8 Marine Protected Areas

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8.1 Background and History

8.1.1 Introduction

Coastal oceans and marine ecosystems are central to the lives and livelihoods of a large and growing proportion of the U.S. population. They provide extensive areas for recreation and tourism and support productive fisheries. Some areas produce significant quantities of oil and gas, and commercial shipping crosses coastal waters. In addition, coral reefs and barrier islands provide coastal communities with some protection from storm-generated waves. In their global analysis of the value of ecosystem services, Costanza *et al.* (1997) estimated that the value of coastal marine ecosystem services was over one-third of all terrestrial and marine ecosystem services combined ($12.5 of $33 trillion). Despite their value, coastal ecosystems and the services they provide are becoming increasingly vulnerable to human pressures, and management of coastal resources and human impacts generally is insufficient or ineffective (Millennium Ecosystem Assessment, 2005).

As a result of coastal and shore-based human activities, marine ecosystems are exposed to a long list of threats and stressors, including overexploitation of living marine resources, pollution, redistribution of sediments, and habitat damage and destruction. There is an equally long list of regulatory responses, including management of fisheries for sustainability, restricting ocean dumping, reducing loads of nutrients and contaminants, controlling dredge-and-fill operations, managing vessel traffic to reduce large-vessel groundings, and so on. These regulations are managed by coastal states and the federal government, with state jurisdiction extending three nautical miles (nm) offshore (9 nm in the Gulf of Mexico) and federal waters (the U.S. Exclusive Economic Zone, or U.S. EEZ) on out to 200 nm or the edge of the continental shelf. The total area of the U.S. EEZ exceeds the total landmass of the coterminous United States by about one-half (Pew Ocean Commission, 2003).

Broad-scale protections in the U.S. EEZ cover a wide range of types of marine ecosystems, from low to high latitudes and across the Atlantic and Pacific Oceans. Shallow areas of these systems share basic features in the form of biologically generated habitats: temperate kelp forests and salt marshes, tropical coral reefs and mangroves, and seagrass beds throughout. These habitats are fundamental to ecosystem structure and function and support a range of different community types (Bertness, Gaines, and Hay, 2001). In addition, there are significant deep-water coral formations about which we are just starting to increase our understanding (Rogers, 1999; Watling and Risk, 2002).

Embedded within the general protections of the U.S. EEZ are hundreds of federal marine protected areas (MPAs) that are designed to provide place-based management at “special” places (Barr, 2004) and other areas that have been identified as meriting protective actions. The term “marine protected area” has been used in many ways (*e.g.*, Kelleher, Bleakley, and Wells, 1995; Agardy, 1997; Palumbi, 2001; National Research Council, 2001; Agardy *et al.*, 2003). We use the following definition: “Marine protected area” means any area of the marine environment that has been reserved by federal, state, territorial, tribal, or local laws or regulations to provide lasting protection for part or all of the natural and cultural resources therein (Executive Order

13158, quoted in National Center for Marine Protected Areas, 2006). It is important to emphasize at the onset that MPAs are managed across a wide range of approaches and degrees of protection (Wooninck and Bertrand, 2004; National Center for Marine Protected Areas, 2006). At the highly protective end of the spectrum are fully protected (no-take) marine reserves (Sobel and Dahlgren, 2004). These reserves eliminate fishing and other forms of resource extraction and enable some degree of recovery of exploited populations and restoration of ecosystem structure and function, generally within relatively small areas. It is also important to highlight at the onset that management of waters surrounding MPAs is critically important both to the effectiveness of the MPAs themselves as well as to the overall resilience of larger marine systems.

Federal MPAs have been established by the Department of the Interior (National Park Service and U.S. Fish and Wildlife Service) and the Department of Commerce, National Oceanic and Atmospheric Administration (National Marine Fisheries Service, National Estuarine Research Reserve System, and National Marine Sanctuary Program) (Table 8.1). A 2000 executive order established the National Center for Marine Protected Areas (http://mpa.gov/) to strengthen and expand a national system of MPAs. The total area of MPAs within the U.S. EEZ is miniscule, and an even smaller area lies within fully protected marine reserves (Table 8.2). Only 3.4% of the U.S. EEZ lies within fully protected marine reserves, with most of this area due to the 2006 Presidential proclamation that designated the Papahānaumokuākea (Northwestern Hawaiian Islands) Marine National Monument; excluding the Monument reduces the percentage to 0.05%.

