very slow (Roberts 1995). Highly targeted fishes may have life history characteristics (e.g., large size at sexual maturity, long lifespan, low fecundity or recruitment success) that make them both vulnerable to overfishing (Roberts and Polunin 1993) and slow to recover following protection. Thus, fishing pressure in this region might have reduced these groups of species to very low densities or extirpated them from the system (Roberts 2005). However, when studies reported the effect of protection as trophic group averages (primarily biomass estimates), the magnitude of recovery was more evident for predators than for other trophic groups (Fig. 7). This result suggests that biomass is a better indicator of reserve benefits than density. Even slight increases in density combined with increases in body size can result in strong biomass effects, which are relevant for conservation and fisheries goals.

Dramatic increases in biomass of top predators has been recorded inside two reserves with two decades of protection, the Exuma Cays Land and Sea Park (ECLSP, Bahamas) and Jardines de la Reina (Cuba) (Mumby et al. 2006, Newman et al. 2006), suggesting that with longer protection the effects could be profound at the community level. Longer periods of protection inside marine reserves not only allows for the recovery of large predators (Russ and Alcala 2004, Russ et al. 2005, McClanahan et al. 2007), but also restoration of important ecological interactions such as herbivory (Mumby et al 2006). Densities of large parrotfish species increased in the ECLSP (possibly as a result of size refuge from predation), which doubled the overall grazing effect on macroalgae inside the reserve and led to a fourfold reduction in macroalgal cover (Mumby et al. 2006, Fig. 7). This control on macroalgae has profound effects in maintaining vulnerable coral reef systems such as those in the Caribbean (Hughes et al. 2006).

In contrast to other findings (Mosquera et al. 2000, Micheli et al. 2004), species level analysis did not reveal a clear relationship between responses to protection and species-specific characteristics. It seems that species from different trophic levels, with different adult dispersal abilities, body sizes and population resilience benefit from protection, and the direction and magnitude of these relationships could be driven by the size of the reserve or habitat type (Fig. 10), which in our dataset were highly confounded.

The species characteristics that we examined tend to be correlated, for example large bodied species are highly mobile as adults, have generally high trophic levels, and low resilience. Thus, an effect of fishing pressure acting at different trophic levels simultaneously can obscure the pattern of other species characteristics. In contrast to other marine environments where overfishing operates gradually from high to low trophic levels (Pauly et al. 1998), in the Caribbean, over-harvesting levels among distinct trophic groups appear to be similar. This pattern is attributed to the widespread use of non-selective fishing gear and the high density of fishermen in small areas (Jackson et al 2001). Similar levels of extraction of organisms from different trophic groups have also been recorded in Chilean rocky shores (Castilla 1999, Moreno 2001).

We found negative responses within reserves that included organisms with low mobility and low trophic level status. Micheli et al (2004) suggested that the co-occurrence of negative responses with such characteristics might be evidence for indirect effects. The lowest species response to protection in our dataset comes from a mussel (*Perumytilus purpuratus*). It is the main prey of a carnivore gastropod (*Concholepas concholepas*) that increased dramatically inside intertidal marine reserves in Chile. The decrease of mussel abundance due to predation led the increase of barnacles and algae that now are the main space occupiers of this rocky intertidal reserve, producing a well-documented trophic cascade in this system (Castilla and Durán 1986; Durán and Castilla 1989, Castilla 1999). The role of humans as predators can be evaluated in areas where fishing activities are prevented, in this context marine reserves have been perceived as large-scale exclusion experiments where predation effects from humans can be evaluated (Moreno 2001).

Our analysis suggests that the magnitude of responses to protection might vary differentially depending on the number of species studied (Fig. 6). By sampling only one or few species, it is more likely that extreme outcomes (either highly positive or highly negative) will occur. In contrast, when many species are sampled, some strong responses may be obscured when averaged. Moreover, most studies that evaluate single species effects have focused their efforts on organisms that are likely to show positive outcomes from protection. This approach is valid in terms of monitoring specific goals of reserves such as rebuilding populations of particular species severely depleted by fishing. However, it may over or underestimate the effect of protection on the entire community. Thus, defining clear goals for marine reserves is important not only for monitoring efforts but the ecological scale from which expected outcomes may be inferred. Additionally, area surveyed may have a strong effect on the magnitude of responses to protection. However, this pattern is not clear because a significant relationship was driven by only one case, which used a different sampling methodology than the majority of studies included in our review (Stoner and Ray 1996).

