Ecosystem-based management of coral reefs and interconnected nearshore tropical habitats

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1. Introduction

The shallow, nearshore and intertidal areas of tropical regions across the globe harbor some of the most productive and diverse, but also most heavily impacted, marine ecosystems in the world (Wilkinson 2008; De'ath et al. 2012). By virtue of their proximity, coral reefs are inextricably linked to mangrove forests and seagrass beds, in terms of both facing common threats and supporting common species. These interconnected ecosystems provide a wide range of goods and services to human communities, often in otherwise impoverished areas. Indeed, developing nations are disproportionately distributed within tropical latitudes, in part because the hot climate at low latitudes is conducive to drought, famine, disease and natural disasters (Harrison 1979). Yet, coral reefs and other coastal marine ecosystems in the tropics, provide rich reservoirs of natural resources. However, terrestrial and climate impacts, and scientific and governance limitations, have resulted in severe degradation of many nearshore tropical ecosystems, and continued threats to many more. In fact, coral reefs might become the first marine ecosystem to be driven to extinction by anthropogenic activity, perhaps within the next century (Sale 2011).

This chapter will primarily focus on ecosystem-based management of coral reefs, as they are the most diverse marine ecosystem in the tropics capable of producing perhaps the widest array of goods and services (at least on a per-unit-area basis), yet are also the most vulnerable. By necessity we will also examine ecosystem-based management of adjacent seagrass beds and especially mangroves, given the ecological and, at least in some cases, governance linkages between these habitats and coral reefs.

2. Ecological overview

2.1 An improbable equilibrium

Coral reefs are wondrously improbable things, wondrous because of their sheer biological diversity and intricate inter-relationships among resident species, and improbable because the conditions necessary for their existence are both precise and rare. Coral reefs are always living on the edge, and in past geological periods they have been absent for periods of tens of millions of years.

Coral reefs are the only totally biogenic ecosystem, although deep-sea vents and rainforests approach this condition. With the exception of basaltic intrusions in locations such as Hawaii, terrigenous sediments in some reefs near continental coastlines, and exposed underlying bedrock in subtropical locations where coral growth is scant, everything on a coral reef is produced by living organisms. Many Pacific reefs perch on top of thousands of meters of biogenic limestone deposited on a slowly subsiding base over thousands of years. Coral reef formation is tightly constrained by the narrow tolerances of corals and other calcifiers such as calcareous reef-building algae. While there are differences among species, corals generally require a firm substratum, depths less than 50m (unless the water is exceptionally clear), oceanic salinity and pH, temperatures between about 18^{o}C and 35^{o}C, and minimal pollution or sediment load (Wells 1957; Brown 1997).

Every reef is a dynamic equilibrium between accretive and erosional forces. Accretive forces are the active growth of corals, coralline algae, and other taxa that form skeletal structures out of calcium carbonate. These are augmented by physical forces building sand cays and rubble banks, and by chemical processes leading to production of 'beach rock', or consolidated carbonate sands. The erosional forces include waves, tidal scour, and storm damage, and the actions of a diverse group of bioeroders that dissolve, burrow into, or scrape the surfaces of live and dead coral formations (Glynn 1997).

The dynamic equilibrium between accretion and erosion has played out in a world of more or less stable sea level for the last 5,000 years. Prior to that, reefs experienced rising sea levels back to 20,000 years BP. Eustatic forces can cause local sea level to change, and reefs can be drowned or desiccated by these changes, as well as by changes in global sea level. If sea level rises or reefs subside more quickly than they can grow, calcification slows or may stop completely, in which case the calcifying species die and the reef exists as a deep-water, flat-topped guyot. If sea level falls or the substratum rises, reefs are elevated above low tide level and exposed corals die. We are now entering a new period of global sea level rise, and coral reefs will need to maintain a rate of calcification sufficient to counter erosional forces and permit progressive upward growth to keep the reef in shallow water.

2.2 Geological history of coral reefs

2.2a Rugose, tabulate and early scleractinian reefs

As sketched by Veron (1986), coral reefs first appear in the Ordovician, about 490 x 10^{6} years BP, with the evolution of rugose, and subsequently tabulate corals. Reefs dominated by tabulate corals persisted through the Devonian, but dwindled and disappeared by the close of the Permian, about 245 x 10^{6} years BP. Reef-building did not reoccur until the mid-Triassic, about 230 x 10^{6} years BP, with the appearance of early scleractinian corals. Reef-building prospered and Scleractinia evolved through the Triassic and into the Jurassic so that by mid-Jurassic, 175 x 10^6 years BP, most modern orders of corals were present. In the early Cretaceous reef-building was uncommon, but it increased until reef development was as strong by the end of the Cretaceous, 65 x 10^6 years BP, as it is today. The end-Cretaceous mass extinction eliminated nearly all reefs, which were globally absent for about 10 x 10^6 years. Reef-building commenced again and the approximately modern coral fauna was present by the Eocene, 54 x 10^6 years BP. There were further fluctuations in abundance until the widespread reef-building episode coinciding with the end-Pleistocene sea-level rise commencing 10-20 x 10^3 years BP (Veron et al. 1996).

2.2b History of modern coral reefs
Stationary sea levels of the past 5,000 years (Tornqvist and Hijma 2012) have permitted a considerable diversification of habitats in shallow waters as reefs at sea level are forced to grow laterally by some minimal accretion on the seaward face and deposition of eroded sediments on the leeward. Some of these leeward sediments become secondarily cemented and colonized by coral. The result is a considerable variation in habitat type from the leeward to windward edge of a reef and distinctively different windward and leeward slopes. At times when sea level is rising rapidly, reef growth is primarily vertical and the diversity of habitats may be less. However, a variety of differing habitat types represented over distances of l km or less is a feature of all reefs. The considerable drop in sea level during the Pleistocene means that most reefs are relatively young structures that have grown on top of a substratum that was dry land as little as 10,000 years BP. That substratum may limestone of earlier reefal origin, but it can also be limestone of other origin or rock of other type.

2.3 Intricate linkages in a complex ecological structure

2.3a High biodiversity
An obvious ecological feature of coral reefs is high biodiversity. A widely cited estimate is that coral reefs currently support 25% of all marine species (Knowlton et al. 2010). A number of studies have considered how that high biodiversity is accommodated. Considerable emphasis was initially placed on the evident segregation of similar species among habitats, such that between-habitat diversity is particularly high. This partitioning can be over-emphasized, however. Even on relatively less diverse Caribbean reefs there are abundant examples of groups of ecologically similar taxa occurring together in a single habitat but not occupying other habitats that are in turn occupied by other groups of species. Moreover, high temporal variability of assemblage structure within habitats suggests more dynamic conditions than would generate strong niche partitioning (Fig. 12.1). The simplest explanation for their high diversity may be the correct one: species have evolved over a relatively long time in an environment not interrupted by glaciation, and with a substantial area of habitat to be occupied (Sale 1991).
[Figure 12.1]

2.3b Sedentary organisms
Most reef species are sedentary, if not sessile, due to high habitat diversity and typically high habitat specificity. Despite dividing almost equally into species that spawn in mid-water with no egg care and species that care for the eggs until hatching, only a handful of fish species extend care beyond hatching. For the overwhelming majority, larvae are pelagic. The larval phase ranges from about 10 days in a few small Gobiidae and the pomacentrid genus *Amphiprion*, to

about 2-3 months in the Acanthuridae and some other taxa. A typical larval duration for reef fish species is about 1 month, and the larvae develop substantial, though still not well documented, sensory and behavioral capabilities. At the end of larval life, fish settle to reef or other nursery habitat, and while many change habitat with age, the initial selection appears to be a choice, not a process of elimination of individuals that settle to the 'wrong' habitat. Numerous studies that have censused fish in reef habitats immediately following settlement makes untenable the idea that precise habitat 'preferences' are actually due to post-settlement winnowing.

Behavioral studies have revealed that most fishes move about over small home ranges within preferred habitat (Meyer et al. 2010, 2011), and range size scales with fish size (Sale 1978). During post-settlement life stages, reef fishes exhibit a range of fascinating and complex social, communication and other behavioral systems. In fact, before reef fish ecologists began addressing population, metapopulation, and ecosystem-level questions, the earliest research breakthroughs were mostly in the realm of behavior and life history (reviewed by Kritzer and Sale 2006), due in large part to the clear, shallow waters typical of most coral reefs that make them highly amenable to observation and experimentation (Sale 1991).

The tendency toward habitat specificity and a sedentary nature is furthered by the patchiness of reef systems. This patchiness is revealed on scales from meters to kilometers, and the result is even over contiguous habitat, individuals are quite restricted to a site. Species are then represented as numerous, small populations within patches of suitable habitat. Indeed, reef fish species are best viewed as metapopulations, with dynamics both at the local population and metapopulation scale functioning accordingly (Kritzer and Sale 2004, 2006; Fig. 12.2). Such patchiness means that localized impacts can rapidly deplete species that remain abundant only kilometers away.
[Figure 12.2]

2.3c Efficient nutrient capture and recycling
Another characteristic of coral reefs is efficient nutrient cycling. Most reefs occur in waters low in nutrients, but achieve exceptionally high levels of productivity nonetheless. Nutrients are efficiently captured and recycled among organisms on the reef, a process initially described by Odum and Odum (1955), and refined by much subsequent work. Nutrients arrive in the form of plankton that are filtered by fish, corals, sponges, and other planktivores. Some also arrive as dissolved organic carbon (DOC) shed from nearby seagrass beds (Ziegler and Brenner 1999), or as feces of fish that fed over seagrass beds at night, returning to the reef to rest during the day (Meyer et al. 1983). Coral reef food webs are rich with pathways. The mucus of corals and other species are stripped off by water movement and support a rich microbial flora that in turn is fed on by other planktivores or filter-feeders, or accumulates on reef sediments (D'Elia and Wiebe 1990). Feces of fish swimming above the reef are typically consumed multiple times before reaching the substratum (Robertson 1982). Microbial flora contained in reef sediments, distributed as a film over non-living rocky surfaces, or living as endosymbionts of many reef species is rich in cyanobacteria that are highly effective nitrogen fixers (Larkum et al. 1988; D'Elia and Wiebe 1990; Fiore et al. 2010). The result is that water streaming off the leeward side of a reef is notably deficient in the plankton and nutrients found on the seaward side, while primary production on the reef is the highest for a natural ecosystem and rivals that of intensively farmed crops.

2.3d Linkages to other systems

In addition to linkages with the pelagic realm through larval dispersal, coral reefs are linked to other coastal ecosystems. The most important links are with mangrove forests and back-reef lagoons supporting seagrass beds, sponge habitats and sandy, shallow substrata. All of these environments serve as important nursery habitats for reef species, while seagrass beds also serve as important foraging habitat (McFarland et al. 1985). The trophic links between seagrass beds and coral reefs are mediated by fish families including grunts (Haemulidae), goatfishes (Mullidae), emperors (Lethrinidae), and cardinalfishes (Apogonidae). Reef corals have been shown to benefit both from DOC released by seagrasses (Ziegler and Brenner 1999) and from transfer of nutrients by fishes (Meyer et al. 1983).

The ontogenetic links well demonstrated by fish species are also demonstrated by *Panulirus* lobsters, the most important fishery species on coral reefs. Depending on the species, the larval stages settle inshore into precise microhabitats within mangroves, seagrass beds, sponge beds, or other back-reef habitat. They spend days to months in these habitats, perhaps moving to other non-reef habitats as they grow, until they move out to the reef late in juvenile life. On the reef they may occupy a second juvenile habitat before moving into typical adult habitat (Butler et al. 1997). Among fishes, commercially important families such as the groupers (Serranidae), snappers (Lutjanidae), emperors (Lethrinidae) and grunts (Haemulidae) show such behavior, as do numerous families of smaller species.

2.3e Ecological resilience

A final ecological feature worthy of mention is the high resilience of coral reefs not degraded by human activities. High diversity, rich trophic pathways, and intricate relationships among species and habitats that ensure effective transfer of energy and nutrients together ensure that a coral reef is capable of withstanding impacts of storms, outbreaks of disease and pest species such as the crown-of-thorns starfish, *Acanthaster* sp., or fluctuations in recruitment success due to variable larval survival and delivery. Recovery tends to be rapid when the reef is not degraded (Hughes et al. 2010), and the frequency of impacts is not excessive relative to patterns over its evolutionary history (De'ath et al. 2012).

3. Ecosystem services

Coral reefs are distributed among a diverse array of socio-economic and governance contexts, in which opportunities for food and income from terrestrial resources vary just as widely. Consequently, overall reliance upon reef resources and the specific needs and uses are difficult to generalize. However, coral reefs have several attributes discussed above that promote economic importance regardless of location and, despite variability in their value among locales, nowhere is their importance negligible. These attributes include proximity to shore and shallow depth, which together promote easy access. Structurally complex habitat creates an effective coastal barrier and abundant living space. Those habit attributes, combined with efficient capture and conversion of nutrients, generate biomass and diversity that provide cultural, recreational and food resources. However, their nearshore location also creates conflicts with ecosystem services that rely more on the space occupied by reefs, seagrass beds and mangrove forests, rather than their ecological attributes.

This section aims to convey the spectrum of services provided by coral reefs and related nearshore habitats, and provide examples of the scales, importance and challenges associated with those services, but will not attempt a comprehensive review of all services in all contexts (for an earlier, and excellent, complementary review, see Moberg and Folke 1999). The focus is on services that can more readily be measured in quantifiable ecological (e.g., biomass, biodiversity) and economic (e.g., revenue, employment) terms. Therefore cultural, aesthetic and existence values are not covered in depth, although these ecosystems certainly provide those values in varying ways across the globe, and their importance should not be disregarded.

3.1 Shipping

At least since Captain James Cook's *Endeavour* ran aground on Endeavour Reef within the Great Barrier Reef complex in 1770, shipping lanes have generally tried to avoid coral reefs. Unlike most of the high seas, coral reefs occupy shallow water and present a significant navigational hazard. Shipping lanes often run parallel with reef systems, and by necessity must sometimes pass through gaps in reef architecture to reach the open ocean. In many places, development and maintenance of infrastructure needed for cargo ships has meant direct removal, displacement or persistent impacts upon coral, seagrass and mangrove habitats, to the point where these habitats have been severely diminished or lost outright. Therefore, shipping is generally not a key service provided by contemporary coral reefs, seagrass beds and mangroves, and in fact is largely mutually incompatible. One important exception is the cruise ship industry, often a significant component of the tourism sector that is considered below.

3.2 Mining and mineral extraction

Industrial-scale mining of minerals and fuels beneath coral reefs has not been extensive, despite concerns about potentially significant reserves within ancient limestone motivating exploration and extraction (Moberg and Folke 1999). However, mining of emergent and near-surface limestone (Fig. 12.3a) as a building material is practiced in multiple locations such as the Maldives, Tanzania, Sri Lanka, India, Indonesia and the Philippines, where alternatives for construction are limited (Dulvy et al. 1995). Limestone can be mined from fossil deposits that form the foundation of many tropical islands, as well as the skeletal foundation underlying living corals. The presence of fossil corals in many colonial structures across the Caribbean illustrates that this practice has persisted for at least several centuries, if not longer (Fig 12.3b).
[Figure 12.3]

Fossil reef limestone provides the primary architectural foundation for most coral reefs, with living tissue comprising only a very thin surface layer. Therefore, "dead" reef is structurally important, yet takes decades or centuries to accumulate. Furthermore, as sea-level rise drives reefs to migrate landward as well as toward the surface, limestone foundation in shallow areas currently lacking extensive reef development might become more important. In fact, even though structural complexity in shallow back reef areas is generally less than high rugosity fore reef areas, interstitial space in otherwise flat limestone "pavement" provides important shelter for small individuals. Therefore, limestone mining is a service that needs to be managed carefully to promote ecological function and resilience.

3.3 Renewable energy

The most extensive source of renewable offshore energy exploited to date is wind (Inger et al. 2009). Most wind turbines are sited on soft sediments because foundations can be anchored more easily than on consolidated substrates, which require drilling, and the perceived ecological impacts are less (Snyder and Kaiser 2009). Therefore, construction of wind turbines directly on coral reefs seems unlikely, although construction on adjacent soft sediments and seagrass beds that interact with reefs ecologically might be possible. However, wind turbines are also more commonly constructed in developed countries outside the tropical zone, and further offshore where wind fields are greater and conflicts with coastal users are less (Snyder and Kaiser 2009). Since other forms of offshore renewable energy (wave, tidal) are much less developed (Inger et al. 2009), industrial-scale renewable energy is not likely to be a significant service provided by nearshore tropical ecosystems in the near future.

Coastal ecosystems in the tropics do provide renewable energy at an artisanal or subsistence scale through firewood collected from mangrove forests. Like limestone mining, this service is generally utilized to meet local needs where other resources are lacking, and harvest is technologically simple. However, the aggregate importance of this service is significant with mangroves providing wood fuel and construction, and food and medicinal products from fruits and leaves across the globe (e.g., Dahdouh-Guebas et al. 2000). Timber production is sustainable if managed properly, but harvest in many places is done at rates resulting in outright loss of mangrove forests. Valiela et al. (2001) estimate that one-quarter of the 35% loss of mangroves over the last two decades of the 20th century, or 9% of the overall loss, was due to timber harvest. Because mangroves are important nursery habitats for reef-associated species, and provide sediment trapping and nutrient processing services, timber harvest has the potential to compromise other valuable ecosystem services.