Manifestations of climate change are strengthening (IPCC, 2007b) against a background of long-standing alterations to ecological structure and function of marine ecosystems caused by fisheries exploitation, pollution, habitat degradation and destruction, and other factors (Pauly *et al.*, 1998; Jackson *et al.*, 2001; Pew Ocean Commission, 2003; U.S. Commission on Ocean Policy, 2004). Nowhere is the stress of elevated sea surface temperatures more dramatically expressed than in coral reefs, where local-scale coral bleaching has occurred in the Eastern Pacific and Florida for more than two decades (Glynn, 1991; Causey, 2001; Obura, Causey, and Church, 2006). Impacts of climate variability and change in temperate ecosystems have not been as dramatic as coral bleaching. Interestingly, the combined effects of climate change, regime shifts, and El Niño-Southern Oscillation events (ENSOs) can strongly affect kelp forests (Paine, Tegner, and Johnson, 1998; Steneck *et al.*, 2002), but apparently not associated communities (Halpern and Cottenie, 2007).

The purpose of this chapter is to examine adaptation options for marine protected areas in the context of climate change. We will focus on the 14 MPAs that compose the National Marine Sanctuary Program (Table 8.3, Fig. 8.1) because they encompass a wide range of ecosystem types and are the only U.S. MPAs managed under specific enabling legislation (U.S. Congress, 2007). The National Marine Sanctuary Program has explicit approaches to and goals of MPA management, which simplify discussion of existing MPA management and how it may be adapted to climate change.

Figure 8.1. Locations of the 14 MPAs that compose the National Marine Sanctuary System (National Marine Sanctuary Program, 2006c).

The chapter provides background information about the historical context and origins of MPAs, with National Marine Sanctuaries highlighted as an example of effectively managed MPAs (Kelleher, Bleakley, and Wells, 1995; Agardy, 1997). MPAs are managed by several federal organizations other than the National Oceanic and Atmospheric Administration (NOAA) (Table 8.1), but it is beyond the scope of this chapter to cover all entities. National Marine Sanctuaries were selected to illustrate adaptation options for MPAs that apply broadly with respect to major anthropogenic and climate change stressors.

It is also beyond the scope of this chapter to cover issues concerning marine ecosystems from tropical to polar climates. This chapter highlights coral reef ecosystems, which have already shown widespread and dramatic responses to oceanic warming and additional global and local stressors. Mass coral reef bleaching events became worldwide in 1998 and have resulted in extensive mortality of reef-building corals (Wilkinson, 1998; 2000; 2002; Turgeon *et al.*, 2002; Wilkinson, 2004; Wadell, 2005). There now exists a substantial and rapidly growing body of research on impacts of climate change on corals (such as bleaching) and coral reef ecosystems (*e.g.*, Smith and Buddemeier, 1992; Glynn, 1993; Hoegh-Guldberg, 1999; Wilkinson, 2004; Buddemeier, Kleypas, and Aronson, 2004; Donner *et al.*, 2005; Phinney *et al.*, 2006; Berkelmans and van Oppen, 2006). Climate change stressors including effects of ocean acidification on carbonate chemistry (Kleypas *et al.*, 1999; Soto, 2001; The Royal Society, 2005; Caldeira and Wickett, 2005) will be reviewed later in this chapter. Management approaches to coral reef ecosystems in response to mass bleaching and/or climate change have also received some attention (*e.g.*, Salm and Coles, 2001; Hughes *et al.*, 2003; Hansen, Biringer, and Hoffman, 2003; West and Salm, 2003; Bellwood *et al.*, 2004; Wooldridge *et al.*, 2005; Marshall and Schuttenberg, 2006a; 2006b).

Climate-change stressors in and ecological responses of colder-water marine ecosystems only partially overlap those of warmer-water and tropical marine ecosystems (McCarthy *et al.*, 2001; Kennedy *et al.*, 2002). The Channel Islands National Marine Sanctuary is included as a temperate-zone case study to contrast with case studies of tropical coral reef ecosystems from the Florida Keys to Hawaii to Australia, which differ in extent of no-take protection.

8.1.2 Historical Context and Origins of National Marine Sanctuaries and Other Types of Marine Protected Areas

8.1.2.1 Mounting Environmental Concerns and Congressional Actions

In 1972 the United States acknowledged the dangers and threats of uncontrolled industrial and urban growth and their impacts on coastal and marine habitats through the passage of a number of Congressional acts that focused on conservation of threatened coastal and ocean resources. The Water Pollution Control Act addressed the nation’s threatened water supply and coastal pollution. The Marine Mammal Protection Act imposed a five-year ban on killing whales, seals, sea otters, manatees, and other marine mammals. The Coastal Zone Management Act provided a framework for federal funding of state coastal zone management plans that created a nationwide system of estuarine reserves. A final environmental bill that focused on ocean health, the Marine Protection, Research and Sanctuaries Act of 1972, established a system of marine protected areas —National Marine Sanctuaries (NMS)—administered by NOAA (Fig. 8.2).