We suggest that to fully understand the impact of marine reserves, monitoring and/or data analyses should assess the effects of protection on (1) target species, and (2) on different trophic and taxonomic groups that better represent community level impacts of protection. Additionally, long-term monitoring programs, using BACI designs are needed. They will not only control for the variation in space and time in order to evaluate the impacts of protection inside reserve boundaries, but will also determine (indirectly) the extent of spillover and propagule dispersal from marine reserves (Castilla et al. in press). If demographic rates (density and biomass) increase inside marine reserves, at some point, some individuals could pass the reserve border and become part of the community that is affected by fishing. If this is the case, comparing biological variables inside versus outside reserve (at one point in time) will produce confounded results of protection, driven by spillover effects. For example, a zero response ratio could obscure the buildup of organisms occurring inside reserve if individuals are moving to fishing grounds. Other ways to detect this effect should be to monitor biological variables in a gradient from the reserve boundary (Chapman and Kramer 1999), or measure the fishing effort in the adjacent fishing area (Roberts et al. 2001). Intensity of exploitation is also a critical factor that should be accounted for in reserve evaluation processes, because it will greatly determine the perceived effects of protection (Mosquera et al. 2000, Halpern 2003).

Our ability to evaluate the importance of habitat type (subtidal versus intertidal) was limited. The studies were biased toward subtidal reserves, primarily from coral reef habitats. In addition, intertidal reserves were largely concentrated in one temperate biogeographic region (Chilean rocky shores), limiting our power to contrast with tropical rocky intertidal habitats. Substantial bias toward hard substrate marine reserves has also been identified here and in other reviews (Lester et al. submitted). The studies were also restricted to fish communities; responses from marine invertebrates were either single assessments of highly commercial species or assemblages of few marine reserves. We are intrigued by the interaction effect between species-specific characteristics, reserve size and habitat type in order to explain species responses to protection. Reserve size and habitat type were highly confounded in our dataset, but our small dataset prevented us to resolve this issue. Thus, further exploration of these associations needs to be performed.

Finally, our results revealed that the number of publications of marine reserve performance in Latin America and the Caribbean is still limited and the distribution is geographically uneven. The majority of marine reserves and studies are located in the Caribbean (Table 4, Guarderas et al. in preparation). It is unfortunate to realize that quantitative information exists for only 21 of the more than 100 no-take reserves or no-take zones within multiple use MPAs created in this region of the world (Guarderas et al. in preparation). Therefore, we highlight here the need to increase the number of studies that document the community level effects of marine reserves, and making such findings available in peer-reviewed journals.

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Table 4. Reserve characteristics included in meta-analyses. Biogeographic provinces were categorized after Sullivan-Sealey and Bustamante 1999; where TSWA = tropical southwestern Atlantic, TNWA = tropical northwestern Atlantic, TEP = tropical eastern Pacific, and WTSEP = warm temperate southeastern Pacific. Type of comparison describes the method for calculating the RR:R/O = Inside reserve versus outside comparisons, and BACI = comparisons with baseline data both inside and outside the reserve. Biological variable records the variable used as follow: D = density, B = biomass, SR = species richness, and S = body size. Analysis level describes whether the dataset was used for reserve (R), trophic (T) or species level analyses (S). Some reserves names were abbreviated as follow: ACLSP = Admiral Cockburn Land and Sea Park, EHCLR = East Harbour Conch and Lobster Reserve, ECLSP = Exuma Cays Land and Sea Park, and FN = Fernando do Noronha.