3.4 Communications

By virtue of their nearshore location, coral reefs, mangroves and seagrass beds seem likely locations for laying of undersea telecommunications and energy cables. However, this use has received little discussion in the scientific and management communities relative to others. This may be due in part to the fact that coral reefs are predominantly located in the developing world, where technology lags developed nations. As tropical nations increase coastal infrastructure and standards of living, communications might become a service extracted more frequently from these ecosystems. This service has been provided more extensively by coral reefs in South Florida, U.S.A., one of the developed nations with considerable coral reef area, and analysis and debate about the costs and benefits has ensued (Sultzman et al. 2002; Spurgeon 2003). The Florida experience might foretell similar uses and debates to come in currently less developed areas, although increasing importance of wireless communications might reduce the frequency of such conflicts.

3.5 Tourism

Worldwide, tourism is estimated to be the most valuable service provided by coral reefs in terms of revenue generated. This high value is due to the fact that reefs are readily accessible from shore, in shallow water, often clear and sheltered, have diverse and complex habitats, and harbor abundant and diverse fish and invertebrate fauna, not to mention marine mammals, reptiles and seabirds. Costanza et al. (1997) estimate this value to be approximately US$187 billion, or around half of the total value of coral reefs. However, their methodology might over-extrapolate location-specific values resulting in a gross overestimate, especially in light of the considerable variability in tourism value across the globe (Dixon 1998; Brander et al. 2007). Cesar et al. (2003) estimated that coral reefs could generate US$9.6 billion in global revenue annually if managed sustainably, a more modest but not inconsequential figure.

Notably, tourism accounted for approximately one-half of the overall value of coral reefs estimated by Costanza et al. (1997) and one-third of the value estimated by Cesar et al. (2003). So, the relative importance of tourism is comparable by either approach, and it is the most economically important service provided by coral reefs. Using the analysis of Costanza et al. (1997) for a relative comparison if not absolute estimate, the direct tourism value of mangroves and seagrass beds is much less than coral reefs. However, the importance of these habitats in supporting coral reef fish communities makes them an important contributor to the high value of reef-based tourism that might not be captured.

The nature and importance of reef-based tourism varies substantially across the globe. Scuba diving, snorkeling and forms of eco-tourism are by far the most common activities of visitors to coral reefs. The extent to which the tourism industry is developed in a given locale is determined by a combination of factors including inherent natural attributes (water clarity, biodiversity), impacts on those attributes, remoteness (potentially an asset or barrier), transportation and lodging infrastructure, marketing, availability of other activities, and socio-economic attributes (e.g., safety, or perceptions thereof). Consequently, while tourism on coral reefs has been estimated to generate an average of US$184 per visitor day based on data from locales across the globe, the median value is only US$17 (Brander et al. 2007), indicating a distribution with a long tail toward areas with the right combination of the attributes.

Another important determinant of the realized economic benefits of tourism is the management of and reliance upon other services that affect attributes sought by visitors. Burke et al. (2011) compiled region-specific estimates of the value of tourism relative to fisheries, and found that the value of tourism exceeds that of fisheries in all regions considered by a factor of two to one hundred, with the exception of the Philippine and Indonesian region. There, food security is a much larger concern and access to lucrative Asian seafood markets is more direct, so the estimated present value of tourism is approximately one-tenth that of fishing.

3.6 Fishing

Fish provide an average of 15% of daily protein needs for one-third of the world's population, and more than 50% in some African countries (Smith et al. 2010). The majority of those dependent upon fish are in the coastal tropics, where fishing helps meet income and food security needs. The potential aggregate economic value of fishing on the world's coral reefs under sustainable management is estimated to be US$5.7 billion, just over half of the value attainable from tourism or coastal protection services (Cesar et al. 2003). Newton et al. (2007) estimate the aggregate sustainable yield from coral reef fisheries in 49 island nations to be approximately 589,000 mt, although current yields are approximately 64% higher than this level. Like tourism, the absolute and relative importance of fishing varies considerably across the globe (Burke et al. 2011). Moreover, because mangroves are important nurseries for snappers and groupers, species often targeted by reef fisheries, the fisheries production value of mangroves should not be overlooked. In Mexico's Gulf of California, mangroves are estimated to generate US$37,500$\cdot$ha$^{-1}\cdot$yr^{-1} in fisheries production (Aburto-Oropeza et al. 2008).

Sport fishing plays a significant and growing role in many regions, especially where development of the broader tourism sector results in greater visitation by anglers. According to one estimate, the value generated by sport fishing on coral reefs approximates that of snorkeling and other forms of wildlife viewing, other than scuba diving (Brander et al. 2007). Arguably, sport fishing is a service more similar to tourism than other types of fishing because the value derives less from the product caught than from all of the expenses necessary for a successful trip (airfares, lodging, meals, etc.), and the catch is often released. However, in some locales (e.g., Belize) what is often referred to as 'recreational' fishing by local anglers is actually a form of subsistence fishing.

Commercial, subsistence or recreational fishing can be incompatible with diving, snorkeling and wildlife viewing because each use compromises the other. Non-fishermen tourists can occupy space sought by fishermen and affect the behavior of quarry so that catchability is reduced. Fishermen can likewise occupy space sought by tourists and lower abundance of species tourists hope to see either by dispersing or removing them, or removing attractors such as forage fish. Maximizing value of both services therefore requires carefully managing, and most likely separating, these activities. Also, use of reefs for fishing and tourism must not degrade habitat integrity lest these and other services be compromised and value lost.

3.7 Aquaculture

Aquaculture development on coral reefs is more limited than elsewhere. The only aquaculture production to take place directly on the primary reef architecture might be rearing of giant clams, *Tridacna gigas* (e.g., Rubec et al. 2001), whereby young clams are spawned shoreside and moved to rearing beds on or near the reef. In contrast, broodstock of commercially important finfish (e.g., groupers; Tucker 1999) or coral fragments (Delbeek 2001) are removed from reef for grow-out in facilities on land, in deeper water off the reef, or in back reef locations. If managed well, these limited removals and subsequent culture have the potential to reduce impacts of wild harvest for food and the aquarium trade (Pomeroy et al. 2004). Propagation of coral fragments can also help reef restoration, although the scale of degradation relative to the current scale of production means this is likely to primarily be a very localized benefit at present (Hoegh-Guldberg et al. 2007). Coral propagation as a restoration strategy is almost certain to be ineffective, and is therefore not advisable, if the conditions that led to reef decline in the first place are not reversed or at least reduced (Edwards 2010).

Although *in situ* culture of marine flora and fauna in off-reef or back reef areas can reduce effects on wild populations, such operations can still compromise ecosystem function. For example, algal farming in coastal lagoons in Tanzania has been shown to affect abundance, species richness, trophic identity and community composition of nearby fish assemblages (Bergman et al. 2001). Seaweed farming in Tanzania also results in finer sediment, lower levels of organic matter in sediments, declines in sea grass and macroalgae, and reduced abundance and biomass of macrofauna, particularly suspension-feeding bivalves. These attributes are more similar to sand banks than more productive biotic habitats (Eklof et al. 2005). Additionally, some of these ecosystem changes have been linked to health problems among farmers (Frocklin et al. 2012).

The most significant aquaculture development in the coastal tropics takes place not on reefs, but rather in lagoons and the intertidal zone often occupied by mangroves. There, extensive facilities for production of shrimp and other species have been constructed around the world. Systems for integrating aquaculture facilities with mangroves have been piloted in Vietnam (Binh et al. 1997), but these systems have not been widely adopted. Much more commonly, shrimp aquaculture involves removal of mangroves, which means that this is less an ecosystem service and more a displacement of the ecosystem. This displacement is often accompanied by significant pollution of nearby reef and seagrass systems (Rosamond et al. 2000). The overall scale of mangrove displacement worldwide is on the order of 25-50% or more in some regions (Ellison 2008), and more than 50% of that total loss is due to aquaculture (Valiela et al. 2001).

3.8 Coastal protection and buffering

Given their proximity to coastal settlements, coral reefs and mangroves might be expected to serve important buffering roles. These benefits are apparent, but should not be overstated since even healthy, intact habitats might do little to lower effects of the strongest events (e.g., the 2004 Indian Ocean tsunami; Alongi 2008). Still, estimates suggest that, despite the much larger literature devoted to fisheries benefits, coral reefs and mangroves collectively generate greater value through protection and buffering services. Like estimates of the value of tourism services, figures from Costanza et al. (1997) and Cesar et al. (2003) differ by at least an order of magnitude (US$170 billion versus US$9 billion, respectively). However, both studies again agree that the value of buffering and protection services approximates that of tourism. These benefits are derived both from protection from storms and day-to-day sheltering that facilitates access to the sea for fishing, tourism and other uses.

The relative difference between buffering and protection services provided by coral reefs and vegetated intertidal habitats, including mangroves, is much less than the difference in tourism value (Costanza et al. 1997), although the amount attributable to mangroves alone cannot be readily determined since they are combined with non-tropical intertidal vegetation in the analysis. Other global estimates of the value of protection and buffering services of mangroves, akin to those for coral reefs provided by Cesar et al. (2003), are not readily available, although regional estimates are available and generally corroborate the comparative value relative to coral reefs (reviewed by Wells et al. 2006; Conservation International 2008). In fact, many estimates are not for one habitat type or the other, but rather for the combined coral reef/mangrove complex, further illustrating the importance of habitat mosaics in delivering ecosystem services.

4. Threats and impacts

Like ecosystem services, the nature and magnitude of threats to coral reefs, mangroves and seagrasses vary considerably among regions. However, there are attributes common to all coral reefs and associated habitats making them especially vulnerable to certain threats and impacts. Many of these attributes also allow them to provide such diverse ecosystem services. The long-term evolution of reef systems that has generated such high ecological complexity and diversity also makes them vulnerable to rapid and irregular changes relative to those experienced over their evolution. Their shallow and nearshore location that allows easy accessibility also places these systems in direct contact with marine, coastal and land-based anthropogenic activity relative to ecosystems in deeper water and further offshore. We address a broad range of threats and impacts, but again without attempting a comprehensive review. For a thorough review of the long-term decline of a reef system and assessment of the primary drivers, see De'ath et al. (2012).

4.1 Habitat loss

Habitat loss might be the most severe impact compromising the ability of coral reefs, mangroves and seagrasses to deliver goods and services, especially the most valuable ones (tourism, fishing, coastal protection). Three-dimensional structural complexity extending into the water column, and above the sea surface in the case of mangroves and some emergent corals, absorbs and dissipates swells and surge. Structural complexity also creates abundant space for spawning, sheltering and feeding by fishes and invertebrates sought by both fishermen and tourists. Moreover, in addition to resident fauna, these habitats are an attraction in their own right. Unfortunately, all of the habitats in coastal tropical ecosystems are highly vulnerable, and therefore have experienced significant declines (Valieila et al. 2001; Hoegh-Guldberg et al. 2007; Waycott et al. 2009). Direct removal of mangrove wood for fuel, limestone for construction or coral for the aquarium trade, displacement of habitat for development, and incidental damage by ships, divers and fishermen all exacerbate habitat loss. Physiological stresses linked to land-based inputs of sediment, fertilizers, pesticides and other pollutants further enhance losses. Prospects for corals in particular might be dim in light of increasing temperatures, sea level and acidity brought on by climate change and acidification (Hoegh-Guldberg et al. 2007), although seagrasses and mangroves are far from immune to climate-induced stresses.

Shipping has interactions with and impacts upon nearshore habitats in two important ways. First, poor charts, navigational errors or simple carelessness can lead to ships running aground, like Cook's *Endeavour* more than two centuries ago. Such collisions cause localized damage to reef architecture, which can persist for many years due to slow growth rates (Dennis and Bright 1988). Second, contamination of contacted sediments by anti-foulants can inhibit coral recruitment (Negri et al. 2002), can further delay recovery. In addition to groundings, development and use of shipping infrastructure can have impacts. Most shipping infrastructure is at coastal population centers, where reefs are either naturally absent or have been lost or severely degraded by urban and industrial impacts. However, in contrast to cargo vessels, cruise ships are increasingly using more remote and pristine locations that appeal to tourists. Cruise ship anchors can cause significant localized damage (Allen 1992). Of greater concern, construction and maintenance of cruise ship terminals in otherwise less developed areas can lead to more widespread and persistent effects. These include damage to submerged and intertidal vegetation by herbicidal antifoulants (Gardinali et al. 2004), and increased turbidity and sedimentation due to dredging (Goreau et al. 2008).

Anchors, chains and lines from dive, fishing and other vessels can damage reefs, although many dive destinations now use fixed moorings. However, scuba divers themselves damage reefs through intentional or incidental contact. These impacts are each minor compared with anchor damage, but the large number of divers and other users means aggregate effects can be more significant. Areas with high diver traffic have lower coverage live coral and higher coverage of dead coral and rubble (Hawkins et al. 1999; Tratalos and Austin 2001), although the magnitude of effects depends upon topography and other attributes (Rouphael and Inglis 1997). Fortunately, successful strategies for mitigating diver impacts exist, including limiting the number of dives at a site (Hawkins et al. 1999), briefings on responsible practices (Medio et al. 1997), and targeted intervention of high-risk user groups (Rouphael and Inglis 2001). Similarly, destructive fishing practices can significantly reduce habitat cover and complexity (e.g., Mangi and Roberts 2006), but cooperative management, education and enforcement can mitigate this impact (see 6.3b below).

4.2 Harmful algal blooms

In contrast to more enclosed systems such as estuaries and lakes, harmful blooms of planktonic algae (i.e., harmful algal blooms, or HABs) seem to be less of a concern on coral reefs, the waters of which are more open and well-mixed. Indeed, the frequency, magnitude and effects of HABs have received comparatively little attention on coral reefs (Bauman et al. 2010). However, HABs are a threat to coral reefs, and one that has been recognized since at least the 1980s, when severe dinoflagellate outbreaks caused up to 100% coral mortality in near-surface waters of Costa Rica and Panama (Guzman et al. 1990). More recently, a 2008 dinoflagellate outbreak in the Gulf of Oman caused complete loss of two branching corals, and substantial reductions in the abundance, richness, and trophic diversity of fish communities (Bauman et al. 2010). Documenting such effects is motivating research into mechanisms underlying HABs.

A more immediate and significant threat to coral reefs are blooms of fleshy or turf macroalgae that can overrun settlement space for coral and other sessile fauna. Littler et al. (2006) developed a trophic model showing the complex interplay between top-down herbivory and bottom-up nutrient loading in driving algal dynamics on coral reefs. This interplay illustrates the potential severe synergistic effects of multiple stressors, here terrestrial run-off increasing nutrient load and loss vegetation that processes nutrients, alongside overfishing of parrotfishes and other herbivores that reduces algal grazing. Climate-induced effects on coral reproduction, growth and recruitment can also exacerbate ecosystem shifts by creating additional imbalances between algae and corals. Mumby and Steneck (2009) provide a comprehensive overview of these complex dynamics that are increasingly becoming better understood (as they unfold and are documented, unfortunately), changing paradigms about the stability of coral reef systems.

4.3 Introduced species

Like any ecosystem, coral reefs, mangroves and seagrass beds are not immune to introduction of non-native species through accidental or intentional transport. Like most threats, the risks and severity of introduced species vary among systems based on proximity to sources of introduction, such as shipping terminals (ballast water), shoreside aquaculture facilities raising non-native species (e.g., Silva-Oliveira et al. 2011), and residential areas where aquarium species are prevalent (Semmens et al. 2004). As high diversity ecosystems, coral reefs might have greater inherent resilience to effects of introduced species, as the reef community is evolved to cope with a greater and more variable array of residents. Of course, the impact of any introduction will depend upon the characteristics of both the intruder and the system it joins. Two well-known case studies illustrate the range of potential effects: blueline snapper *Lutjanus kasmira* in Hawaii and lionfishes *Pterois volitans* and *P. miles* in the Caribbean.

Blueline snapper were intentionally introduced through documented efforts in the late 1950s and early 1960s by the State of Hawaii to enhance local fisheries, and had spread across the archipelago by 1992 (Gaither et al. 2010). That same year, the first reported release of lionfish into the Caribbean and western Atlantic took place when Hurricane Andrew destroyed a large aquarium at a private residence in Florida, releasing six animals into Biscayne Bay (Carus et al. 2006). This invasion may have been enhanced by subsequent inadvertent, or at least unsanctioned, releases that were not documented. Despite the smaller scale and unintentional nature of the lionfish introduction, its spread was even more rapid, with fish present across the wider Caribbean, along the U.S. Atlantic coast and into the Gulf of Mexico by 2010 (Schofield 2010). Lionfish abundance remains comparatively low in most places it has invaded (Ruiz-Carus et al. 2006), whereas blueline snapper are one of the most abundant species on Hawaiian reefs (Friedlander et al. 2002).

Despite their abundance, Friedlander et al. (2002) detected no strong ecological effects of blueline snapper on the fish community, which perhaps reflects the system having adapted to their presence over four decades. It is too early to determine whether lionfish will have severe ecological effects, although they have already served as a vector for a non-native parasite (Ruiz-Carus et al. 2006). Furthermore, blueline snapper do not represent a fundamentally different species morphologically, behaviorally, taxonomically or toxicologically in their new home. In contrast, lionfish are completely unique among Caribbean fishes. Therefore, predation by or predation on lionfish could have more severe consequences if their abundance dramatically increases. Finally, although introduction of fishes is more visible and garners more attention, introductions of corals, algae and other benthos have been documented and, as the structural foundation for the system, might have greater effects (e.g., Silva et al. 2011).