Figure 8.2. Timeline of the designation of the national marine sanctuaries in the National Marine Sanctuary Program (National Marine Sanctuary Program, 2006a).

8.1.2.2 Types of Federal MPAs and Focus on National Marine Sanctuaries

In addition to the 13 national marine sanctuaries and one marine national monument, there are hundreds of marine managed areas (MMAs) under other, sometimes overlapping jurisdictions (Table 8.2) (National Research Council, 2001; National Center for Marine Protected Areas, 2006). The National Park System, administered by the National Park Service of the Department of the Interior, includes more than 70 ocean sites (Davis, 2004). Certain national parks such as Everglades (founded in 1947), Biscayne (founded in 1968 as Biscayne National Monument), and Dry Tortugas National Parks (founded in 1935 as Fort Jefferson National Monument) have much longer histories of functioning as MPAs than the 35-year history of National Marine Sanctuaries. The National Marine Sanctuary Program and National Park Service have collaborated on ocean stewardship for a number of years (Barr, 2004). The U.S. Fish and Wildlife Service, also under the Department of the Interior, manages more than 100 national wildlife refuges that include marine ecosystems (Table 8.2). In some cases, jurisdictions overlap. For example, there are four national wildlife refuges within the Florida Keys National Marine Sanctuary (Keller and Causey, 2005), three of which cover large areas of nearshore waters (Fig. 8.3).

Figure 8.3. Map of the Florida Keys National Marine Sanctuary. The 1990 designation did not include the Tortugas Ecological Reserve located at the western end of the sanctuary, which was implemented in 2001. The Key Largo NMS corresponded to the Existing Management Area (EMA) just offshore of the John Pennekamp Coral Reef State Park; the Looe Key NMS corresponded to the EMA surrounding the Looe Key Sanctuary Preservation Area and Research Only Area (National Oceanic and Atmospheric Administration, 2007d).

NOAA's National Marine Fisheries Service has jurisdiction over a large number of fishery management areas (Table 8.2). Collectively, these areas are more than an order of magnitude greater in size than all the other MMAs combined, but with a very small area under no-take protection (Table 8.2). NOAA also administers the National Estuarine Research Reserve System, which is a partnership program with coastal states that includes 27 sites.

This chapter is focused on NOAA's National Marine Sanctuary Program (NMSP), because it is dedicated to place-based protection and management of marine resources at nationally significant locations and has gained international recognition over the years (Barr, 2004) (Fig. 8.4). The principles of adaptation of MPA management to climate change (*i.e.*, institutional responses) that are identified will be broadly applicable to MPAs under other jurisdictions and forms of management, though institutional responses to adaptation likely will differ among agencies responsible for resource management (Holling, 1995; McClanahan, Polunin, and Done, 2002). As the only federal program for the management of MPAs, the NMSP is in a unique position to respond to challenges and recommendations in reports by the U.S. Commission on Ocean Policy (U.S. Commission on Ocean Policy, 2004) and Pew Oceans Commission (Pew Ocean Commission, 2003). Both reports encourage the use of ecosystem-based management, which is one of the hallmarks of the NMSP.

Figure 8.4. Organizational chart of the National Marine Sanctuary Program (NOAA National Ocean Service, 2006).

8.1.2.3 The National Marine Sanctuary Program

The NMSP was established to identify, designate, and manage ocean, coastal, and Great Lakes resources of special national significance to protect their ecological and cultural integrity for the use and enjoyment of current and future generations. In addition to natural resources within national marine sanctuaries, NOAA's Maritime Heritage Program is committed to preserving historical, cultural, and archaeological resources (National Marine Sanctuary Program, 2006b).

The inclusion of consumptive human activities as a major part of the management programs in national marine sanctuaries distinguishes them from other federal or state resource protection programs. Sanctuaries are established for the long-term public benefit, use, and enjoyment, both recreationally and commercially. However, it is critical that sanctuary management policies, practices, and initiatives ensure that human activities in sanctuaries are compatible with long-term protection of sanctuary resources.