Reserve	Biogeographic province	Country	Reserve size (km^2)	Year of creation	No. Spp. Sampled	Type of organism	No. studies	Habitat type (a)	Habitat type (b)	Compariso	Biological variable	Analysis level
Abrolhos	TSWA	Brazil	802	1983	114	fish	1	reef	subtidal	R/O	D	R
ACLSP	TNWA	Turks & Caicos	4	1992	3	fish	1	coral reef	subtidal	R/O	D, S	R, T, S
Baha Loreto	TEP	Mexico	1.28	2001	56	fish, inv	1	rocky reef	subtidal	BACI	D, SR, S	R, S
Barbados	TNWA	Barbados	2.1	1981	48	fish	2	coral reef	subtidal	R/O	D, SR, S	R, S
EHCLR	TNWA	Turks & Caicos	17.5	1993	1	inv	1	seagrass bed	subtidal	R/O	D	R, S
ECLSP	TNWA	Bahamas	409	1959	21	fish	4	coral reef	subtidal	R/O	D, B, S	R, T, S
FN	TSWA	Brazil	95.8	1988	1	fish	1	subtidal	subtidal	R/O	D	R, S
Glover's Reef	TNWA	Belize	74	1993	226	inv	4	coral reef	subtidal	R/O	D, B	R, T, S
Gran Caiman	TNWA	Caiman Islands	6.94	1980	41	fish	1	coral reef	subtidal	R/O	D	R, T
Half Moon Caye	TNWA	Belize	39.25	1982	21	fish, inv	1	coral reef	subtidal	R/O	D	R
Hol Chan	TNWA	Belize	2.6	1987	226	fish	3	coral reef	subtidal	R/O	D, B, SR, S	R, T
Las Cruces	WTSEP	Chile	0.04	1982	4	inv, algae	4	rocky intertidal	intertidal	BACI	D, B, S	R,T, S
Los Roques	TNWA	Venezuela	4	1972	2	inv	2	seagrass bed	subtidal	R/O	D, S	R, S
Manuel Antonio	TEP	Costa Rica	6.82	1972	1	inv	1	rocky intertidal	intertidal	R/O	D	R, S
Mehuin	WTSEP	Chile	0.01	1978	2	inv	3	rocky intertidal	intertidal	R/O	D, S	R, T, S
Punta Francés	TNWA	Cuba	30.15	1968	41	fish	1	coral reef	subtidal	R/O	D, B	R, T
Punta Coloso	WTSEP	Chile	2.18	1988	1	inv	1	rocky intertidal	intertidal	BACI	D	R, S
Saba	TNWA	Netherland Antillas	0.9	1987	40	fish	1	coral reef	subtidal	R/O	D, B, S	R, S
Luis Peňa	TNWA	Puerto Rico	6.36	1999		fish	1	coral reef	subtidal	BACI	B, SP	R, T
Soufriere	TNWA	St. Lucia	0.03	1995		fish	1	coral reef	subtidal	BACI	B, SP	R, T
Discovery Bay	TNWA	Jamaica		1997	3	fish	1	coral reef	subtidal	R/O	S	R, S

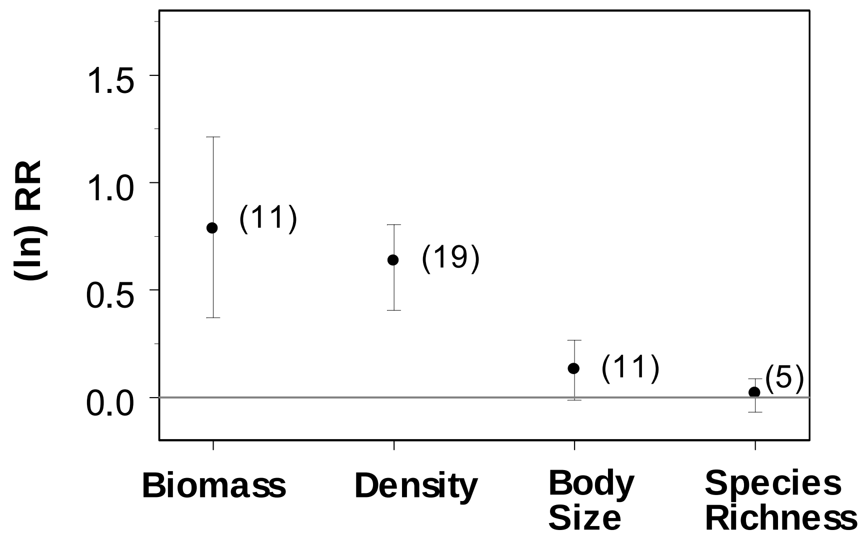

Figure 5. Average (± CI) response ratios (RR) for biomass, density, body size, and species richness of organisms within 21 marine reserves in Latin America and the Caribbean (Table 4).