4.4 Noise pollution

Coral reefs are incredibly noisy places. Butterflyfish emit a variety of sounds to establish territories and warn off intruders (Tricas et al. 2006). Success in conflicts between snapping shrimp depends heavily both producing and interpreting auditory cues (Herbholz and Schmidt 1998). The result of sound production by these and other species is a complex auditory environment, with “pop”, “trumpet”, “drumming” and “banging” calls emitted by multiple species with different daily, lunar, spatial, and seasonal patterns (McCauley and Cato 2000). Beyond the intraspecific interactions driven by these sounds, the collective noise seems to be important in driving larger scale population and community dynamics. Specifically, free-swimming larvae of many fishes use sound as a cue in locating a reef on which to settle (Vermeij et al. 2010). Therefore, sound is a key determinant of inter-reef connectivity and metapopulation dynamics.

Despite the importance of sound, understanding the severity of noise pollution for most marine ecosystems is limited and anecdotal (Popper and Hastings 2009). McCauley et al. (2002) used an air gun to generate high intensity sound that simulated seismic soundings, and showed severe damage to the ears of pink snapper. As their name implies, sound is important for communication in many snappers. Effects of noise pollution on marine mammals have received the most attention of any taxonomic group in the ocean (see Weilgart 2007). Marine mammals are rarely resident on coral reefs, but many interact with reefs seasonally or opportunistically, primarily for feeding. Therefore, effects of noise pollution on marine mammals can have top-down effects on reef ecosystems, whereas effects on prey species can have bottom-up effects on marine mammals.

Noise pollution in the ocean is increasing across the globe. Because sound travels so effectively through water, this can affect vast areas. However, the significance of this impact for any given reef will be dependent on its proximity to urban areas, shipping centers, energy or mineral extraction, military installations and activity, and nearshore industrial development.

4.5 Chemical spills

Pollution impacts have long been a focus of marine ecologists working on coastal habitats. Run-off of sediment, fertilizer and pesticides from agricultural activity and coastal development can substantially alter the physical and chemical profile of waters in which coral reefs, mangroves and seagrasses grow (reviewed by Rawlins et al. 1998). Research has shown that habitats are especially vulnerable in sheltered areas with lower wave energy and flushing, which increases residence time of the contaminant (Gundlach and Hayes 1978).

Beyond localized effects, ship groundings can have widespread and long-lasting effects when oil or other contaminants are dispersed. Persistent petrochemical pollution through leakage or accidents from extraction, processing and distribution facilities is a more severe problem in some locales. This is most notable in the highly enclosed Arabian (Persian) Gulf (Banat et al. 1998; El-Sammak 2001), where isolation, high temperatures and high salinity have shaped reefs with lower diversity but higher relative endemism (Coles and Tarr 1990). While they persist, petroleum products can increase mortality of vegetation and corals (Jackson et al. 1989), and have longer term impacts by compromising recruitment, growth and viability of eggs and larvae (Loya and Rinkevich 1980).

4.6 Biodiversity loss

The high diversity of coral reefs means species loss has two opposing implications. On the one hand, loss of even a small proportion of species means a greater loss of marine species globally than comparable proportional loss in other ecosystems. On the other hand, high diversity ecosystems often have high functional redundancy, and are buffered against species loss to a degree. Of course, some species are more abundant, more unique, play more important ecological roles, or have greater social, cultural or economic importance than others, so the implications depend upon which species are lost. For example, herbivorous parrotfishes play an especially important role regulating algal cover on coral reefs, particularly where sea urchin die-offs have been severe (Mumby et al. 2006). Furthermore, even high diversity systems will eventually suffer functional breakdown if species loss becomes too high.

On coral reefs, overall diversity is strongly tied to coral cover and habitat diversity. An 8-year study of coral cover and fish diversity in Papua New Guinea by Jones et al. (2004) documented declines in coral cover from 66% down to 7%, with associated decline in fish diversity of 22%. Pratchett et al. (2011) reviewed a range of studies like Jones et al. (2004), and found consistent results in many cases. Substantial decreases in fish diversity were evident whenever coral cover decreases by 60% or more. Loss of fish diversity is not directly proportional to loss of coral cover because the rate of decline will vary among species depending upon their dependence on coral. Many species derive refuge, spawning or feeding benefits from dead coral structures, even if demographic rates suffer, until severe structural degradation eventually reduces habitat complexity too much. Corallivores will be affected most quickly due to the loss of an important food source.

The drivers of geological cycles of reef loss and recovery outlined above illustrate the strong potential for climate change to significantly reduce reef biodiversity. Veron (2008) concludes that acidification is the most significant cause of mass coral extinctions. However, acidification also has synergistic effects with sea level rise and temperature regimes, including extreme highs, extreme lows, and rapid fluctuations. Importantly, Veron (2008) also notes that responses of corals to climate-driven stressors varied through time due to the precise conditions present during each episode of extinction and recovery and the largely unknown, but undoubtedly varying, aspects of coral biology that affect susceptibility and resilience. Under ongoing climate change, modern corals might have more capacity to withstand thermal stress than once thought due to ability to alter the composition of symbiotic zooxanthellae in their tissues such that more heat-resistant strains dominate (Berkelmans and van Oppen 2006). However, if acidification remains the major factor in reef decline (Veron 2008), then such adaptations might have limited benefit.

In addition to global loss of corals, Valiela et al. (2001) highlight the fact that global loss of mangrove forest has exceeded that of coral reefs or rainforests with at least 35% of the historical area worldwide lost by the end of the 20^{th} century. This trend will also exacerbate biodiversity loss on coral reefs (e.g. Mumby et al. 2004). Seagrass loss has been nearly as severe, with estimated 29% global loss (Waycott et al. 2009). The combined losses of mangroves, seagrasses and corals within these habitat mosaics might result in nonlinear loss of biodiversity at a more rapid rate than that due to loss of any single habitat type.

4.7 Yield foregone

Mismanagement of harvest, consequent overfishing and other impacts have resulted in tremendous declines in fish stocks and foregone yield in most other marine ecosystems. Worldwide, lost value due to unsustainable fishing in all marine ecosystems is estimated to be US$50 billion annually (World Bank 2009). Lost value attributable to fisheries dependent upon coral reefs, mangroves and seagrasses is unknown, but management effectiveness varies considerably among coral reef fisheries. Newton et al. (2007) synthesized data from coral reef fisheries from 49 island nations. They found harvest levels and resource status relative to estimates of sustainable yield ranged considerable overfishing and even collapse, to potential for increased yields in some nations.

However, Newton et al. (2007) estimate overall yield to be 64% greater than the aggregate sustainable level. Excess yield does not yet represent yield foregone at the global scale, but these fisheries are accruing an ecological debt that will eventually result in substantially reduced harvests. This has already been observed in nations with fully collapsed fisheries, representing around 20% of those studied by Newton et al. The authors predict the offsetting this growing ecological debt would require nearly 200,000 km^2 of additional reef area by 2050, which represents a near doubling of the 255,000 km^2 of reef area worldwide estimated by Spalding and Grenfell (1997).

5. Context and approaches for ecosystem-based management of coral reefs

Principles, tools (science and policy) and processes for ecosystem-based management are outlined in Part I of this volume, building on an extensive literature preceding (e.g., Pikitch et al. 2004; Levin et al. 2009; Worm et al. 2009; Thrush and Dayton 2010). Most proposals for the practice of ecosystem-based management call for attention to a range of ecosystem components, including species, benthic habitats, oceanography, climate, and pollutants, as well as the comparably complex socio-economic systems that affect, extract value from, and are themselves part of the ecosystem (see, e.g., Kritzer 2004 for an example of the consequences of failing to account for human behavior). Some or all of these components can be combined into complex models that illustrate how different uses and impacts affect other services through physical, chemical or biological linkages. This means that ecosystem-based management in its idealized form can be data-hungry and scientifically sophisticated. Moreover, governance systems required to implement ecosystem-based management are complex because of the larger number of economic sectors and stakeholders involved.

This complexity should not be prohibitive and the costs are likely worth bearing, for the alternative is often greater uncertainty in decisions due to including only a limited number of drivers while excluding those that fall within a different scientific discipline, governance entity, or both. For example, the scientific basis for managing fisheries is based primarily on estimates of demographic traits for target species, which are typically time-averaged and not linked to extra-population factors. However, factors such as temperature (e.g., MacKenzie et al. 2008) or predation (e.g., Moustahfid et al. 2009) are taken into account, estimates of reference points, stock status or uncertainty can change drastically. Errors introduced by such omissions can lead to overexploitation and lost value, as well as increased costs stemming from scarcity, economic hardship and legal disputes. Even if the science informing management of a single service is sound, failing to address interactions among uses is likely to lead to inefficiencies, lost value, greater conflicts and higher costs. In other words, complexities and costs associated with ecosystem-based management are likely to be offset by the unplanned costs associated with failing to adopt such an approach.

Unfortunately, it is difficult to test this conclusion because ecosystem-based management has not been adopted comprehensively in any setting, and at best has been implemented in incomplete or incremental ways where it has been tried at all. All nations face the scientific and governance challenges associated with implementing ecosystem-based management to some degree. However, these challenges are often much greater in nations in the developing tropics extracting harboring coral reefs. There, capacity for research, analysis and governance are often insufficient for the demands of ecosystem-based management in its idealized form.

The ecological attributes outlined above make it even more difficult to develop the scientific foundation called for by many ecosystem-based management proponents. The spatially heterogeneous environment (Fig. 12.2) and tendency for organisms to be sedentary and habitat specialists combine to facilitate, and perhaps demand, management at a much smaller spatial scale. The physical structure of reefs and their nearshore and near-surface location force management to be differently structured than in other marine ecosystems. This need for finer scale management does not, however, negate the need for management at larger scales, as climactic and oceanographic processes create ecological linkages and synchronies through dispersal and recruitment processes operating at larger scales (i.e., 10s to 1000s of km). Therefore, ecosystem-based management of coral reefs and associated habitats must be inherently multi-scale in nature. This is true of most marine ecosystems, but the range of scales that must be accommodated is arguably much greater for coral reefs. In addition to spatial complexity, high biodiversity and spatio-temporal variability in assemblage structure (Fig. 12.1) make modeling coral reef food webs and assessment of trade-offs much more complicated than systems with lower diversity and turnover.

In most developing countries, coral reef management usually strives to achieve two goals: sustainable catch for local consumption and trade, and conservation of the structurally and biologically diverse reef environment to deliver other services, especially coastal protection and tourism. While these goals are compatible, limited management capacity in many developing countries means that a rather complex task is not done successfully, and most coral reefs are poorly managed if at all. Management limitations are exacerbated by scientific limitations. Even in developed nations harboring coral reefs (Australia, United States), detailed understanding of more than a few target stocks is rare. Understanding more complex ecosystem-level dynamics to a level that allows informed, quantitative decisions concerning trade-offs between uses is even rarer.

Nevertheless, it is eminently feasible, and quite effective, for coral reef management to be spatially structured, on scales from less than a kilometer to a hundreds or even thousands of kilometers. The use of spatially explicit management strategies, in particular the establishment of marine protected areas (MPAs) of various types, is readily implemented and commonly used. MPAs can take a variety of forms, from fully no-take marine reserves offering full protection from extractive and possibly other uses as well, to those that restrict a more limited range of uses. Where management of coral reefs has shown success, MPAs are often a central, if not sole, strategy, and one that is in many ways inherently ecosystem-based, even if that is not explicit. This is because the underlying rationale for MPAs is to preserve a more unimpacted ecological state, including population structure, behavior, trophic structure, and habitat complexity. To be most effective, this must happen at the scale of individual MPAs and networks that capture processes and spread risk at larger scales (Sale et al. 2005). Also, because coral reefs are inextricably linked to mangroves, seagrass beds, adjacent soft sediment areas (Jones et al. 1991), and inter-reef areas containing oases of “stepping stone” structure (Kritzer and Sale 2006), spatial management should not focus solely on primary reef architecture.

Despite the importance of MPAs, management outside MPA borders cannot be neglected. Fisheries management on coral reefs is usually complicated by the fact that these are inevitably multi-gear, multi-species fisheries. These fisheries are seldom capital-intensive, can develop rapidly in new locations, and operate in a relatively unregulated way in many locations, imposing significant effort on target stocks. Unfortunately, limited capacity means that management of water quality, coastal pollution, and terrestrial run-off is frequently neglected, with pronounced consequences for reefs.

6. Case studies

We review three case studies where management of coral reef systems has shown some measure of success and is ecosystem-based to some degree. The case studies span a range of capacity in terms of science, policy, governance, monitoring and enforcement, and all have developed systems that work within the capacity at hand. All adopt spatially-explicit strategies, including management at comparatively small scales. Still, although these case studies show effectiveness, all continue to confront remaining challenges in terms of emerging threats, competing uses, knowledge gaps, and implementation effectiveness.

6.1 High capacity: Great Barrier Reef

The Great Barrier Reef (GBR) sits within a well-established, though complex, governance context that is relatively well resourced and well informed by research. Ecological sustainability, ecosystem-based management, and precautionary decision-making are common principles in most of the legislation affecting anthropogenic activities on the GBR. Explicit reference to these principles is relatively recent, however, reflecting amendments since the 1990s to older legislation that predated the rise of ecosystem-based management. Hence, ecosystem-based management has not been based on assessments of ecosystem responses to threats, but rather has emerged as a principle shaping the revision of management strategies in the 21st century. The combination of extensive spatial regulation of allowable uses and use-specific strategies, such as conventional fisheries management, grew out of a desire for conservation of ecosystem function in the context of multiple uses in establishment legislation in the 1970s and 1980s. This means the GBR is likely to be managed relatively conservatively in the interests of protecting ecosystem function against anthropogenic activities. Nevertheless, challenges remain in managing activities in the coastal hinterland that affect key processes. Adequately integrating catchment management with reef management will be essential for true ecosystem-based management of the GBR.

6.1a The setting

The GBR is the Earth's largest coral reef system. It is a biologically diverse system including over 4,000 reefs, shoals, and islands across 70 bioregions spread over 15 degrees of latitude along the western margin of the Coral Sea off north-east Australia (Fig. 12.4). It extends from the south coast of Papua New Guinea at around 9^{o}S to around 24.5^{o}S, or about a third of the way down Australian's east coast, and from the coast to over 250km offshore at its widest point, with the outer margin sitting at the continental shelf margin for most of its length.
[Figure 12.4]

Most of this vast system was declared a multiple use marine park (Fig. 12.4) through the Great Barrier Reef Marine Park Act in 1975, motivated mainly by conservation concerns and the desire to prevent oil exploration and mining. The Great Barrier Reef Marine Park (GBRMP) includes 344,400 km^2 of the GBR, comprising over 3,000 reefs, shoals, and islands and all habitats from low water to just beyond the continental shelf break between 10.7^{o}S and 24.5^{o}S. A total of 348,000 km^2 of the GBR, including all of the GBRMP, was included in the World Heritage Register in 1981, meeting all four natural heritage criteria of geological, ecological, aesthetic, and biological diversity significance.

The GBRMP coastal hinterland is extremely sparsely populated over its northern half (to around 17^{o}S), mainly by Australian indigenous people, and moderately populated over the remainder, mostly by widely spaced urban centers of up to 200,000 people. Coastal areas paralleling the GBR generally have higher population density than the national average of 3 people·km^{-2}, with densities upwards of 100 people·km^{-2} in the population centers (Queensland Treasury and Trade 2012). These population centers support a flourishing tourism industry attracting an estimated 6 million visitors annually. Diverse recreational and commercial fishing activities extend across the GBR from coastal towns. They also are commercial and transportation gateways to mining and grazing industries further inland, and sugar cane and fruit growing along the intervening coastal lowlands. Several major ports along the GBR coast are centers for export of ores or livestock and some have ore processing industries, usually close to the coast. The GBR lagoon is also a major shipping route, with an estimated 6,000 transits per year by vessels over 50m long.

The northern extremity beyond the Marine Park and along the eastern Torres Strait to near Papua New Guinea is populated by 19 traditional communities of Torres Strait Islanders, who have inhabited the region for thousands of years. Major human uses of this part of the GBR are commercial fishing, subsistence fishing and hunting, relatively light tourism, and as the northern extent of the alongshore shipping route.

The GBR is a well researched system. The region is home to six island research stations that been established for over three decades, spanning the length of the GBR at 9, 14, 16, 18, 23, and 23.5°S. The Australian Institute of Marine Science and James Cook University have been established in Townsville (19°S) for over 30 years, each with very strong research programs focused on the GBR covering biology, ecology, hydrodynamics, geology, water quality, fisheries, and social sciences. The Universities of Queensland and Sydney and the Australian Commonwealth Scientific and Industrial Research Organisation (CSIRO) also have long-standing research on the GBR. World Heritage Area status and iconic standing have meant the Australian government has invested heavily in research and many international researchers work on projects with collaborators based in Australia.

6.1b Governance

Governance of the GBR is complex and multi-jurisdictional but generally well established in legislation. The GBRMP Act (1975) declared the GBRMP and established the Great Barrier Reef Marine Park Authority (GBRMPA) to manage the park, originally with the goal, '*to provide for the protection, wise use, understanding and enjoyment of the Great Barrier Reef in perpetuity through the care and development of the Great Barrier Reef Marine Park*'. The Act was amended in 2008 to align better with ecosystem-based management principles and other legislation including the Environment Protection and Biodiversity Conservation (EPBC) Act (1999) and Australia's Oceans Policy statement (1998). The Act now requires the GBRMPA to recommend management strategies that, "*… provide for the long term protection and conservation of the environment, biodiversity and heritage values of the Great Barrier Reef Region*" and, "*… allow ecologically sustainable use of the Great Barrier Reef Region ...*". The Act is specific in making '*reasonable use*' subject to maintaining ecological integrity and heritage values and invokes explicitly ecosystem-based management, intergenerational equity, and the precautionary principle. Other Commonwealth Acts apply to activities on the GBR, their application to the GBRMP is expected to be consistent with the GBRMP Act and management strategies in place. In most respects the specific requirement for ecosystem-based management is relatively recent even though the original intent and subsequent management of the GBR have been consistent with principles underpinning ecosystem-based management.