Thirteen national marine sanctuaries and one marine national monument, representing a wide variety of ocean environments as well as one cultural heritage site in the Great Lakes, have been established since 1975 (Table 8.3; Fig. 8.1). The national marine sanctuaries encompass a wide range of temperate and tropical environments: moderately deep banks, coral reef-seagrass-mangrove systems, whale migration corridors, deep sea canyons, and underwater archaeological sites. The sites range in size from 0.66 km^2 in Fagatele Bay, American Samoa, to more than 360,000 km^2 in the Northwestern Hawaiian Islands (Table 8.3), the largest marine protected area in the world.

The NMSP has implemented a regional approach to managing the system of sanctuaries (National Marine Sanctuary Program, 2006c). Four regions have been established to improve support for the sites and to enhance an integrated ecosystem-based approach to management of sanctuaries. An important function of the regions is to provide value-added services to the sites, while taking a broader integrated approach to management. The four regions are the Pacific Islands; West Coast; Northeast-Great Lakes; and the Southeast Atlantic, Gulf of Mexico, and Caribbean. Boundaries for these regions are focused on physical and biological connectivity among sites and not on political boundaries.

8.1.3 Enabling Legislation

8.1.3.1 Enabling Legislation for Different Types of MPAs

The U.S. National Park System Organic Act established the National Parks System in 1916. Several parks and national monuments have marine waters within their boundaries or are primarily marine; they were the earliest federal MPAs. Similarly, a large number of national wildlife refuges function as MPAs (Table 8.1) under the authority of the U.S. Fish and Wildlife Service. The 1966 National Wildlife Refuge System Administration Act was the first comprehensive legislation after decades of designations of federal wildlife reservations and refuges (U.S. Fish and Wildlife Service, 2007).

NOAA's National Marine Fisheries Service implements and manages more than 200 fishery management areas (Table 8.1) under several different statutory authorities, with four major categories: Federal Fisheries Management Zones, Federal Fisheries Habitat Conservation Zones, Federal Threatened and Endangered Species Protected Areas, and Federal Marine Mammal Protected Areas (National Center for Marine Protected Areas, 2006). The purposes of these fishery management areas include rebuilding and maintaining sustainable fisheries, conserving and restoring marine habitats, and promoting the recovery of protected species. NOAA's National Estuarine Research Reserve System was established by the Coastal Zone Management Act of 1972 (U.S. Congress, 1972a). This system consists of partnerships between NOAA and coastal states to protect habitat, offer educational opportunities, and provide areas for research. At this time Congress also established a system of national marine sanctuaries.

8.1.3.2 The Marine Protection, Research and Sanctuaries Act

The Marine Protection, Research, and Sanctuaries Act (1972b) established both the NMSP and a regulatory framework for ocean dumping, which was a major issue at the time. In Title III of the Act, later to be known as the National Marine Sanctuaries Act (NMSA), the Secretary of Commerce received the authority to designate national marine sanctuaries for the purpose of preserving or restoring nationally significant areas for their conservation, recreational, ecological, or esthetic values. The NMSA is reauthorized every four to five years, allowing for updating and adaptation as necessary.

8.1.3.3 Legislation Designating Particular National Marine Sanctuaries

On November 16, 1990, the Florida Keys National Marine Sanctuary and Protection Act (FKNMS Act), P.L. 101-605, set out as a note to 16 U.S.C. 1433, became law. The FKNMS Act designated an area of waters and submerged lands, including the living and nonliving resources within those waters, surrounding most of the Florida Keys (Fig. 8.3). This was the first national marine sanctuary to be designated by an act of Congress.

The FKNMS Act immediately addressed two major concerns of the residents of the Florida Keys. First, it placed an instant prohibition on oil drilling, including mineral and hydrocarbon leasing, exploration, development, or production, within the sanctuary. Second, the Act created an internationally recognized area to be avoided (ATBA) for ships greater than 50 m in length, with special designated access corridors into ports (Fig. 8.3). The ATBA provides a buffer zone along the coral reef tract to protect it from oil spills and groundings by large vessels.

The FKNMS Act also called for a comprehensive, long-term strategy to protect and preserve the Florida Keys marine environment. The sanctuary seeks to protect marine resources by educating and interpreting for the public the Florida Keys marine environment, and by managing those uses that result in resource degradation. The greatest challenge to protecting the natural resources of the Keys and the economy they support is preserving water quality. To address this challenge, the FKNMS Act brought together various agencies to develop a comprehensive Water Quality Protection Program (WQPP). The U.S. Environmental Protection Agency (EPA) is the lead agency in developing and implementing the WQPP, the purpose of which is to "recommend priority corrective actions and compliance schedules addressing point and nonpoint sources of pollution to restore and maintain the chemical, physical, and biological integrity of the sanctuary, including restoration and maintenance of a balanced, indigenous population of corals, shellfish, fish, and wildlife, and recreational activities in and on the water" (U.S. Department of Commerce, 1996).