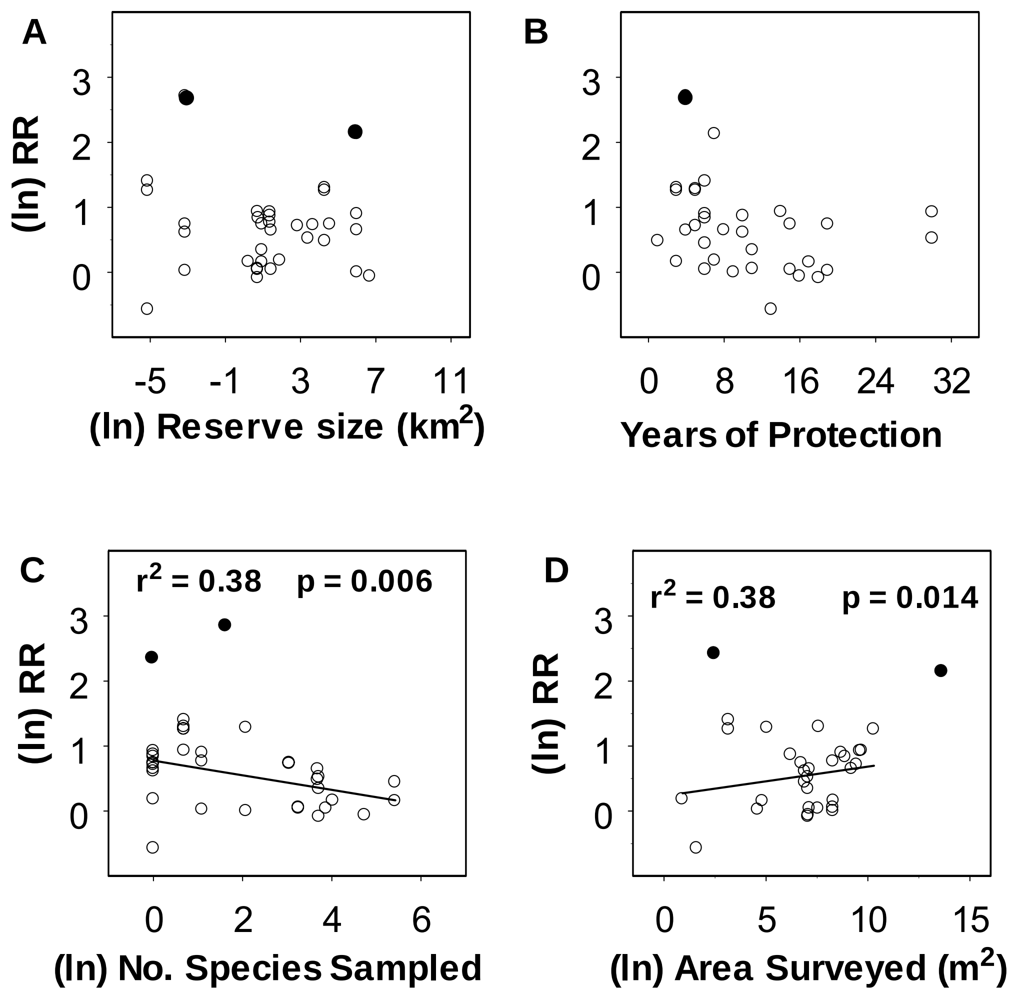

Figure 6. Relationship between the response ratio (RR) and (A) reserve size, (B) years of protection, (C) number of species sampled, and (D) area surveyed using density values in 21 marine reserves in Latin America and the Caribbean (Table 4).