The continental islands of the GBR and the coastal hinterland are governed under Queensland law and local regulations. A plethora of State legislation applies to activities in or around the GBR, including Acts to manage conservation, coastal development, water quality, fishing, ports, and boating, among others. Much of the State legislation dates from the 1990s or earlier but has been updated to give primacy to ecological sustainability, and be consistent with the GBRMP Act. For example, the objectives of the Queensland Fisheries Act (1994) when drafted were, "…

> *(a) ensuring fisheries resources are used in an ecologically sustainable way; and*
> *(b) achieving the optimum community, economic and other benefits obtainable from fisheries resources; and*
> *(c) ensuring access to fisheries resources is fair*".

The "*primary purpose*" of the Act was amended in 2002 to, "*…provide for the use, conservation and enhancement of the community's fisheries resources and fish habitats in a way that seeks to —*

> *(a) apply and balance the principles of ecologically sustainable development; and*
> (b) *promote ecologically sustainable development*",

increasing the emphasis on ecological sustainability and introducing requirements for habitat protection and application of the precautionary principle and intergenerational equity. There is no explicit requirement at this time for ecosystem-based management.

Hence, overarching management of the GBRMP for conservation purposes vests with the GBRMPA but management of many specific activities including fishing, shipping, and coastal development sits with multiple Commonwealth or State agencies outside the explicit jurisdiction of the GBRMPA. Day-to-day management and enforcement across the GBR are shared between the GBRMPA and the State of Queensland, through relevant State government departments including those responsible for parks and wildlife, boating, and fisheries, under "The Emerald Agreement" reached between the Commonwealth and the State in 1977. Enforcement includes structured, randomized aerial surveillance and on-water vessel patrols over the entire region, various regular compliance checks, such as catch and vessel inspections, at multiple ports and boat ramps, water quality monitoring around key ports, and strong penalties for discharge or spillage of oils and other pollutants.

6.1c Conservation and biodiversity management

The GBRMPA relies on spatial regulation as its primary management strategy. Spatial management is implemented via legislated Zoning Plans that classify all areas of the GBR into zones in which various activities are either allowed or prohibited. All forms of mining or drilling are prohibited over the whole GBR. International shipping is restricted to defined routes through the GBRMP and all ships are required to have locally qualified pilots in charge during transit. Zoning for regulation of all other activities ranges from General Use Zones in which most activities, including demersal prawn trawling, are allowed to Preservation Zones, to which all access is prohibited unless for specifically permitted exceptions such as non-destructive research.

Slightly over 33% of the GBRMP is zoned Marine National Park (effectively 'look but don't touch', no extraction) or more restrictive, equivalent to IUCN Categories IA-II, whilst the remainder of the park is accessible for commercial and recreational fishing or collecting to some degree, equivalent to IUCN Categories IV or VI. These figures are the result of a comprehensive revision of zoning in 2004, informed by a detailed bio-regionalization based on a wide array of

ecological, geomorphological, sedimentary, and bathymetric datasets (Fig. 12.5). That review resulted in each of the 70 bioregions, ranging from deep muddy habitats to emergent coral reefs, having at least 20% of their area zoned Marine National Park or a more protected designation, and several bioregions having more than 40% of their area similarly protected (Fernandes et al. 2005a,b). All zones are represented in most bioregions in at least two, and often more, areas that are distributed across the latitudinal and cross-shelf extent of each bioregion. No similar regime of area management exists in the Torres Strait.
[Figure 12.5]

The principle underlying use of zoning as the primary conservation management instrument is that protecting ecological systems from direct interference by human activities will enable the maximum amount of ecological resilience to cope with periodic perturbations. Hence, spatial management could be considered as a coarse instrument for ecosystem-based management. In most cases, spatial management is not accompanied by ecosystem-based assessments of performance, although many single or multi-species abundance-based assessments of the effect of zoning have been made since the 1980s (see McCook et al. 2010 for review). The zoning approach on the GBR also recognizes that explicit and direct control of many activities is beyond the jurisdiction of GBRMPA and so protecting selected areas from those activities might effectively provide refuges, or buffers, against overall adverse effects of those activities. For example, exclusion of fishing from 20-67% (depending on bioregion) of available habitat effectively means that approximately the same proportion of a harvestable stock will be protected from harvest.

Many commercial and all research activities in the GBRMP are regulated by the issue of permits or licensing under Commonwealth or Queensland State regulation in order to identify and manage explicitly the commercial and social values that are dependent on the ecological wellbeing of the GBR. Eco-tourism and local recreational users of the GBR arguably are those most directly benefitting from effective conservation (spatial) management and close regulation of permitted uses, but most other activities (e.g. commercial fishing) also depend on a healthy ecosystem to provide essential economic benefits that flow through the local and national communities.

Other threatening processes, however, are less likely to be moderated directly by spatial management. Ocean warming from climate change, ocean acidification, and the distribution of flood plumes from GBR hinterland catchments, for example, will not respect zoning boundaries and so will affect protected and ‘exploited’ (e.g., through fishing) areas of the GBRMP equally. A recent assessment of significant decline in live coral cover since the 1980s, for example, attributed the main causes of that decline to be repeated outbreaks of the coral-eating crown of thorns starfish (COTS; *Acanthaster planci)*, repeated physical damage from severe tropical storms, and coral bleaching associated with rising ocean temperatures (De’ath et al. 2012). COTS outbreaks have been linked, at least theoretically, to increased nutrient loading through terrestrial runoff from agricultural areas resulting in periodically increased survival of the larval stages leading to population booms. The latter two phenomena have been linked to anthropogenic climate change, though it remains unclear whether the frequency or intensity of tropical cyclones can be attributed to climate change. It remains unknown whether a protected system will be more resilient to such effects than a system already affected by human interventions. A recent review of diverse measures of effectiveness of the GBRMP zoning plans (McCook et al. 2010) demonstrated that areas closed to access or extractive use had significantly better ecological status by a variety of criteria. That review also provided initial evidence that protected communities may be less vulnerable to pan-system perturbations such as COTS outbreaks, terrestrial runoff or ocean warming, though spatial protection from local threats is likely to provide at best an attenuation of global impacts rather than lasting resilience.

6.1d Threats

The GBRMPA in 2009 published a major review of the status, pressures, and threats of the GBR and an outlook of the future challenges for management, including challenges associated with hinterland catchment processes affecting (Anonymous 2009). The main threats were identified as: climate change, including ocean warming and ocean acidification; catchment inputs of sediment, nutrients, and pollutants; coastal development, especially port and major tourism and marina developments; and potential overfishing and the ecological effects of demersal prawn trawling. The aforementioned study by De'ath et al. (2012) adds to that list increases in the frequency and severity of tropical cyclones and COTS outbreaks as the most important drivers of declining coral cover in particular. Cyclone patterns are linked to climate change, and COTS outbreaks might be linked to altered nutrient profiles due to watershed impacts, so the findings are consistent with those of the GBRMPA assessment.

Climate change is driven by factors largely beyond the GBR and responses tend to be based on management that is believed to facilitate resilience. Coastal development and terrestrial runoff are regulated mainly by Queensland State agencies outside the jurisdictional control of the GBRMPA. Prospects of a boom in coastal tourism and port development recently prompted a 'reactive monitoring mission' from World Heritage Commission of UNESCO to review the status of the GBR World Heritage Area (WHA) that concluded that the 'Outstanding Universal Value' of the GBR remained but also included adverse statements on prospective threats to World Heritage status, especially from coastal development and terrestrial runoff (Douvere and Badman 2012). Considerable effort is put into regulation of these activities in the interests of minimizing adverse effects, assisted by sympathetic legislation in many areas that increasingly requires explicit attention to ecological sustainability.

Fishing is perhaps the human use with the most direct effects on components of GBR ecosystems *per se*. Area closures to fishing will provide protection from direct disturbance of included habitats but the extent to which area protection from fishing allows the maintenance of unexploited stock structure will depend on movement of individuals between protected and fished zones at all stages of a species' life cycle, with highly sedentary, low dispersal species being most likely to develop unexploited stock characteristics in protected zones over time (Little et al. 2009a). The likelihood of some exchange between protected and fished zones at some life history stage, however, makes the coexistence of adequate regulations in areas open to fishing an important adjunct to spatial protection. We therefore outline below the interaction between management of one commercial fishery and conservation objectives for the GBRMP as an example of the benefits of complementary spatial and use management.

6.1e Managing fishing with conservation

Fishing within the GBRMP is managed under the Queensland Fisheries Act (1994) through a variety of input controls (e.g., licensing, effort regulation, gear restrictions), output controls (e.g., catch quotas, bag limits, size limits), or access controls (seasonal closures). Commonwealth marine park legislation, including zoning plans, is superior to State legislation applicable in the GBRMP, so fishery management strategies apply only to areas where fishing is allowed. However, the GBRMP and Fishery Acts largely are sympathetic to one another in that both emphasize ecological sustainability, caution, and intergenerational equity.

The line fishery on the GBR is regulated via the 2003 Fisheries (Coral Reef Fin Fish) Management Plan under the Queensland Fishery Act. The fishery is comprised of a commercial line sector, a charter sector, and a recreational sector. The charter fishery caters mainly to paying recreational fishermen whereas the recreational fishery mainly involves people operating under their own control. Harvest by all sectors is highly diverse, with over 125 species taken, though over 80% of the catch is comprised of just two species: common coral trout (*Plectropomus leopardus*) and red throat emperor (*Lethrinus miniatus).*

Commercial and charter fisheries are regulated by limited entry licensing, but there is no recreational licensing. The commercial sector underwent a significant effort reduction in 2003-04 concurrent with the introduction of the Management Plan and revised GBRMP Zoning Plan, as did most GBR fisheries, accompanied by a major structural adjustment package funded by the Commonwealth government to offset the effects of the major increase in area closed to fishing. Commercial fishing is subject to gear restrictions, effort controls, and annual individual transferable quotas for each of the two primary target species and collectively for other species. These quotas were introduced in 2003-04 based on historical catches and are not based on formal stock assessment and adjustment procedures, although a stock assessment has been conducted for *L. miniatus*. Management Strategy Evaluations for coral trout (Mapstone et al. 2004, 2008) and red throat emperor (Little et al. 2009b) have predicted that quotas are unlikely to be realized given current effort controls and associated fishing power, and that has been so to date, though the same analyses indicate that the underlying stocks will remain robust. Recreational and charter harvest is limited by 'in possession' bag limits for day and extended trips.

Minimum legal sizes apply to take of all species by all sectors and some species also have maximum allowable sizes. Extensive biological information exists for the main target species and for many of the minor species and minimum legal sizes generally are set conservatively at sizes likely to have allowed 100% of individuals to spawn in at least one year prior to recruitment to the fishery. Most species are long lived (>10 years), mature at young ages, and have 'table top' asymptotic growth by which they reach near-maximum size typically at less than 20% of expected longevity. These characteristics mean that minimum size limits for several secondary harvest species are set close to maximum size, effectively almost precluding harvest.

The combination of the GBRMP zoning plans that protect over 30% of the stocks from harvest and the conservative off-reserve fishery management strategies are likely to satisfy the objectives of both the GBRMP and Queensland Fishery acts, at least for line fishing, though arguably neither conservation nor fishery management strategies have been designed specifically via assessments of their viability as instruments for ecosystem-based management.

6.1f Summary

The GBR is recognized for high biodiversity value that underpins multiple social and economic values for the Australian and international community. Most of the GBR is included in a highly regulated, strongly managed marine park with well-informed and comprehensive spatial management for biodiversity conservation complemented by generally conservative activity-specific management of human uses. The combination of specific legislative requirements for ecological sustainability and intergenerational equity, stable and well-resourced multi-jurisdiction regulation, and a relatively good understanding of the biophysical system is likely to result by default in successful ecosystem management, even though the current management strategies reflect the evolution of strategies put in place before ecosystem-based management was an explicit policy driver. A future challenge will be to better integrate regulatory frameworks for the GBR and the coastal hinterland such that future ecosystem-based management planning can include comprehensive assessment of all processes ultimately determining ecological sustainability. An associated challenge is to develop robust ecosystem assessments necessary to verify that the system is operating explicitly, rather than implicitly and serendipitously, under ecosystem-based management.

6.2 Intermediate capacity: Cuba

Cuba is the most important island in the Western Hemisphere in terms of biological diversity. Its nearly 110,000 km^2 terrestrial surface area (Fig. 12.6) comprises about one-third of the land area of the Caribbean, but the nation boasts nearly four times as many plant species as Jamaica and almost 12 times as many as Puerto Rico. Recent biological surveys show that 40% of macrofauna species are endemic. More than 50% of the most important ecosystems and 55% of the endemic species of Caribbean islands are found in Cuba. Cuban marine and coastal ecosystems are particularly diverse and productive, as more than 95% of the 3,800 km shelf is fringed by coral reefs (Fig. 12.7a), sea grass beds cover half of the Cuban shelf and mangroves represent one-fifth of Cuban forests (Fig. 12.7b) (CITMA 2010). Although Cuba remains a developing nation, ambitious and evolving policy goals, and a strong and growing scientific infrastructure, are moving the nation in the direction of ecosystem-based management. Spatial management through MPAs and zoning systems is the foundation of Cuba's efforts to more effectively manage its rich ecological diversity for both conservation and human use.
[Figure 12.6]
[Figure 12.7]

6.2a Principle uses and threats

Cuban coastal and marine ecosystems support human settlements, commercial, subsistence and recreational fishing, shipping, tourism, aquaculture, oil and gas development, and other industries (e.g., sugar mills and nickel mining). Nearly four million people, or about one-third of the Cuban population, live in approximately 70 coastal cities and towns that depend on those activities as the cornerstone of residents' livelihoods. Most tourism infrastructure and activity are located in the coastal and marine areas of Cuba. Cuba's average population density is approximately 106 people·km^{-2} (World Bank 2010), with much higher densities in the six cities with 200,000 people or more. Because Cuba is a fairly narrow island (i.e., <200 km wide over its entire 1,250 km length), land-based activities anywhere on the island can have significant impact on coastal and marine systems.

Although most of Cuba's shoreline, marine waters and small islands have suffered less degradation than other Caribbean and coastal tropical locales, several threats to living and non-living resources are of concern. In its National Environmental Strategy for 2011-2015, the Cuban government identified the major threats and impacts as soil degradation, loss of forest cover, loss of biological diversity, pollution, water shortages, climate change, and human health issues (CITMA 2010).

6.2b Policy and governance context

Cuba has a diverse set of legal, policy, scientific, and governance tools to deal with threats to its natural heritage (CITMA 2010). In 1994, Cuba created the Ministry of Science, Technology and the Environment (CITMA), elevating environmental issues to the ministerial level. With the creation of CITMA, the National Center for Protected Areas (CNAP) was born in 1995. CNAP has played a critical role in creating the National System of Protected Areas (SNAP) and achieving important conservation benefits. The Executive Committee of the Council of Ministers (CECM), one of the highest decision-making bodies in Cuba, has the authority to declare protected areas. This provides important government support and a vehicle for potential rapid implementation, although scientific limitations, stakeholder conflicts, and other barriers still present delays. Decisions are also supported by a SNAP Coordination Board, a collaborative body directed by CNAP and including five other key decision-making institutions: Fishery Regulations Office, State Forestry Service, National Enterprise for Flora and Fauna, Corps of Forest Wardens, and Directorate of the Environment. The approval and planning of protected areas occurs through a participatory process, including national and provincial agencies, local communities, and relevant socio-economic groups (e.g., fishermen, tourism workers) through formal and informal meetings, negotiations, planning workshops, conflict resolution, and education.

The three main legal tools governing management of shallow coastal tropical habitats in Cuba include Environmental Law 81, which is the framework law for environmental management, Decree Law 201 outlining operations of the SNAP, and Decree Law 212 for Coastal Zone Management (CITMA 2010). Law 81, adopted in 1997, defines the SNAP as an integrated marine-terrestrial system and establishes its objectives and basic principles. Decree Law 201, adopted in 1999, is the primary legal instrument for the SNAP. It defines protected area categories (which are the same for terrestrial and marine areas), administrative issues, mechanisms for proposals and approvals, and guidance for participatory area planning. Decree Law 212, adopted in 2000, defines the coastal zone and different sub-zones within it based on shoreline type (sandy, rocky, mangrove, etc.). Decree Law 212 also outlines procedures for uses of the coastal zone.

Beginning in 2003, MPAs established under the SNAP include 5-year management plans (CNAP 2008a), supported by methodologies for preparation and operations (CNAP 2008b), and for determining carrying capacity of visitors in MPAs (CNAP 2006). In addition to MPAs created primarily for conservation, Cuba has been implementing different types of multi-use management zones. These include spatial management strategies established by fisheries institutions known as Zones under Special Regimes of Use and Protection (ZBREUP). These zones are effectively fishery reserves that act as additional protected areas.