The FKNMS Act called for an Interagency Core Group to be established to compile management issues confronting the sanctuary as identified by the public at scoping meetings, from written comments, and from surveys distributed by NOAA. The Core Group consisted of representatives from several divisions of NOAA, National Park Service, U.S. Fish and Wildlife Service, EPA, U.S Coast Guard, Florida Governor's Office, Florida Department of Environmental Protection, Florida Department of Community Affairs, South Florida Water Management District, and Monroe County.

The FKNMS Act also called for the public to be a part of the planning process using a Sanctuary Advisory Council (SAC) to aid in the development of a comprehensive management plan. A 22-member SAC was selected by the Governor of Florida and the Secretary of Commerce. The council consisted of members of various user groups; local, state, and federal agencies; scientists; educators; environmental groups; and private citizens.

It quickly became evident that the Congressional option to designate national marine sanctuaries would expedite the designation process. In 1992 four other national marine sanctuaries were designated by Congress, including the Flower Garden Banks, Monterey Bay, Hawaiian Islands Humpback Whale, and Stellwagen Bank (Fig. 8.1). These designations were very similar to the FKNMS Act in that they laid out a process by which sanctuary management should proceed.

8.1.3.4 Recent Proclamation of the Papahānaumokuākea (Northwestern Hawaiian Islands) Marine National Monument

In 2000 President William J. Clinton signed Executive Orders that created the Northwestern Hawaiian Islands (NWHI) Coral Reef Ecosystem Reserve. The orders also initiated a process to designate the waters of the NWHI as a national marine sanctuary. Scoping meetings for the proposed sanctuary were held in 2002. In 2005 Hawaii Governor Linda Lingle signed regulations establishing a state marine refuge in the nearshore waters of the NWHI (out to 3 nautical miles, except Midway Atoll) that excluded all extractive uses of the region, except those permitted for research or other purposes that benefited management. In 2006, after substantial public comment in support of strong protections for the area, President George W. Bush issued Presidential Proclamation 8031, creating the Northwestern Hawaiian Islands Marine National Monument.

The President's actions followed Governor Lingle's lead and immediately afforded the NWHI the highest form of marine environmental protection as the world's largest MPA (360,000 km^2). Administrative jurisdiction over the islands and marine waters is shared by NOAA/NMSP, U.S. Fish and Wildlife Service, and the State of Hawaii.

8.1.4 Interpretation of Goals

The mission of the NMSP is to identify, protect, conserve, and enhance natural and cultural resources, values, and qualities. The NMSP has developed a draft strategic plan with a set of goals (Box 8.1) to provide a bridge between the broad mandates of the NMSA and daily operations at the site level.

At the site level, management and annual operating plans for each national marine sanctuary and the marine national monument identify specific plans and tasks for day-to-day management of the 14 sites. Sanctuaries work closely with their stakeholder Sanctuary Advisory Councils in the processes of developing and revising management plans. Sanctuary staff work with council members to form working groups to analyze each of the action plans that comprise a management plan. There are public scoping meetings to ensure the opportunity for participation by the public. The NMSA stipulates that plans should be reviewed and revised on a five-year time frame, and various sanctuaries are at different phases of this process (Table 8.3). Three Central California sanctuaries are undergoing a joint management plan review, some revisions have been completed, and some are nearing completion. Examples of management plans are provided in the case studies that appear later in this chapter.

8.2 Current Status of Management System

8.2.1 Key Ecosystem Characteristics on Which Goals Depend

In keeping with the goals of the National Marine Sanctuary Program (Box 8.1), sanctuaries within U.S. waters are generally set aside for the preservation of biological or maritime heritage resources. Sites such as the Florida Keys and Channel Islands NMS are of the former, while the Monitor NMS is of the latter. Sites designated to protect marine biological resources have their primary focus on maintaining biodiversity or preserving key species and are therefore directly related to NMSP Goals 1 and 4. These sites are also the ones most in need of management in response to climate change.