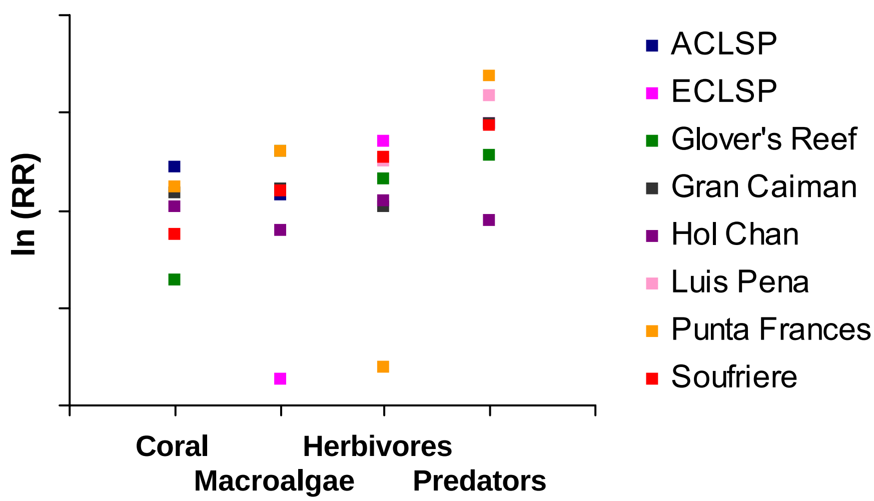

Figure 7. Response ratios (RR) for coral, macroalgae, herbivores, and predators in 8 reserves (Table 4) in the Caribbean. Responses are based on density values for two reserves (Punta Francés and Gran Caiman) and biomass for the other 6 reserves.

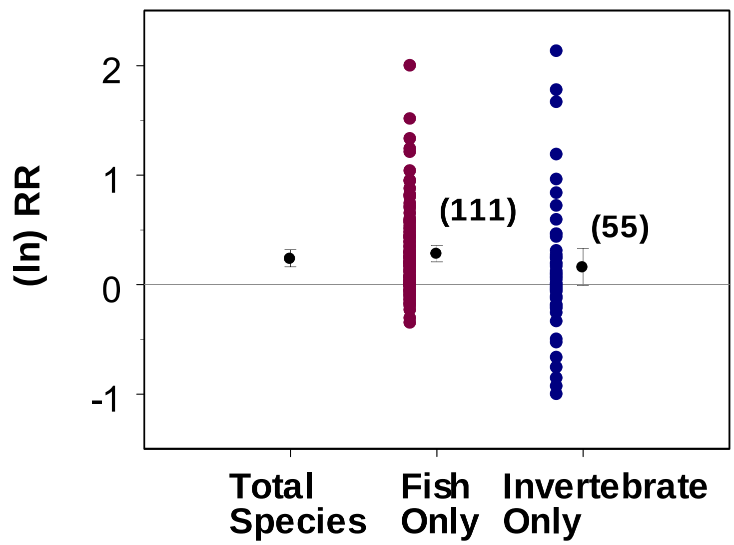

Figure 8. Average (± CI) and single response ratios (RR) for individual species in marine reserves in Latin America and the Caribbean (Table 4). Average and individual species responses are plotted for all species, invertebrates, and fishes using density values.

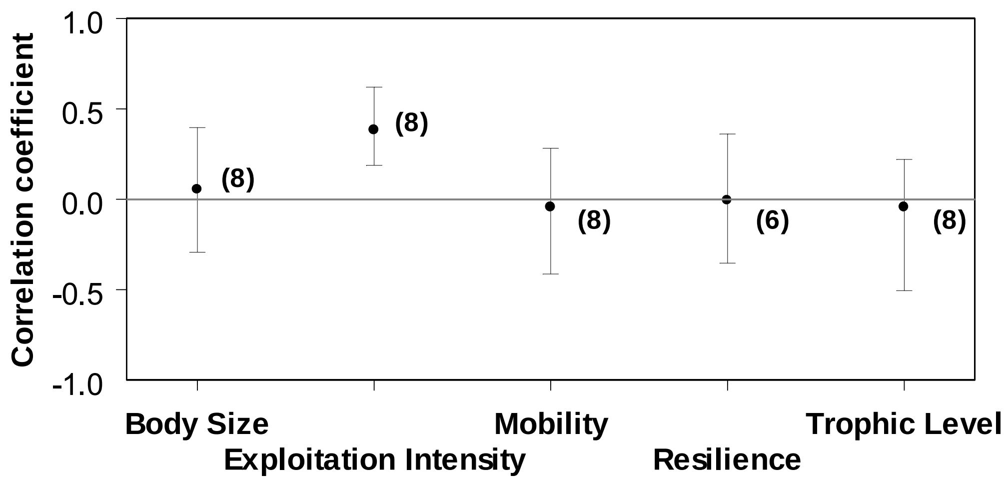

Figure 9. Average (± CI) correlation coefficients for species response ratios and species-specific characteristics (body size, exploitation intensity, mobility, resilience, and trophic level) in 7 reserves in Latin America and the Caribbean (Table 4).