Building upon lessons learned within the fishing sector through implementation of ZBREUPs, a similar and relatively new type of zone is being developed to solve conflicts among a more diverse array of uses and users. These Zones under Regimes of Integrated Coastal Management (ZBRMIC) are intended to coordinate government, community, science and the economy in the pursuit of conservation and socio-economic development in coastal and marine zones. In 2007, CITMA established a nationwide procedure to assess and approve processes of integrated coastal management throughout the country. This procedure verifies that each proposal has guaranteed minimal organizational, functional, technical and methodological requirements to start a process of integrated coastal management, and oversees the implementation and functioning of the ZBRMICs.

6.2c Scientific basis for spatial management

Cuban MPAs are proposed, selected and designed based on high biological diversity (species richness), presence of coral reef, presence of endangered, rare, charismatic and depleted species, high tourism interest or potential, and low conflicts among users. Among developing countries, Cuba has a relatively strong, and growing, scientific foundation with which to measure sites against these criteria, although considerably more information is still needed to more effectively manage most existing MPAs and to design new ones. However, research emerging from Jardines de la Reina (Gardens of the Queen) National Park is an especially good example of an information base being built to guide design and management of Cuban MPAs.

A comprehensive checklist of fish species has been developed for the Jardines de la Reina (Pina-Amargós et al. 2007), and several reports of species new to science on both global (Ortea and Espinosa 1998; Espinosa and Ortea 1998a,b,c, 1999) and national (Abreu y del Valle 1998; Ibarzábal 1998; Claro et al. 2000, 2001; Hernández-Fernández and Varela 2007; Varela et al. 2008; Guimarais et al. 2009; Acevedo et al. 2010; Parada-Isada et al. 2012) scales have been made. More detailed information on life histories, behavior, connectivity and population dynamics has been compiled for many of the larger species with greatest tourism value and conservation concern, including sharks (Graham et al. 2007; Fig. 12.7c), goliath grouper (*Epinephelus itajara*; Pina-Amargós and González-Sansón 2009; Fig. 12.7d), sea turtles (Moncada 1998, 1999, 2006, 2010; Medina et al. 2010; Fig. 12.7e), and other fish species (Pina-Amargós et al. 2008a; Castellanos-Gell et al. 2012).

Other research has focused on community- and ecosystem-level processes. Characteristics of coral reef habitats have been described (Pina-Amargós et al. 2008b), with particular attention to community composition of octocorals and stony corals and patterns of recovery from bleaching (Hernández-Fernández et al. 2011). Similarly, overall trophic structure of coral reefs in the Jardines de la Reina has been described (Newman et al. 2006), with more detailed investigation into demography of longspine black sea urchin (*Diadema antillarum*; Martín-Blanco et al. 2010) and their effects on algal communities (Martín-Blanco et al. 2011). Impacts of hurricanes on marine ecosystems have been studied to understand implications of severe and large-scale disturbances (Pina-Amargós et al. 2008c). Effects of management actions have been studied in terms of spillover of fish from MPAs (Pina-Amargós et al. 2010). Finally, to supplement this body of biological and ecological research with socio-economic information, gamefishing (Figueredo-Martín 2010a) and diving (Figueredo-Martín, 2010b) activities have been characterized.

Beyond efforts to build scientific foundations for individual sites, an important strength of the SNAP is that it has taken into account complex macro-scale issues. These include the interconnections between terrestrial and marine ecosystems, and issues specific to marine habitats. SNAP also considers connectivity among marine populations and intends to build a network of interconnected MPAs through knowledge of oceanographic processes, larval dispersal (Paris et al. 2005), and spawning aggregations (Claro and Lindeman 2003). Resilience in the face of climate change is also considered and has been taken into account within several components of SNAP to produce a system that more effectively manages shallow coastal habitats and their biological diversity.

6.2d Progress toward spatial management

Cuba began efforts to declare protected areas in the early 20th century. Sierra Cristal National Park was established in 1930 under Presidential Decree No. 487 (CNAP 2008a; Fig. 12.6, 12.8). By the century's end, many of Cuba's ecosystems were represented in protected areas, and Cuba's progress toward ecosystem-based management is founded upon this spatially explicit framework. The ambitious policy related to protected areas is novel in that it integrates marine, coastal and terrestrial systems. Cuba's SNAP has 253 protected areas planned, which will help to protect 20% of the national territory, 17% of terrestrial area and 25% of marine area, including 26% of Cuban wetlands and 57% of coral reefs (CNAP 2008a).
[Figure 12.8]

Progress toward these ambitious goals has been dramatic in the 21st century. In 2001, the Executive Committee of the Council of Ministers (CECM) declared the first 18 Cuban MPAs (CNAP 2008a; Fig. 12.8). In 2004, Cuba had 45 protected areas approved, including 21 with coastal and marine components, which represented 28% of the SNAP and covered 6% of Cuba area (CNAP 2008a). By 2012, Cuba had almost doubled these figures: 103 protected areas approved, including 56 with coastal and marine components representing 41% of the SNAP, covering 17% of Cuba's area and 20% of marine waters (CNAP 2012; Fig. 12.8).

The Ministry of the Fishing Industry (now the Ministry of Food Industry) through the Office of Fishing Regulations began declaring ZBREUPs in 1994. Initially, most did not allow harvest of finfish, with the exception of catch and release gamefishing, but did allow lobster fishing. Currently, ZBREUP are moving towards a more diverse set of regulations and zoning types, similar to protected areas zoning (Fig. 12.6), although a finfishing ban remains. ZBREUP have been the initial phase for many Cuban MPAs, such as Jardines de la Reina National Park (Fig. 12.6), which was declared in 2010. Jardines de la Reina Marine Reserve, established in 1996, now sits within the National Park, and is the largest marine reserve in the Caribbean (Appeldoorn and Lindeman 2003).

The biodiversity value, size, and importance of Jardines de la Reina National Park and Ciénaga de Zapata (Zapata Swamp) National Park (Fig. 12.6) warrant special mention. Jardines de la Reina has been effective in protecting large, economically important fish such as sharks (Fig.

12.7c), snappers and groupers (Fig. 12.7d) inside the marine reserve (Pina-Amargós 2008), and also supplying biomass to surrounding fishing grounds (Pina-Amargós et al. 2010). A special protected area classification, related to IUCN category VI, is the Special Regions of Sustainable Development (SRSD) category. This type of protected area includes large areas of high economic and conservation importance. Iconic Cuban SRSD are the two largest archipelagos in Cuba (Sabana-Camagüey and Canarreos) and Ciénaga de Zapata, the largest wetland in the insular Caribbean (CNAP 2008a; Fig. 12.6).

In the direction of integrated management of multiple uses, Cuba now has in place 15 ZBRMIC since the first six were approved in 2008 (Fig. 12.8). The largest are Península de Guanahacabibes, Golfo de Guacanayabo, and Ciénaga de Zapata. Thirteen ZBRMIC are accomplishing their objectives and eleven have protected areas within. There is strong synergy between ZBRMIC and protected areas, especially MPAs, given that protected areas complement and make feasible ZBRMIC objectives including conservation and reducing user conflicts. Conservation actions and protected area planning often become part of ZBRMIC management plans. Some existing MPAs have become part of a larger ZBRMIC. Mechanisms developed in creating the SNAP have served as a basis to effectively implement integrated coastal management programs. One CITMA objective is to approve at least three ZBRMIC annually. Thus, by 2015, Cuba is expected to have 28 ZBRMIC comprising 63% of its coastline.

ZBRMIC have made important contributions to management in Cuba. They provide space for conflict resolution, while promoting integration among management plans inside the ZBRMIC (forestry, fisheries, territorial planning, protected areas). Participating in the ZBRMIC process has facilitated qualification and environmental education of institutions and communities, and created room for citizen participation through representation on management bodies. An especially important result has been greater participation, commitment and integration of local governments in management. ZBRMIC have created opportunities to carry out specific environmental actions with an ecosystem-based and multi-user approach.

Ecosystem-based management through MPAs, ZBREUP and ZBRMIC has been applied in projects such as the Cuba/GEF/UNDP Sabana-Camagüey Project from 1993 to the present (Proyecto Sabana-Camagüey 1999) and Cuba/GEF/UNDP Archipiélagos del Sur Project from 2010 to the present (Proyecto Archipiélagos del Sur 2009). Integration of these tools has increased protection and sustainable use of natural resources with seven ZBRMIC and eight MPAs already declared in Sabana-Camagüey archipelago (Proyecto Sabana-Camagüey 1999), and eight ZBRMIC, five extended MPAs, six new MPAs and several ZBREUP proposed within Cuba's southern archipelagos (Proyecto Archipiélagos del Sur 2009).

By 2014, the Archipiélagos del Sur Project will increase MPA coverage from 28% to 35%, MPA area inside ZBRMIC from 0% to 68%, total ZBRMIC coverage from 0% to 47%, and total ZBREUP coverage from 12% to at least 20% in Cuba's southern archipelagos (Proyecto Archipiélagos del Sur 2009). Habitat representation in Cuba's southern archipelagos will increase substantially: coral reefs in MPAs will increase from 12% to 20% and in ZBRMICs from 0% to 48%, seagrass beds in MPAs will increase from 11% to 19% and in ZBRMICs from 0% to 36%, and mangroves forests in MPAs will increase from 66% to 74% and in ZBRMICs from 0% to 73% (Proyecto Archipiélagos del Sur 2009).

6.2e Remaining challenges
Its progress notwithstanding, Cuba faces important challenges in implementation of its multi-faceted spatial management system, many of which are discussed in the National Environmental Strategy (CITMA 2010). Operationally, enforcement to prevent and punish illegal fishing and hunting and trade of endangered species (Fig. 12.7e,f) is weak, which limits effectiveness within and outside managed areas. Procedures for protected area approval are often too time-consuming and complicated, unnecessarily delaying implementation. Also, environmental impact assessment procedures and resulting spatial planning have not always sufficiently considered intrinsic and use value of biodiversity. In terms of geographic coverage, ZBRMIC are underrepresented in southern Cuba, accounting for just 27% of the national total.

The three components of Cuba's spatial management system (MPAs, ZBREUP, ZBRMIC) are still mainly seen by decision makers and the public as separate tools. A key challenge is articulating, designing and implementing these components in a way that not only allows each to achieve its objectives, but also to perform synergistically toward common goals. Although integrating these tools in the sea will be a significant challenge in its own right, integration with other structures such as Watershed Management Units will be necessary as well. Better educating decision makers and stakeholders about ecosystem-based management concepts and practices, and then conducting pilot projects to demonstrate effectiveness, are critical steps toward more effective integration. To maximize success, greater capacity for adaptive management than available at present is necessary so deficiencies are quickly identified and corrected. For example, many MPAs boundaries are not well placed to facilitate enforcement, and adaptive management strategies could help adjust boundaries as lessons are learned after implementation.

In addition to improved coordination, education, and enforcement, overcoming scientific deficiencies will be critical. Baseline studies to determine trends of biodiversity loss and recovery after impacts and mitigation are lacking. Knowledge gaps exist for several taxonomic groups, primarily invertebrates and microorganisms, limiting the ability to determine status and threats, and adopt conservation measures. Effects of climate change, ocean acidification, increased frequency and strength of storms, sea level rise and invasive species are poorly understood in Cuba, as is the value of ecosystem services needed to understand the full economic impact of those processes. Individual managed areas and networks should be designed taking into account oceanographic phenomena, larval dispersal, location of spawning aggregations, juvenile and adult movement, and nursery grounds.

Building the needed science is compromised by funding. Although all MPAs and ZBRMIC are financially supported through national and international projects, funds are insufficient for thorough management, enforcement, research, and monitoring. Reliance upon international sources is especially high for MPA management, which are far from being self-sufficient. Additional economic and social science research could better estimate values of ecosystem goods and services, allowing more appropriate allocation of resources and development of financing mechanisms to cover costs.

6.2f Fisheries management

Although Cuba faces important challenges in management of MPAs, ZBREUP and ZBRMIC, far greater challenges exist in the majority of national waters outside of managed areas (Fig. 12.6). A variety of uses take place in these areas, including transportation, tourism and growing interest in energy production (Pinon and Muse 2010), fishing is the most widespread activity involving the largest number of people. The Cuban fishing industry experienced drastic changes following the break-up of the Soviet Union in the early 1990s, and with it the loss of Soviet markets and subsidies. The fleet transitioned from one comprised of fewer and larger vessels targeting high volume and low value species to one comprised of more numerous smaller vessels targeting a greater diversity of low volume but high value species (Adams 2000). Application of Schaefer models to lobster, shrimp, and tuna suggests management of those stocks has been reasonably successful since this transition (Baisre Hernandez 2006). However, concerns persist about sustainable harvest of other species, particularly reef-associated finfish (Claro et al. 2009). Scientific deficiencies are again a barrier, and research is needed on many harvested species, as well as impacts of multi-species and multi-gear fisheries on an ecosystem scale.

To address the changed character of Cuba's fisheries, management has been moving toward more decentralized approaches (Adams 2000), which might lead to adoption of cooperatives, territorial user rights for fishing (TURFs), and other rights-based approaches that have been demonstrated to be more effective in lower capacity settings (Gutierrez et al. 2011). The Cuban government passed a new decree law in 2012 authorizing creation of cooperatives in a variety of industries, including fisheries. This policy has the potential to fundamentally shift the nature of incentives, responsibilities, and interactions among fishermen, scientists and decision makers. Given the large number of small-scale fishermen spread among many small and widely dispersed ports, those changes hold great potential for more effective management (Prince 2003). However, carefully planned pilot projects designed to determine necessary elements of these governance systems in the Cuban context will be essential to building workable models.

6.2g Summary

Cuba has made important progress toward ecosystem-based management, even without explicitly defining it as such, through an ambitious spatial management program. Sustaining marine and coastal ecosystems will be essential to preserving Cuba's natural heritage for its socio-economic well-being. Replicating successes, and especially increased attention to scientific and management deficiencies, will continue moving Cuba toward ecosystem-based management.

Of course, Cuba's efforts are relatively young, so it is too early to fully evaluate the outcomes. However, two case studies show potential benefits to local communities of MPAs that are well designed, well managed, and supported by science. After declaration of Punta Francés National Park, protection of fish and coral, combined with improved administration, are predicted to generate more than US$100 million annually the nation (Angulo-Valdés 2005). JRMR has already fueled a thriving tourist economy (Figueredo Martín et al. 2009) through protection of large charismatic species such as sharks, snappers and groupers (Pina-Amargós 2008), high quality dive sites (Figueredo Martín et al. 2010a), and opportunities to fish for bonefish, tarpon and permit (Figueredo Martín et al., 2010b). Additionally, JRMR benefits coastal communities dependent on commercial fishing (Figueredo Martín et al. 2009) through spillover of biomass to surrounding fishing grounds (Pina-Amargós et al. 2010).

6.3 Low capacity: Kenya

Kenya's 600km coastline is situated just south of the equator (Fig. 12.9). Extensive coral reefs, mangroves, and nearshore islands are found in the far south, and a fringing reef, approximately 3 km from the shore, extends for much of the coast as far north as Malindi. This fringing reef creates a wide lagoon where the majority of marine resource extraction occurs. The coast has a high population density (~40 people·km^{-2}) but below the national average (~66 people·km^{-2}) (Kenya Bureau of Statistics 2012). Activities along the coast are a major contributor to the national economy, although the region remains geographically, historically, and socially distinct.
[Figure 12.9]

6.3a Ecosystem services

Coral reefs in the western Indian Ocean provide multiple ecosystem services (Fig. 12.10). Services that provide economic benefits are most commonly quantified, but social and cultural services such as education (i.e., local ecological knowledge) and bequest (i.e., knowledge that future generations will benefit) are recognized, and in some areas their importance exceeds economic benefits (Rönnbäck et al. 2007; Hicks et al. 2009). Appreciation of these services can contribute to adaptive management, enabling learning from and adapting to changes in the coupled social-ecological system.
[Figure 12.10]

The port of Mombasa, in Kenya's principal coastal town (Fig. 12.9), was originally established as part of the spice trade route in the sixteenth century. The port has the best natural harbor in East Africa, and has become the largest port in the region (Foeken et al. 2000). Mombasa port handles more than 15 million tons of cargo each year, contributing significantly to the region's economy (Kenya Ports Authority 2012). Along the coast, erosion is a critical issue, as many hotels are built within 30m of the high tide mark (legal limit) and have had to invest in coastal defenses (i.e., sea walls). The reef provides some protection and for most of the year calm conditions exist in the lagoon, making access to the coral, seagrass, and sand habitats by swimming or dugout canoe possible.

This reef system, and the easy access to it, sustains livelihoods and nutritional needs of many coastal communities. The majority of fishing in Kenya is conducted in the lagoon. Fishing tends to be small scale, artisanal, or subsistence, and fishermen use a diversity of gears to land a diversity of species (McClanahan et al. 2008b). Approximately 30% of coastal households are involved in fishing (Cinner et al. 2009a) predominantly as a primary occupation. Subsistence farming, fish trading, or the informal sector (i.e., economic activity not regulated by the government) provide secondary occupations (Cinner and Bodin 2010). This suggests that, although fishing may not generate a great deal of revenue, it is vital to coastal livelihoods. Limited industrial-scale fishing also takes place, and the government receives limited revenue in international trawler license fees.

Tourism is a major industry in Kenya contributing 12% to GDP. Sandy beaches and coral reefs attract many tourists. Although tourism is important to the national economy, and its value dwarfs that of other coastal ecosystem services (Hicks et al. 2009), local and regional benefits are minimal (Sindiga 1995). The informal sector is responsible for local benefits accruing from tourism, and only ~7% of households along the coast are involved in tourism (Cinner et al. 2009a).