8.2.1.1 Biodiversity

The extraordinary biodiversity of tropical and subtropical coral reef sites is well recognized (see the case studies in sections 8.4.1, 8.4.2, and 8.4.3), but recent findings underscore the fact that high biodiversity is also characteristic of many temperate sanctuaries. For example, the recent discovery of deep, temperate corals in the Olympic Coast NMS raises the possibility that benthic invertebrate and associated fish diversity is significantly higher than previously thought. Though receiving substantially less attention from the scientific community than their tropical counterparts, subtidal temperate reefs may be no less important in promoting species diversity and enhancing production (Jonsson *et al.*, 2004; Roberts and Hirshfield, 2004). In the past these reefs have been overlooked and under-studied primarily because of accessibility: they often occur in deeper or lower-visibility waters. Recently and primarily because of greater accessibility to deep water ecosystems, the importance of temperate reefs as critical habitat has begun to be fully recognized (*e.g.*, Reed, 2002; Jonsson *et al.*, 2004; Roberts and Hirshfield, 2004; Roberts, Wheeler, and Freiwald, 2006). These reefs may host an array of undescribed species, including endemic gorgonians, corals, hydroids and sponges (Koslow *et al.*, 2001; Jonsson *et al.*, 2004). Furthermore, the value of these offshore reefs to fisheries has long been recognized by commercial and recreational fisherman. Fish tend to aggregate on deep-sea reefs (Husebø *et al.*, 2002), and scientific evidence supports the contention by commercial fishermen that damage to temperate reefs affects both the abundance and distribution of fish (Fosså, Mortensen, and Furevik, 2002; Krieger and Wing, 2002).

8.2.1.2 Key Species

Key species within sanctuary boundaries may be resident as well as migratory and may or may not represent species that are extracted by fishing (*i.e.*, NMSP Goal 5; Box 8.1). For example, three adjacent sanctuaries off the California coast—Cordell Banks, Gulf of the Farallones, and Monterey Bay—are frequented by protected species of blue (*Balaenoptera musculus*) and humpback (*Megaptera novaeangliae*) whales. In contrast, during the spring of each year king mackerel (*Scomberomorus cavalla*) migrate through Gray's Reef NMS off the coast of Georgia and represent a vibrant and sought-after recreational fishery. Under various climate change scenarios, management strategies employed to protect these key species may differ. Furthermore, key species within sanctuaries may not be limited to subtidal marine organisms but, depending on the sanctuary, may also include intertidal species (*e.g.*, *Mytilus californianus* in Monterey Bay NMS) or even sea and shorebirds. It has been suggested that these intertidal species are more likely to be stressed by climate change and may serve as a bellwether for change in other ecosystems (Helmuth, 2002). In all sanctuaries protected for biological reasons, biodiversity may be affected by climate change and must be managed to meet sanctuary goals. This topic is addressed by case studies presented later in this chapter.

8.2.1.3 Habitat Complexity

National marine sanctuary sites, especially subtidally, are characterized by complexity of habitat that is either biologically or geologically structured. This habitat complexity is an invaluable resource supporting biodiversity. Subtidal habitats in sanctuaries that are biologically structured are represented most notably by temperate kelp forests and tropical corals reefs, whereas geologically structured habitats are centered around sea mounts and rocky outcrops. The topographic complexity of geologically structured habitats, especially in temperate systems, is often enhanced by settlement and growth of sessile benthic invertebrates such as sponges, arborescent bryozoans, and ascidians (*e.g.*, Grays Reef NMS).

Habitat complexity is a key ecosystem characteristic that must be protected in order to achieve NMSP Goals 1 and 4 (Box 8.1). Biologically structured habitats, rather than geologically structured, are probably most susceptible to degradation resulting from climate change. As indicated in section 8.2.2 (*Stressors of Concern*), excess CO_2 absorbed by sea water lowers pH and results in reduced calcification rates in organisms that provide complex structure, such as arborescent bryozoans, bivalves, coralline algae, and temperate and tropical corals (Hoegh-Guldberg, 1999; Kleypas *et al.*, 1999; Kleypas and Langdon, 2006). Non-calcifying biological structures, such as kelp, as well as all shallow water structures are also at risk primarily from changes in storm activity, ocean warming, and reduced upwelling associated with climate change (see Case Study: Channel Islands National Marine Sanctuary).