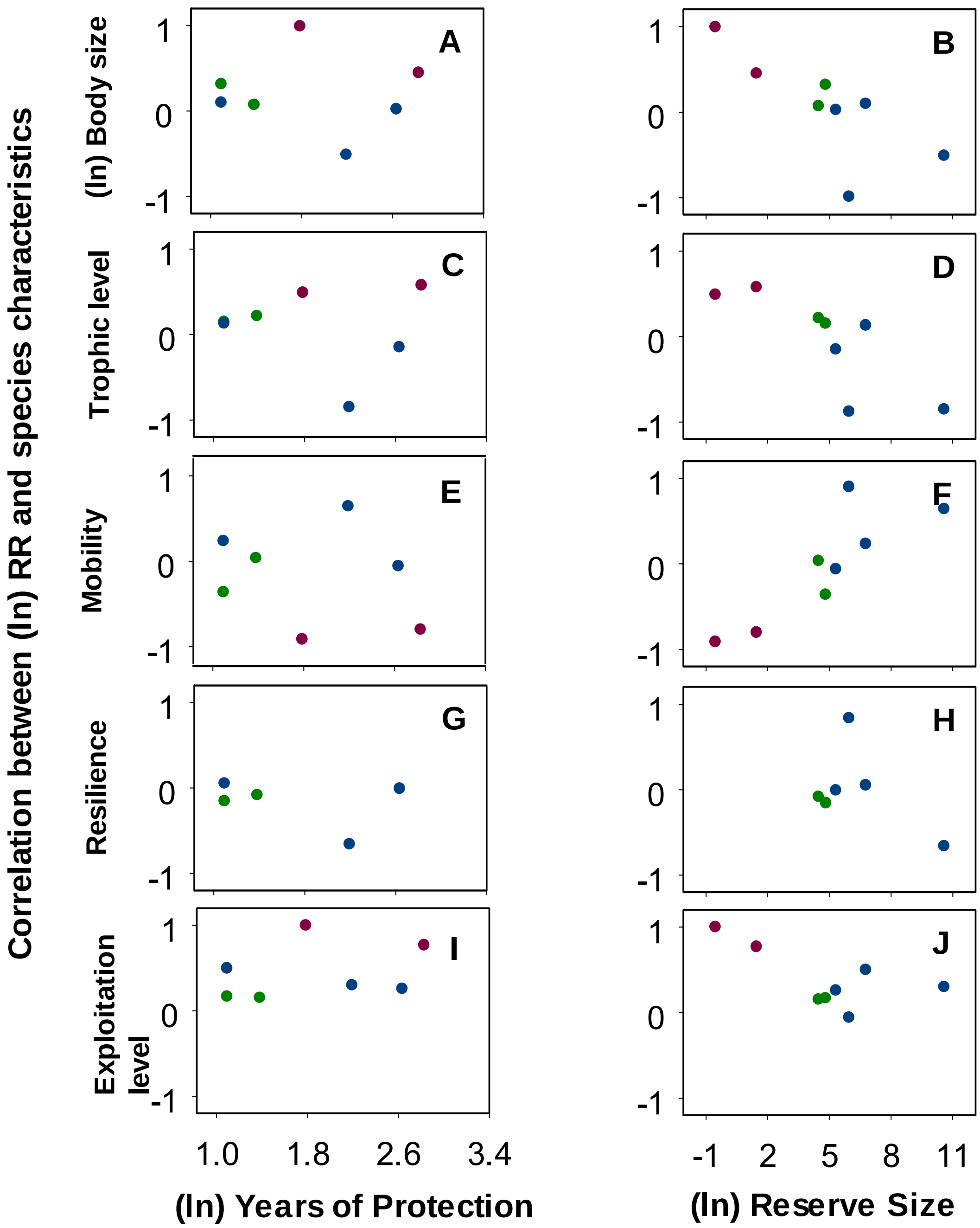

Figure 10. Single correlation coefficients for species response ratios and species-specific characteristics (body size, exploitation intensity, mobility, resilience, and trophic level) plotted against years of protection and reserve size in 7 reserves in Latin America and the Caribbean (Table 4). Red dots represent intertidal reserves, blue dots coral reefs, and green dots rocky reefs.

Chapter 4: General Conclusion

The quantitative analysis of the conservation level of coastal marine ecosystems in Latin America and the Caribbean presented in this thesis may contribute to the advancement of regional initiatives that are urgently needed to achieve marine conservation goals.