A cement quarry set back from the coast excavates fossil reef for construction. This cement factory generates income and employment. Associated with the quarry are ecosystem restoration projects and visitor education centers providing recreational and educational benefits. Mangroves bordering the coast provide local provisioning, regulating, cultural and other services (Rönnbäck et al. 2007). Perhaps most significant at a global scale is the carbon sequestration service mangroves provide.

6.3b Threats and impacts

Climate change and fishing are the two greatest threats along the Kenyan coast. These threats influence different components of the ecosystem with impacts potentially reinforcing one another. Fishing targets large-bodied fish and climate change induces habitat loss and declines of small species (Graham et al. 2011). Other common climate effects are bleaching and mortality of corals. Kenya's worst coral bleaching event occurred in 1998 when the El-Niño Southern Oscillation (ENSO) coincided with the Indian Ocean Dipole (IOD), resulting in the warmest ocean temperatures on record and up to 90% coral mortality across the western Indian Ocean (McClanahan et al. 2007a; Graham et al. 2008). Loss of coral cover following bleaching (Fig. 12.11) opens up space for recolonization by fast growing algae or weedy coral species (Green et al. 2008; Darling et al. 2010). Changes in benthic composition have substantial impacts on fish assemblages, leading to reductions in species richness and abundance of small-bodied and strongly coral-dependent species (Wilson et al. 2006; Pratchett et al. 2008). Substantial changes in fish communities occurred across the western Indian Ocean following the 1998 bleaching event (Graham et al. 2008). However, changes in Kenya were relatively small, likely because the reefs and fish assemblages were already altered by fishing.
[Figure 12.11]

Destructive and unsustainable fishing is the main local threat to Kenya's coastal systems. Dense human populations lead to heavy fishing pressure, and population density at Kenya's coast is predicted to increase. Furthermore, natural population growth is likely be exacerbated by climate-induced coastward migration. Population density increases demand for fish and the number of fishermen pursuing a dwindling resource, creating a Malthusian tendency for fishermen to use increasingly destructive methods (Pauly 1989; McClanahan et al. 2008b). These three factors – growing population, climate-induced coastward migration, and destructive fishing – combine to intensify fishing impacts. Local threats are more amenable to management, so there is greater potential for more effective management of fishing effort and destructive methods.

Fishing has the greatest impact on species with slow demographic rates, which are typically at higher trophic levels (Jennings et al. 1999; Rochet et al. 2000; Cheung et al. 2005) and fetch higher prices (McClanahan 2010). Fishing in Kenya has consequently caused fast-growing herbivores to dominate the assemblage (e.g., green parrotfish *Leptoscarus vaigensis* and white-spotted rabbitfish *Siganus sutor*; McClanahan et al. 2008b; McClanahan and Hicks 2011; Hicks and McClanahan 2012). The result is that where fishing is unregulated, effort is higher, destructive gears dominate, and incomes are lower (McClanahan 2010). Dragging gears or trampling whilst fishing can also directly damage corals and other benthos (Mangi and Roberts 2006). Further indirect effects of fishing influence competition, settlement, and predation by altering the balance of functional groups. For example, removing higher trophic level species may reduce predation, resulting in prey release (Sandin et al. 2010). This has been most noticeable in Kenya through removal of red-lined trigger fish *Balistipus undulates*, the dominant predator of sea urchins. In areas where fishing pressure is high, such as outside protected areas, densities of red-lined trigger fish are low and urchin populations have exploded (McClanahan 2000). The implications of community shifts are unclear, however system productivity is clearly declining and climate-driven loss of charismatic fish and corals are likely to impact tourism (Cesar et al. 2002). Other stressors include sedimentation and eutrophication, which are more episodic and spatially restricted due to strong currents and tidal flushing (Obura 2001).

6.3c Management and governance

At the beginning of the last century, social norms and traditional ecological knowledge determined which rules governed resource use (Ogwang et al. 2006). Traditionally, an elder leader of a fish landing site, provided advice on seasons, who could fish and which species were prohibited, and dealt with accidents and conflicts. However, the colonial and post-colonial era drive for economic development reduced local authority resulting in an unregulated open access approach to fisheries management. The exception was a parks system established independently of fisheries management to meet the demands of a growing tourism market (McClanahan 2007; Hicks et al. 2009). The lack of fisheries regulation resulted in severe depletion of artisanal fishing stocks (McClanahan and Muthiga 1988).

Decline in fish stocks was the impetus for Kenya's fisheries management to forge a relationship between research and management, promoting research that is solution-oriented. Three institutional developments characterize these changes. First, in 1986 the coral reef conservation project (CRCP), a research non-governmental organization (NGO), was established to understand effects of human influences on Kenya's coral reefs. At its core was assessment and understanding of ecological and biophysical processes, and an appreciation of the link to exploitation and management (McClanahan and Muthiga 1988; McClanahan et al. 2005a). Secondly, in a similar trend, the early 1990s saw a change toward integrated coastal management (ICM), catalyzed by the first meeting between sectors where the interdependency of ecology and economics was recognized (McClanahan et al. 2005a).

Finally, in 2006, the fisheries department introduced legislation that devolved management to local landing sites again. This legislation enables re-emergence of the role played by elders, recognizing the importance of robust socio-ecological systems that include co-management and build upon existing traditions and institutions, enabling users to develop and enforce rules governing resource use. Recent developments under this system include proliferation of community-led, no-take, locally managed marine areas (LMMAs), similar to those developed in the Philippines (Alcala et al. 2006).

The establishment of Kenya's marine parks was through a top-down command-and-control approach. Although independent of fisheries management, it has resulted in an effectively enforced system of no-take areas along the coast, which in turn has enabled deep understanding of reef recovery dynamics. Ecosystems within protected areas have responded substantially, with fish biomass recovering to asymptotes at 1200 $kg \cdot ha^{-1}$ over periods of ~20 years (McClanahan et al. 2007b), and benthos tending away from fleshy algae toward calcifiers (McClanahan and Graham 2005; O'Leary et al. 2010). It should be noted, however, that a diversity of factors, including human population density, amount of area closed to fishing and perceived compliance levels, influence the likelihood of recovery (Pollnac et al. 2010; Daw et al. 2011).

There has been progress towards understanding complex social-ecological drivers. Investigations have found that economic development is related to reef health (Cinner et al. 2009b), and that poverty can result in fishermen being unable to reduce pressure on dwindling resources (Cinner et al. 2009a). This work has led to development of management frameworks that help fit social and environmental contexts (McClanahan et al. 2008a; Cinner et al 2012). For example, where environmental susceptibility is high and adaptive capacity is low, efforts should be made to reduce reliance on vulnerable resources (McClanahan et al. 2008a).

Kenyan fishermen use several gears that can be broadly categorized into five types: spear gun, beach seine, gill net, trap and line. Gear use and attitudes toward each differ along the coast. Beach seine and spear guns damage habitat and overlap in selectivity with most other gears (McClanahan and Mangi 2004; Mangi and Roberts 2006). Beach seines have been effectively discouraged by traditional elders in many sites for more than 20 years (McClanahan 2007). Consequently, the Kenyan fisheries department took advantage of natural differences, backed by evidence of their destructive nature, to support local restrictions and ban use of beach seines and spears in 2001 (Kenya Gazette Notice 2001, No. 7565). These developments represent co-management between government, local communities, and research organizations. Although enforcement has been spatially variable (McClanahan et al. 2005b), when effective, both area- and gear-based approaches have increased catch per unit effort of seagrass and coral reef fisheries (McClanahan et al. 2008b; McClanahan and Hicks 2010; Fig. 12.13).
[Figure 12.13]

Gear-based approaches can be further designed to improve yields, protect vulnerable life stages, and enable adaptive management in the face of climate change (Cinner et al. 2009c; Hicks and McClanahan 2012). These approaches are favorable in fisheries that support a growing population as they adapt a fishery rather than reducing it (Hilborn et al. 2004). This adaptive quality enables gear restrictions to be tailored to local socio-economic and ecological contexts (McClanahan and Cinner 2008), facilitating design that is more acceptable and less intrusive. Gears selectively target specific life history traits. Modifying the portfolio of gears can protect vulnerable traits, and mesh restrictions can protect key species past vulnerable life stages (Hicks and McClanahan 2012). Information on catch composition by different gears has recently been combined with information on vulnerability of species to population declines following coral mortality and importance of fish species in facilitating ecosystem recovery (Cinner et al. 2009c). This work has shown that spear fishers target species that are vulnerable to population crashes following extensive habitat loss. Furthermore, spear guns and traps also capture the highest proportion of species providing key ecological functions, such as herbivores and macro-invertebrate feeders (Cinner et al. 2009c). Such assessments may provide the basis for adaptive gear regulations following habitat disturbances, to both sustain catches and allow ecosystem recovery. Recently, Kenya has been experimenting with escape gaps in fish traps that allow

vulnerable species to escape (Fig. 12.13), but still maintain catch rates and profits. Similar work in the Caribbean has shown success with these approaches (Johnson 2010).
[Figure 12.13]

A recent study across the western Indian Ocean made the first attempt to identify non-linear thresholds in ecological processes and state variables associated with fishable biomass (McClanahan et al. 2011). Fishable biomass was used as the driver of these ecosystem dynamics as it is tightly linked to fishing pressure, the dominant local stressor in the region. Tipping points were identified, whereby substantial changes in variables occurred as fishable biomass declined, such as the ratio of macro algae to corals, the proportion of herbivores, or the extent of predation. Critically, this analysis enabled identification of reference points and targets for ecosystem-based management (McClanahan et al. 2011). This threshold analysis can form the basis for determining multi-species maximum sustainable yield. If fishable biomass can be maintained between 300 and 600 $kg \cdot ha^{-1}$, dual objectives of sustaining fisheries and maintaining the ecosystem above key tipping points should be achieved (McClanahan et al. 2011).

In Kenya, marine National Parks were maintaining fishable biomass well above these ecosystem-based targets, areas with gear restrictions had fishable biomass within the window, and areas with no restrictions had fishable biomass substantially below the window and below all ecosystem threshold points (McClanahan et al. 2011). This work suggests that gear-based management along Kenya's coast is proving an effective tool for ecosystem-based management and should incorporate more of the coastline.

Management developments are also evident beyond reef fisheries. Recognizing the carbon sequestration value of mangroves, a longstanding rehabilitation program run by the Kenya Marine and Fisheries Research Institute (KEMFRI) became a recent candidate for a trial in 'Blue carbon'. Blue carbon attempts to finance conservation and rehabilitation initiatives through global emissions trade and REDD+ schemes (Grimsditch et al. 2012). Elsewhere, and following a different model, Lafarge ecosystems has a restoration program in its quarries based on biodiversity objectives, and capable of responding to evaluations that use neighboring natural habitats as references (Kahumba 2009). These approaches contribute to climate change adaptation and mitigation (Kitha and Lyth 2011)

6.3d Remaining challenges

The magnitude and persistence of threats mean significant challenges must still be addressed to move Kenya further toward ecosystem-based management of coral reefs and associated coastal systems. Although there has been coordination of Kenya's fisheries management, many successes have been ad hoc and many unregulated areas remain. Consequently, science and management need to scale these efforts up, capitalizing on successes and filling both social and ecological gaps in Kenya's management mosaic. This approach should feed localized, community-specific successes into a regional adaptive strategy. It is critical that this approach equally emphasize social and ecological goals, enabling mechanisms for learning, communication, and conflict resolution. It is often users, who are intimately linked to resources that are most likely to detect and adapt to changes first. Such adaptability needs to be fostered. Adaptive mechanisms are necessary as unexpected threats emerge, presenting new challenges requiring flexible solutions. For example, increased Somali pirate activity is likely to create new social-ecological challenges, potentially undermining effective management strategies.

7. Summary and conclusions

According to many, the future of coral reefs and associated habitats is grim. These systems occupy very precise locales defined by depth, temperature, substrate, salinity, acidity, turbidity and other factors. Even subtle changes these attributes can lead to significant degradation or outright loss. Capacity for habitats to migrate elsewhere when conditions in existing locations become unsuitable is limited as development and other impacts reduce colonization sites, and increasing rates of environmental change outpace rates of dispersal, recruitment, growth and evolution. A growing human population and its demands for food and income exacerbate the strain.

Nevertheless, coral reefs are visible and valuable ecosystems, continually motivating improved management, even if the pace of action too often is slow. The case studies reviewed herein illustrate some of the more ambitious efforts toward managing reef ecosystems, importantly in three very different contexts. The case studies span three major ocean basins (Pacific, Atlantic/Caribbean, Indian). Top-down management by national government is a necessary, but not sufficient, characteristic of all three, but high level policy development and ability to effectively implement policy vary due to a complex combination of history, culture, political model, wealth and associated scientific, monitoring and enforcement capacity, competing uses, and other factors. Despite those differences, all three rely, or are beginning to rely, on smaller scale governance and incentives to complement high level authority and offset its limitations. Spatial management aimed at balancing competing uses, restoring ecosystem function, improving science, and buffering against uncertainty is a central strategy in all three cases. However, spatial management will not be fully effective in isolation.

The case studies show how integration of MPAs with other management tools can increase effectiveness. Australia has paired sophisticated zoning with incentive-based fisheries management in the form of individual transferable quotas (ITQs) to more effectively manage the most heavily exploited stocks. This represents a form of management devolved to the individual level, as ITQs empower fishermen to make more decisions regarding when and how to fish than under inefficiencies forced by input controls (Costello et al. 2008; Worm et al. 2009; Grimm et al. 2012). Similarly, Kenya has devolved important aspects of management to smaller scales, although the key units are local villages and their traditional social structures. Collective management can be effective, especially where it matches finer ecological scales, and builds upon existing social and cultural systems (Prince 2003; Ostrom 2010; Gutierrez et al. 2011). Certain outcomes of Kenya’s management experience illustrate that effectiveness, although key challenges remain. In contrast to Australia and Kenya, Cuba has historically relied more on top-down than smaller scale approaches. However, Cuba’s planning processes for multi-user management zones and recent authorization of fishing cooperatives illustrate how it too is increasingly turning to management at smaller scales.

One important difference between Australia and the other case studies is use of output controls alongside spatial management and multi-scale governance and incentive structures. Output-based management, especially when combined with incentive-based allocation systems, has been shown to be more effective at achieving target biomass, fishing within quotas, and reducing variance in biomass, exploitation rate and catch (Worm et al. 2009; Essington 2010; Grimm et al. 2012; Melnychuk et al. 2012). Of course, scientific, monitoring and enforcement capacity needed to effectively set and implement catch limits are often lacking, which, combined with lack of acceptance for output controls by key stakeholders, can be prohibitive. In such cases, input controls might remain a necessary and reasonably effective strategy, as evidenced by Kenya, until increased capacity and outreach change the logistical and political conditions in favor of output controls. Notably, the work of McClanahan et al. (2011) has made assessment and management of reef fisheries on a multi-species ecosystem level possible. Should Kenya formalize those metrics as part of its management strategy, it will be a rare instance of such an approach pioneered by a low capacity developing nation.

Fishing has been a focus of this discussion because it is one of the most significant uses, and impacts. Also, of the three most valuable services provided by coral reefs (i.e., alongside tourism and coastal protection), it is arguably the least effectively managed in most places, which has implications for other services. However, other uses and impacts still need increased attention in the cases examined here, and just about every other reef ecosystem. Although results have not been adequately measured, and benefits have not yet been fully realized due to the efforts beginning in earnest only within the last decade, Cuba has perhaps demonstrated the most ambitious policies toward integrated management of uses of our three case studies. This is done through two primary vehicles. First, Cuba is establishing marine protected areas in concert with those along the coast, on land, in wetlands, and in freshwater in a rare example of a truly integrated protected area network. This not only combines management of coral reefs with associated coastal habitats (mangroves, seagrass beds) but also with freshwater and terrestrial systems. Second, within marine waters, Cuba's multi-user planning zones (ZBRMIC) are now attempting to plan and coordinate ocean uses in ways that promote equity and maximize effectiveness. More detailed scientific and policy analyses of Cuba's approaches should be a priority to determine their effectiveness and value as a model for other locales.

Daunting challenges continue to confront coral reefs, mangroves and seagrasses. Pessimism is pervasive among many, and not unjustifiably. However, encouraging developments continue to emerge in both developed nations, such as the United States' creation of the 362,073 km^2 Papahānaumokuākea Marine National Monument in the Northwest Hawaiian Islands, and smaller developing nations, such as Palau's policy to place 20% of its land and 30% of its marine waters within protected areas. Increasing awareness and implementation of ecosystem-based management continues as science progresses and degradation unfolds. Whether and to what extent science and management will keep pace with ecosystem change is unclear. We are likely to see reduced area and productivity of coral reefs, mangroves and seagrasses. However, these systems are able to deliver a wide range of goods and services, even in a diminished state. The biggest question is how much value will be lost due to uncertain future trends in the ecosystem and management responses.