8.2.1.4 Trophic Cascades

In addition to biodiversity and habitat complexity, trophic links between the benthos and water column help maintain ecosystem integrity within sanctuaries. In keeping with NMSP Goal 5 (Box 8.1) regarding human use, the strength of these benthic-pelagic linkages must be considered when designating fishing restrictions (Wahle, Grober-Dunsmore, and Wooninck, 2006; Grober-Dunsmore, Wooninck, and Wahle, In Press). Fishing regulations often involve removal of top predators and have direct impacts on trophic cascades that are defined as: 1) having top-down control of community structure and 2) having conspicuous indirect effects on two or more links distant from the primary one (Frank *et al.*, 2005). The consequences of ignoring past experiences regarding these trophic cascades could be deleterious to sanctuary goals (Hughes *et al.*, 2005). As highlighted in a recent workshop sponsored by the MPA Science Institute, however, knowledge in this critical area is lacking (Wahle, Grober-Dunsmore, and Wooninck, 2006). Facilitating a better understanding of trophic cascades by supporting scientific inquiry into this topic would do much to enhance understanding of ecosystem processes in marine sanctuaries (NMSP Goal 4). It may also provide insight into how these processes might be impacted by climate change.

8.2.1.5 Connectivity

The open nature of marine ecosystems means that they do not function, and likewise should not be managed, in isolation (Palumbi, 2003). Connectivity among marine ecosystems and across biological communities contributes to maintaining the biological integrity of all marine environments (Kaufman *et al.*, 2004). While NMS boundaries are well defined, the separation between ecosystems and communities is blurred because of export and import of resources. At the broadest scale these linkages are manifested as sources and sinks of nutrients and recruits (*e.g.*, Crowder *et al.*, 2000).

8.2.1.6 Nutrient Fluxes

While excess nutrients can lead to degradation of offshore ecosystems (Rabalais, Turner, and Wiseman Jr, 2002), it is also hypothesized that the function of offshore ecosystems is dependent on nutrients that have their origins in upland productivity. Estuaries are thought to represent the conduit through which dissolved and particulate material from the continent passes to offshore areas through rivers (Gattuso, Frankignoulle, and Wollast, 1998). This “outwelling” characteristic was first proposed by Odum (1969) and has since been applied to mangroves and seagrasses (Lee, 1995). The direct and indirect trophic links that exist between these ecosystems are thought to be critical to ecosystem function and highlight the importance of assessing the downstream effects that upland and nearshore activities have on increasing and decreasing nutrient availability offshore. In areas where climate change alters historical rainfall patterns, concomitant alteration of the supply of nutrients to offshore ecosystems might also occur.

8.2.1.7 Larval Dispersal and Recruitment

One of the strengths of the NMSP is protection of entire ecosystems rather than management of single species. As such, a key characteristic of these ecosystems rests in their ability to serve as sources of recruits for both fish and invertebrate species and as foci for fish aggregations. Most benthic marine invertebrates and fish species have a planktonic larval stage that results from spawned gametes (Pechenik, 1999). Successful recruitment of planktonic larvae to the benthos depends on processes that function at multiple spatial scales in contrast to non-planktonic larvae, which generally recruit at a small spatial scale. At the broadest scale, hydrodynamic forces may disperse passive larvae long distances, potentially delivering them to suitable settlement sites far from the source population (Williams, Wolanski, and Andrews, 1984; Lee *et al.*, 1992). Alternatively, complex, three-dimensional secondary flows resulting from barriers, such as headlands, islands, and reefs, as well as cyclonic motion can retain passive larvae within estuaries, around islands, or within ocean basins, resulting in more settlement to natal populations (Black, Moran, and Hammond, 1991; Lee *et al.*, 1992; Black *et al.*, 1995; Lugo-Fernandez *et al.*, 2001).

Because of their small size and limited swimming ability, invertebrate larvae may be passively dispersed at a broad spatial scale (Denny, 1988; Mullineaux and Butman, 1991). Yet larvae of many marine invertebrates, including coral planulae, use swimming behavior, stimulated by chemical or physical cues, to control their position within the water column, thereby increasing the probability that they will be transported to suitable settlement substrates (Scheltema, 1986; Raimondi and Morse, 2000; Gleason, Edmunds, and Gates, 2006; Levin, 2006). In contrast, researchers continue to be surprised by the swimming and sensory capabilities of fish larvae (Stobutzki and Bellwood, 1997; Tolimieri, Jeffs, and Montgomery, 2000; Leis and McCormick, 2002; Leis, Carson-Ewart, and Webley, 2002; Lecchini *et al.*, 2005; Lecchini, Planes, and Galzin, 2005). That these larvae orient in the water column and swim directionally either at hatching or soon thereafter may explain recent evidence for localized recruitment (Jones *et al.*, 1999; Swearer *et al.*, 1999; Taylor and Hellberg, 2003; Cowen, Paris, and Srinivasan, 2006).