The steady increase in the number and extent of marine protected areas (MPAs) over time, including no-take reserves presented in Chapter 2, demonstrates that Latin American and Caribbean countries are actively involved in marine conservation. This constitutes a promising starting point to build a strong regional initiative that can effectively conserve and manage marine ecosystems. But the existing protection is currently insufficient for protecting biodiversity, habitats, ecosystem functioning, and sustainable use of natural resources. Additionally, analyses presented in Chapter 2 provide an important tool for planning marine conservation strategies at a regional scale. Areas that need coverage with MPAs are highlighted, especially the need for including networks of no-take marine reserves in the Eastern Pacific and Southern Atlantic regions.

Moreover, the results in Chapter 2 can be used as a baseline to make future comparisons of the progress of marine biodiversity conservation in this region. The current coverage of MPAs represents only 1.5% of the coastal and marine systems in this region of the world. No-take marine reserves, the core elements for conservation strategies (SCBD 2004), cover only the 0.1 % of the coastal and marine regions that were delineated under the biogeographic boundaries proposed by Sullivan-Sealey and Bustamante (1999).

Chapter 2 pointed out some potential factors that explain the uneven distribution of MPAs in Latin America and the Caribbean. Differential knowledge of the status of marine systems across the entire region, different degrees of organization among countries, and special recognition of some areas as irreplaceable sites for conservation were identified. Additionally, the limitation of MPA coverage as indicator of biodiversity conservation was highlighted because the creation of protected areas does not always result in management effectiveness. Indeed, achievement of management goals of MPAs is generally low for this and other regions around the world (Silva and Desilvestre 1989, Kelleher 1995, Appeldoorn and Lindeman, Mora et 2006), suggesting that improvements in the implementation and management of existing MPAs is essential. This coupled with an increase in the number and extent of well-planned and managed MPAs, in particular no-take marine reserves, can better protect marine biodiversity. In addition, the sustainable use of resources and management of threats outside MPA boundaries are essential. Finally, Chapter 2 pointed out the need for complementing this large-scale assessment with fine-scale analyses (100 km or less) that represent marine habitats between and within biogeographic provinces, and that match the level of community organization and national initiatives (Kappel 2006).

In Chapter 3, using a comprehensive synthesis of ecological effects of marine reserves established in Latin America and the Caribbean, I demonstrated that positive outcomes of reserve protection at assemblage and species levels are occurring. The effectiveness of marine reserves as a conservation tool to rebuild exploited populations was also confirmed, particularly for invertebrates. Less clear was the relationship between responses to protection and species-specific characteristics, suggesting that species from different trophic levels, adult mobility, body size and resilience can benefit from protection. However, the highest density responses observed in our dataset from a spiny lobster (Acosta and Robertson 2003), the queen conch (Stoner and Ray 1996), a carnivorous gastropod (Durán and Castilla 1989), and grunts (McClanahan et al. 2001), suggest that species with high reproductive rates and short life-spans that are targeted by fisheries can respond rapidly to protection (Roberts 2005). In contrast, density of fishes that are highly targeted by fisheries did not exhibit extremely positive outcomes from reserve establishment. When studies reported the effect of protection as trophic group averages (primarily biomass estimates), the magnitude of recovery was more evident for predators than for other trophic groups. This result suggests that biomass is a better indicator of reserve benefits than density. Even slight increases in density combined with increases in body size can result in strong biomass effects, which are relevant for conservation and fishery goals.

Recovery of species from intertidal marine reserves tended to be greater than from subtidal ones, although this pattern needs to be further analyzed with a larger sample size to test its generality and significance. However, it makes sense that intertidal reserves will respond greater and faster than subtidal reserves because of their (1) high levels of human extraction (Moreno 2001, Thompson et al. 2002), and (2) distinct assemblages of species, which generally consist of invertebrate and macroalgal species that tend to recover faster after disturbance events (Hawkins 1999).

Finally, this review showed that the number of publications on marine reserve performance in Latin America and the Caribbean is still limited and the distribution is geographically uneven. The majority of marine reserves and studies are located in the Caribbean (Table 1, Chapter 2). It is unfortunate to realize that quantitative information exists for only 21 of the more than 80 marine reserves created in this region of the world (Chapter 2). Therefore, we highlight here the need for increasing the number of studies that document the ecological effects of marine reserves and making such findings available in peer-reviewed journals.

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