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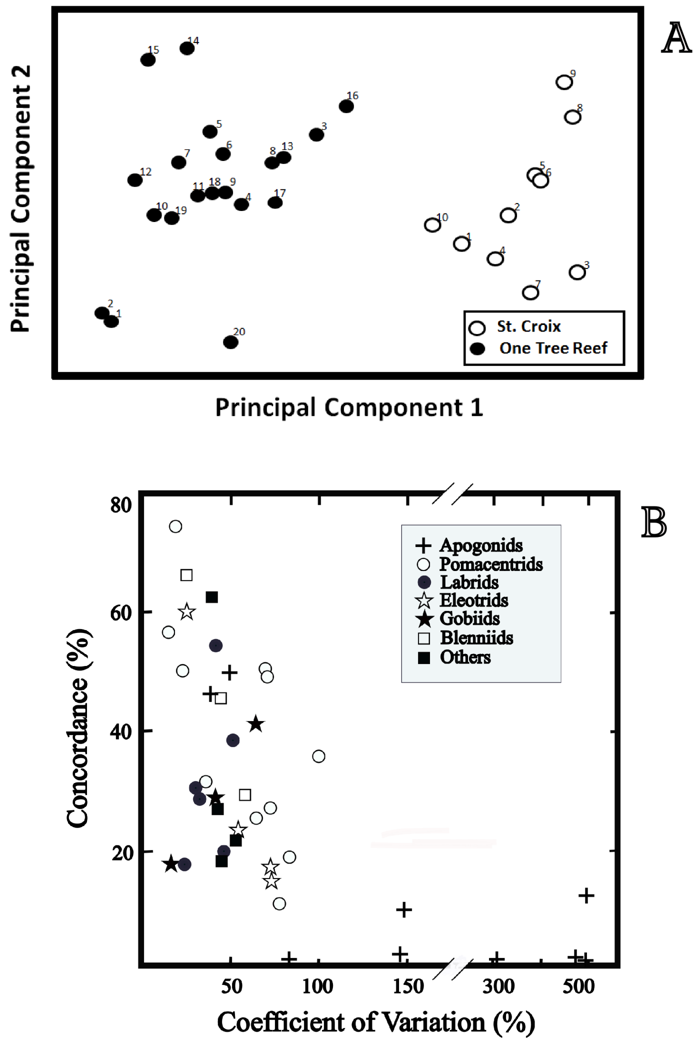

Figure 12.1. The high temporal variability in structure of reef fish assemblages is illustrated in two different ways. Principal components analysis displays changing structure of assemblages across 10 successive censuses over 5 years at St. Croix, U.S.V.I., and across 20 successive censuses over 10 years at One Tree Reef, Australia (A). Data are numbers of fish in each of 11 trophic guilds. There is a clear separation (different structure) between the two sites, but equally substantial changes through time within each site. Underlying reasons for the varying structure using the same One Tree Reef data are revealed by a plot including the 40 most common species across the 20 censuses (B). Each species is located in a space defined by concordance in pattern of distribution across patch reefs as measured by Kendall's W, and the extent to which they varied in abundance across censuses as measured by the coefficient of variation. Few species are located in the upper left quadrant representing those that are consistently present and abundant. (A redrawn from Chittaro and Sale 2003; B redrawn from Sale 1991).

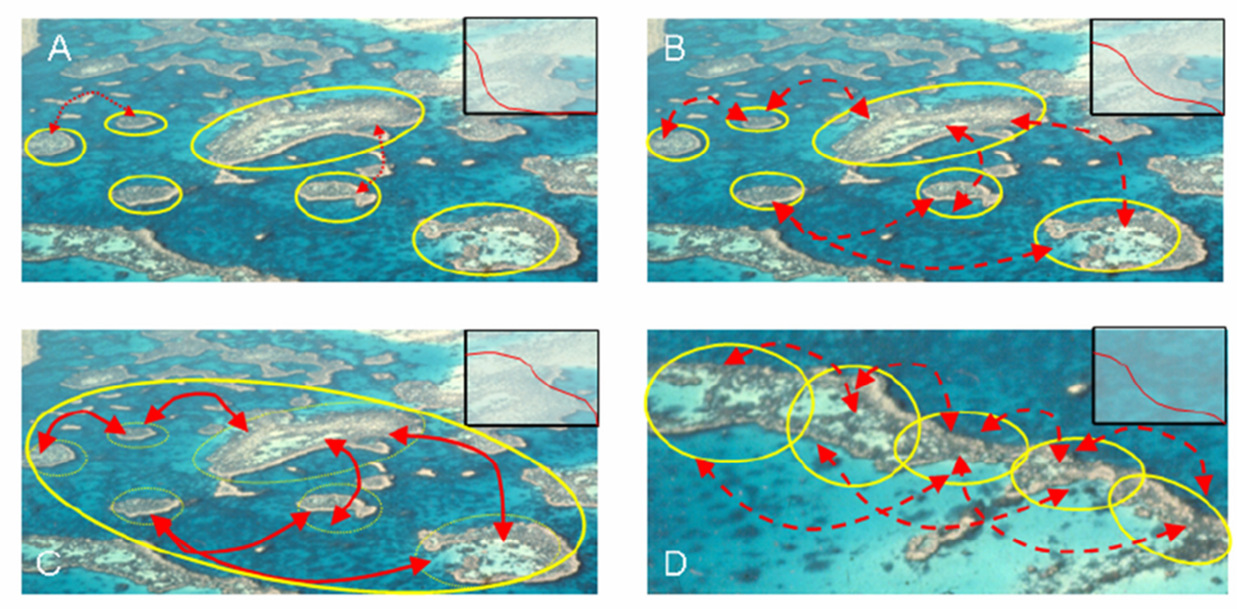

Figure 12.2. Each panel shows a patchy array of reef habitat, some of which is occupied by a reef fish species (ovals = local aggregations of fish). Dispersal (chiefly by larvae) among sites is shown by arrows, graded to show slight (A), moderate (B, D) or extensive exchange (C). Mean scale of dispersal is shown as a graph of proportion of larvae (y axis) against distance from source (x axis) in upper right corner of each panel – mean dispersal distance is least in A, intermediate and identical in B and D, and greatest in C. Cases A, B, and C differ only in the scale of dispersal relative to the scale of patchiness of habitat, yet yield essentially independent local populations (A), a metapopulation (B) in which local populations are sufficiently connected by dispersal for some interaction, and a single, but subdivided, population (C), occupying a number of patches of habitat. Case D is typical of regions where coral reef habitat is more contiguous, yet the spatially explicit mating pattern and scale of larval dispersal still provide a functional metapopulation even though patch structure is primarily an analytical construct. (Reprinted from Sale and Kritzer 2008)

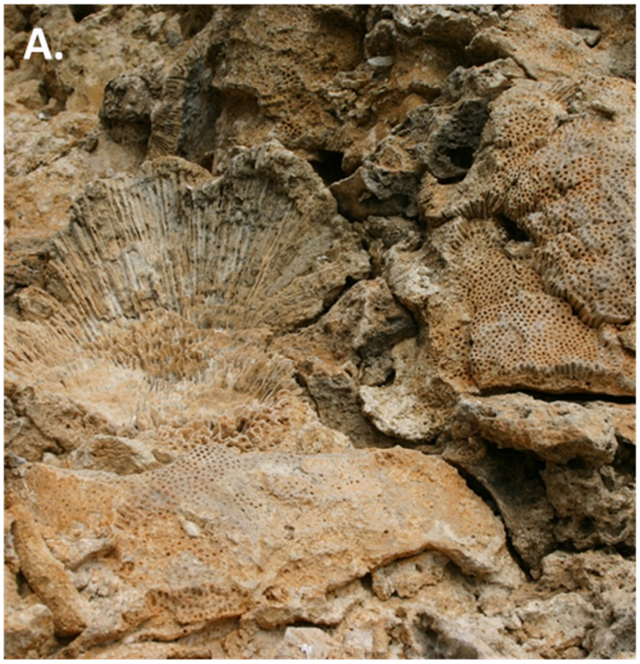

Figure 12.3. Fossil corals embedded in a matrix of compressed biogenic sand along the coast of Curacao, Netherlands Antilles (A). The bell tower of the Cathedral of Havana, Cuba (B), one of many colonial structures in Old Havana and around the Caribbean constructed of coral limestone. (Photos by Jake Kritzer)

Figure 12.4. Map of the Great Barrier Reef, Australia, showing its extent, its archipelagic topography, and the boundaries of the Great Barrier Reef Marine Park.

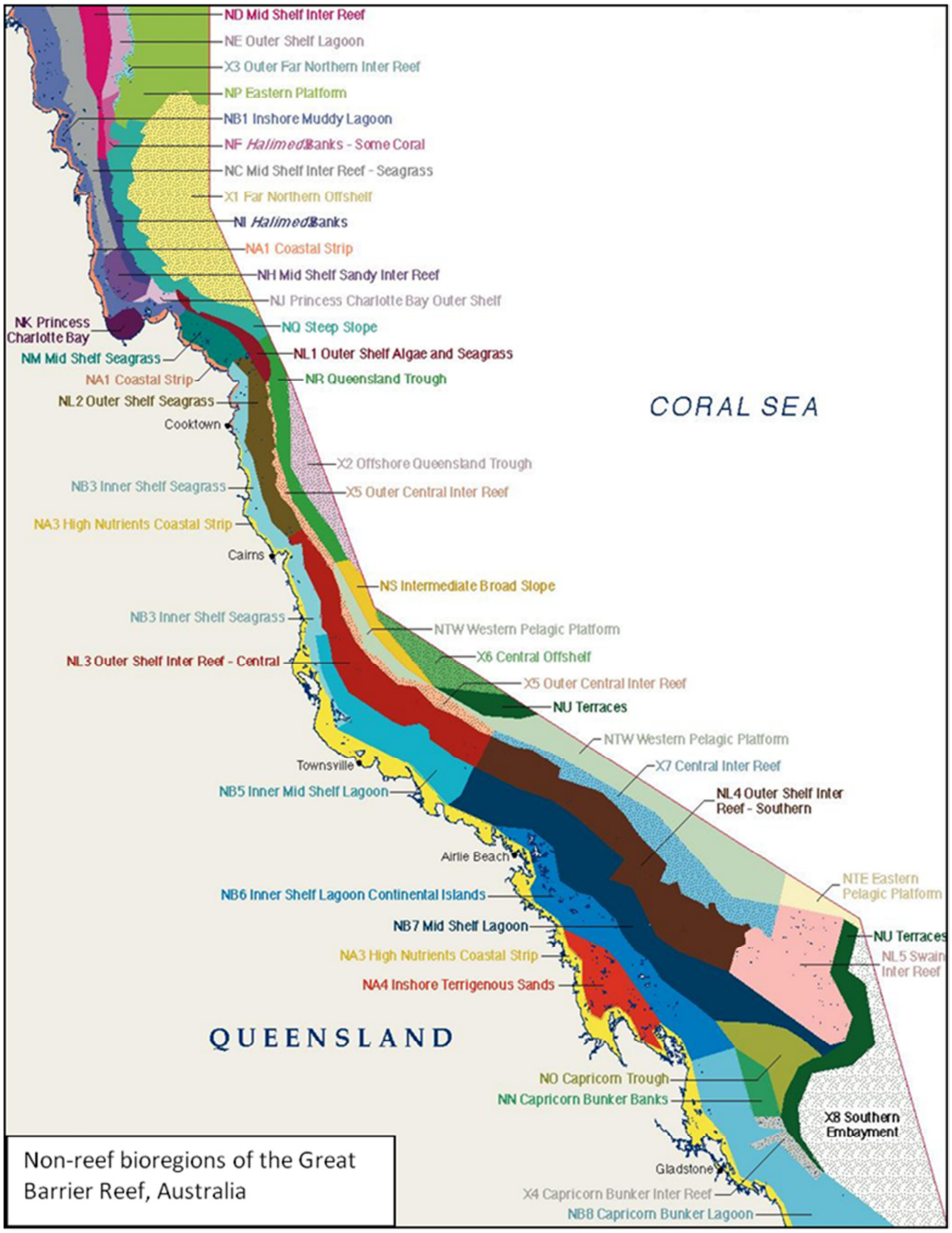

Figure 12.5. Map of non-reef benthic bioregions identified within the Great Barrier Reef Marine Park in preparation for rezoning of the multiple use park in 2004. Seventy reef and non-reef bioregions were identified on the basis of over 40 biophysical datasets and expert opinion and zoning plans were designed to ensure at least 20% of each bioregion was included in multiple no-take areas over the marine park.

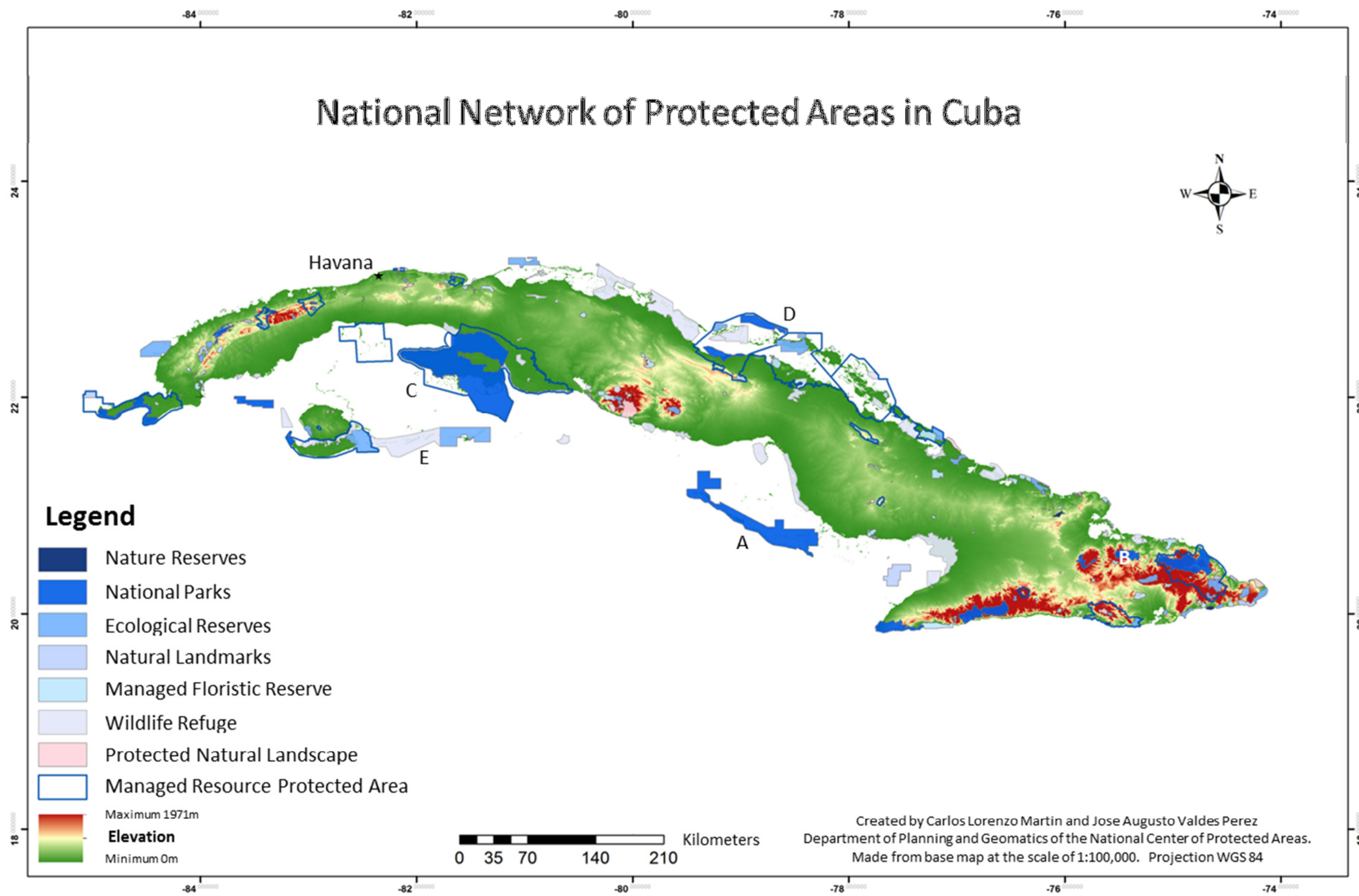

Figure 12.6. Marine, coastal, freshwater and terrestrial protected areas in Cuba. Areas discussed in the text include (in the order referenced): Jardines de Reina (Gardens of the Queen) National Park (A), Sierra Cristal National Park (B), Cienéga de Zapata (Zapata Swamp) National Park (C), Sabana-Camagüey Archipelago (containing multiple protected areas of different designatons; D), and Canarreos Archipelago (also containing multiple designations; E).

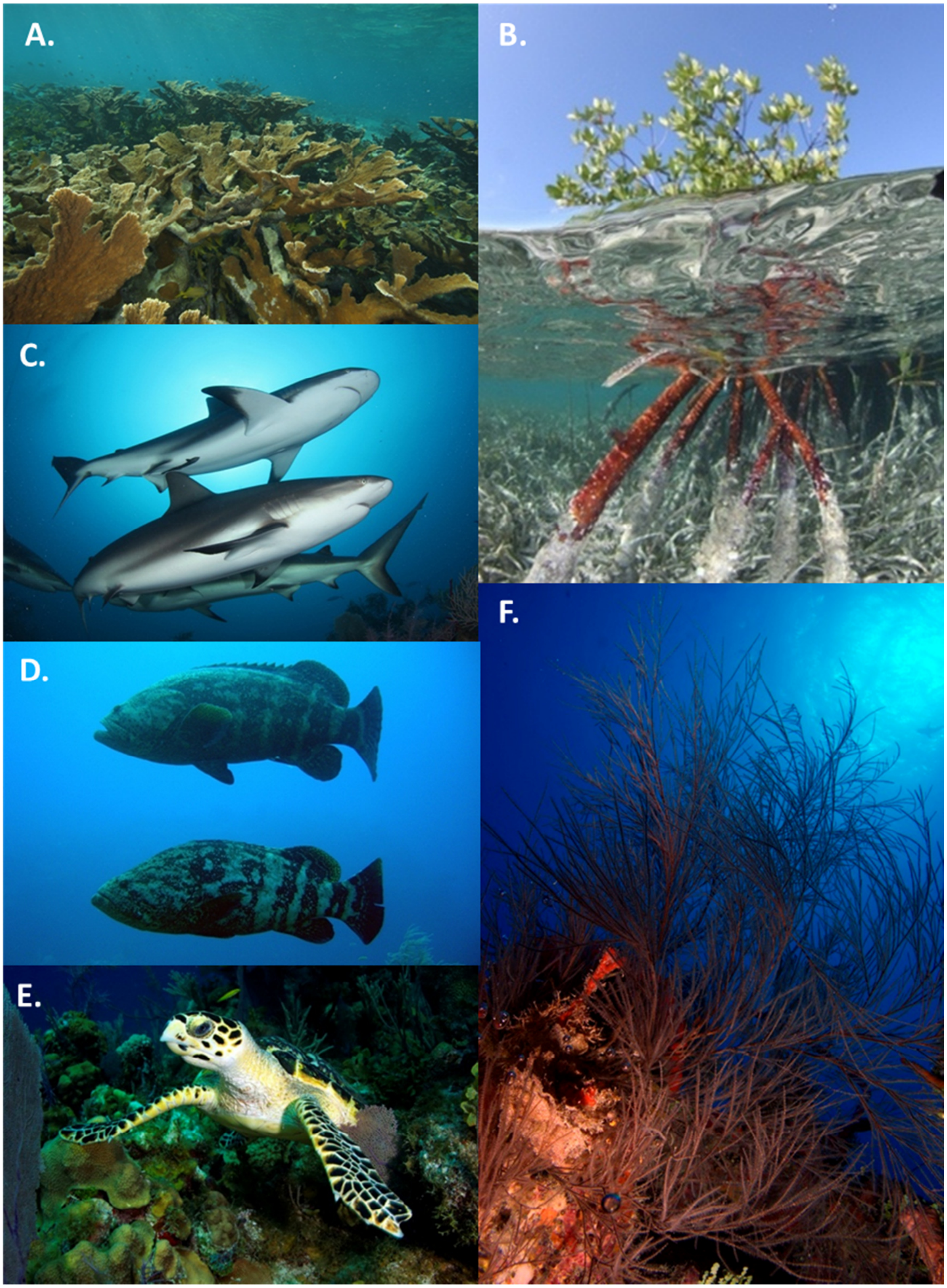

Figure 12.7. Habitat and species diversity in the Jardines de Reina (Gardens of the Queen) National Park, Cuba, including elkhorn coral reef crest (A), red mangrove sapling fringed by eelgrass (B), Caribbean reef sharks (C), Goliath groupers (D), hawksbill turtle (E), and black coral(F). (Photos A-D by Noel Lopez, and E-F by Fausto de Nevi)

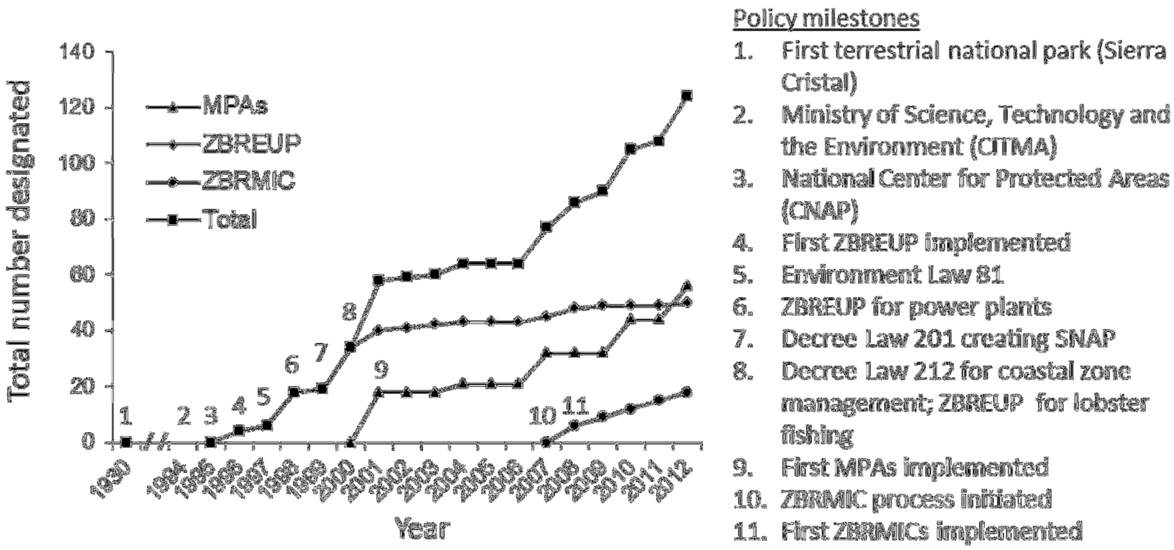

Figure 12.8. History of implementation of Marine Protected Areas (MPAs), Zones under Special Regimes of Use and Protection (ZBREUP), and Zones under Regimes of Integrated Coastal Management (ZBRMIC), with timing of major policy milestones. See text for descriptions of the different types of managed areas and policies.

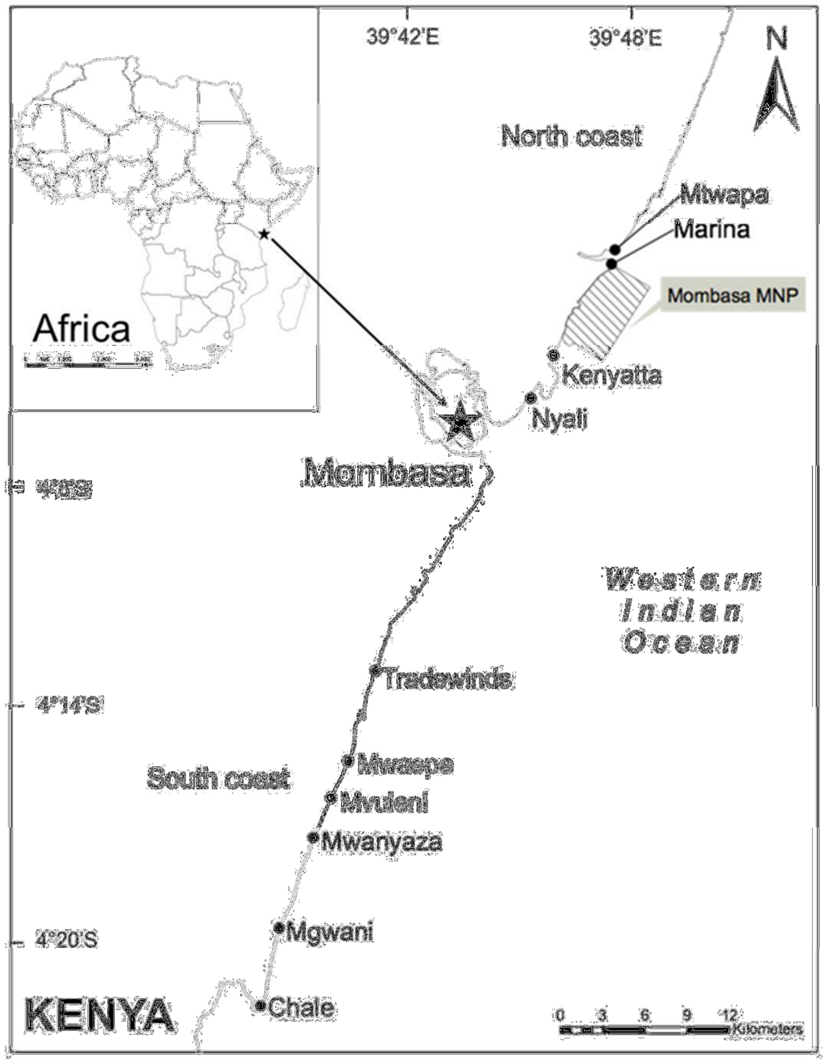

Figure 12.9. Detail of the Kenyan coast centered on the port city of Mombasa. Shipping is an important ecosystem service for Mombasa and the nation as a whole. Outside of the well-protected harbor, fishing, tourism and shoreline protection are important services delivered by the fringing reef adjacent to the many coastal towns. Marine protected areas, such as Mombasa Marine National Park, have been an important component of management in this developing nation. (reprinted from McClanahan et al 2008b)

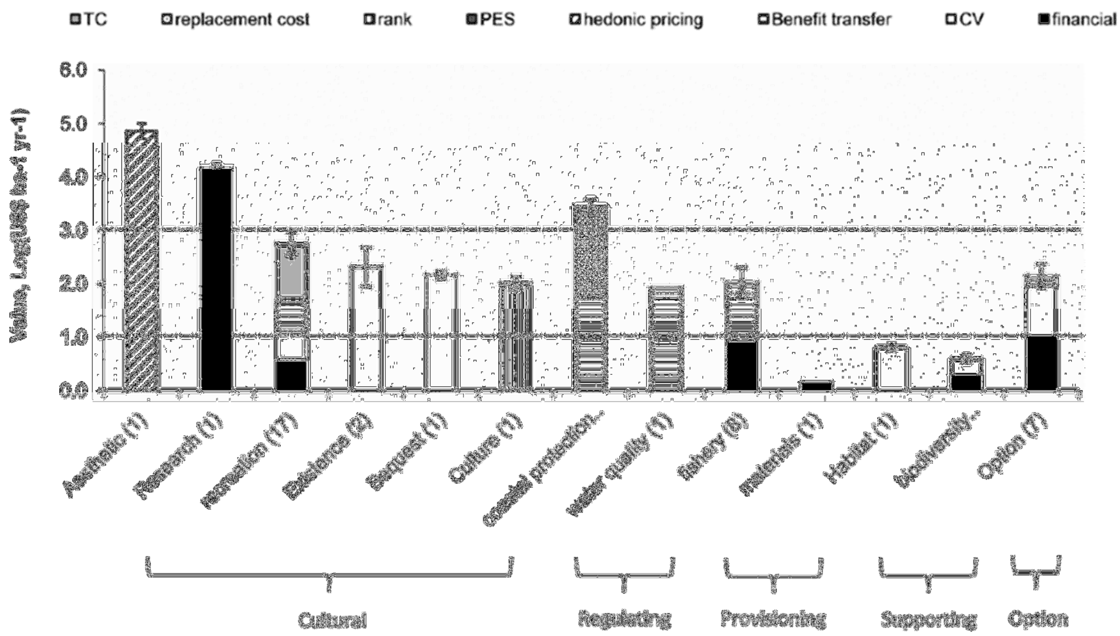

Figure 12.10. Estimated value of ecosystem services in the western Indian Ocean. Services are ordered according to categories from the Millennium Ecosystem Assessment. (reprinted from Hicks 2011)

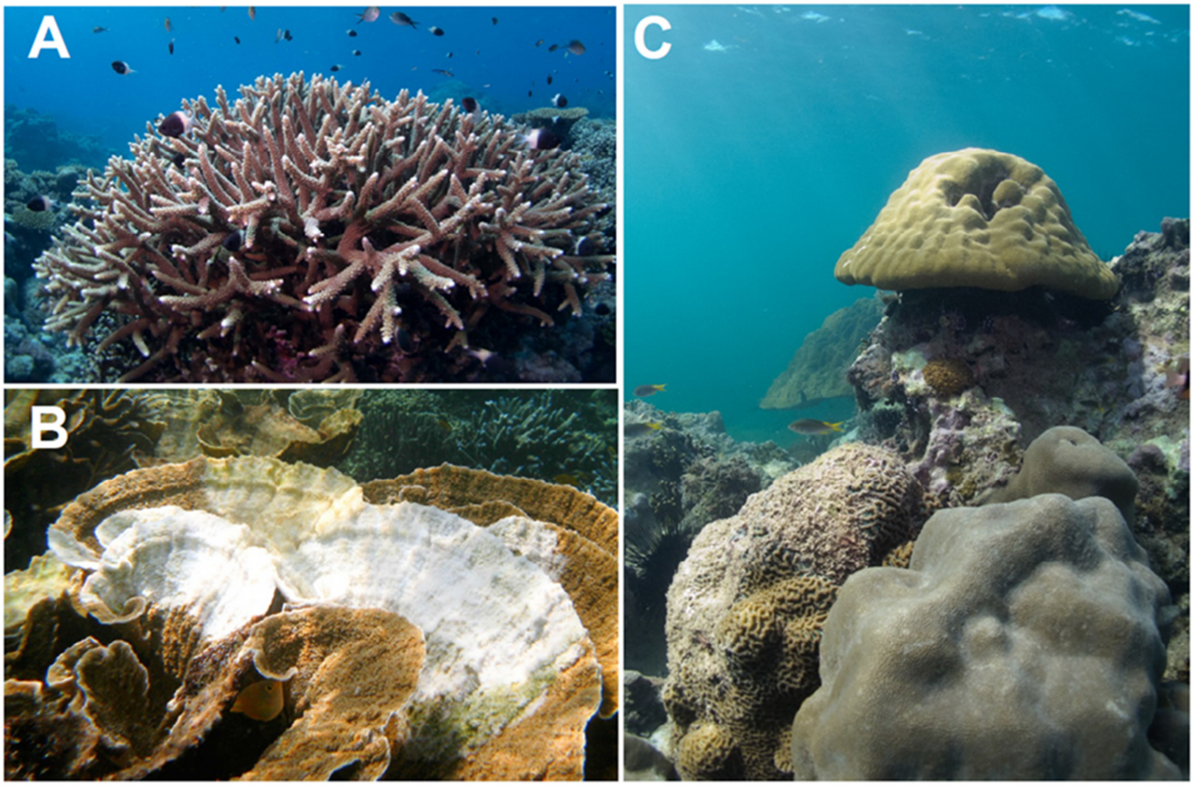

Figure 12.11. Differential responses to thermal stress and bleaching among coral species and morphologies in Kenya. Bleaching-sensitive corals include species of branching *Acropora* (A) and plating *Montipora* (B), while species of massive *Porites* (C) are often less affected by bleaching. (Photos by Emily Darling)

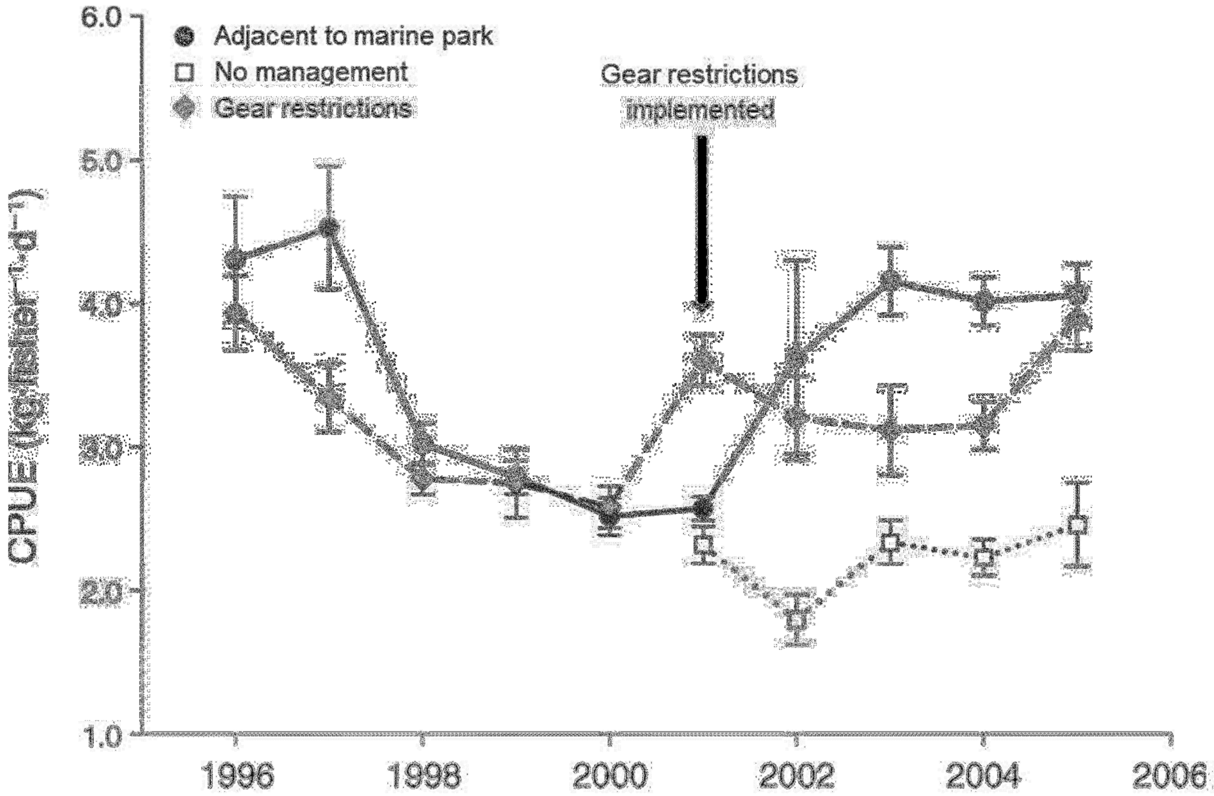

Figure 12.12. Trends in catch per unit effort (CPUE) from Kenya's artisanal fishery. Areas with gear restrictions achieved greater CPUE relative to areas without management. The highest CPUE was achieved in areas adjacent to a no-take marine protected area, but only after gear restrictions were also introduced. (reprinted from McClanahan et al 2008b)

Figure 12.13. Fish trap typical of those used by Kenya's artisanal fishermen (A), with an escape gap that allows bycatch species to escape (B). (Photos by Tim McClanahan)