While connectivity among ecosystems and among biological communities in terms of both nutrients and recruits is an important feature of marine sanctuaries, boundaries of protected areas rarely encompass the continuum of habitats (*e.g.*, rivers to estuaries to mangroves to seagrasses to reefs) or the maximum dispersal distances of critical species. Recent information obtained for dispersal of both fish and invertebrates suggests that sanctuaries must be managed for both self-recruitment and larval subsidies from upstream (Roberts, 1997b; Hughes *et al.*, 2005; Cowen, Paris, and Srinivasan, 2006; Steneck, 2006). Effective exchange of offspring is facilitated by MPA networks that are in close proximity [10–50 km apart according to Roberts *et al.* (2001)]. This would also allow larval exchange among populations and also buffer these populations from climate-driven changes in current regimes. The NMSP should be a critical player in the development of such an MPA network. NMSP Goal 2 provides for the expansion of the nation-wide system of MPAs and encourages cooperation among MPAs administered under a range of programs.

8.2.2 Stressors of Concern

Population growth and coastal development increasingly affect U.S. MPAs; an estimated 153 million people (53% of the U.S. population) lived in coastal counties in 2003, and that number continues to rise (World Resources Institute, 1996; Hinrichsen, Robey, and Upadhyay, 1998; National Safety Council, 1998; World Resources Institute, 2000; National Ocean Service, 2000; U.S. Census Bureau, 2001; Crossett *et al.*, 2004). Growing human impacts are compounded by the fact that, in contrast to most terrestrial conservation areas, MPAs lack fences or other barricades and are subjected to anthropogenic stressors (*e.g.*, coastal development, pollution, fishing and aquaculture, habitat degradation) that originate externally. MPA management has focused on minimizing impacts of these existing anthropogenic stressors. The addition of climate change may exacerbate effects of existing stressors and require new or modified management approaches.

The purpose of this section is: 1) to outline major stressors on marine organisms and communities resulting from climate change and 2) to introduce ways in which major "traditional" stressors may interact with climate change stressors.

There are excellent, extensive reviews of impacts of climate change on marine organisms and communities (*e.g.*, Scavia *et al.*, 2002; Walther *et al.*, 2002; Goldberg and Wilkinson, 2004; Harley *et al.*, 2006). By contrast, the scientific knowledge required to reach general conclusions related to the impact of multiple stressors at community and ecosystem levels is for the most part absent. Thus, information concerning interactions among stressors is limited.

8.2.2.1 Direct Climate Change Stressors

Ocean Warming

According to Bindoff *et al.* (2007), there is high confidence that an average warming of 0.1°C has occurred in the 0–700 m depth layer of the ocean between 1961 and 2003. Increasing ocean temperatures, especially near the surface, affect physiological processes in organisms ranging from enzyme reactions to reproductive timing (Fields *et al.*, 1993; Roessig *et al.*, 2004; Harley *et al.*, 2006). The historical stability of ocean temperatures makes many marine species sensitive to thermal perturbations just a few degrees higher than those experienced over evolutionary time (Wainwright, 1994). However, it is not always intuitive which species might be most intolerant of temperature increases. For example, studies on porcelain crabs (*Petrolisthes*) and intertidal snails (*Tegula*) show that individuals in the mid-intertidal are closer to upper temperature limits and have less capacity to acclimate to temperature perturbations than subtidal congeners in temperature-stable conditions (Tomanek and Somero, 1999; Stillman, 2003; Harley *et al.*, 2006).

What is clear is that increasing sea temperatures will continue to influence processes such as foraging, growth, and larval duration and dispersal, with ultimate impacts on the geographic ranges of species. In fact, poleward latitudinal shifts in some zooplankton, fish and intertidal invertebrate communities have already been observed along the California coast and in the North Atlantic (reviewed in Walther *et al.*, 2002). Within marine communities, these temperature changes may result in new species assemblages and biological interactions that affect ecological processes such as productivity, nutrient fluxes, energy flow, and trophic webs (Barry *et al.*, 1995; Roessig *et al.*, 2004; Precht and Aronson, 2004; O'Connor *et al.*, 2007). Species that are unable to shift geographic ranges (perhaps due to physical barriers) or compete with other species for resources may face local—and potentially global—extinction. Conversely, some species may find open niches and dominate regions because of release from competition or predation.

Impacts at the ecosystem or community level are even more difficult to predict. For example, warmer waters stimulate increases in population sizes of the mid-intertidal sea star, *Pisaster ochraceus*, and its per capita consumption rates of mussels (Sanford, 1999). Continued warming may enable *P. ochraceus* to clear large sections of mussel beds, indirectly affecting hundreds of species associated with these formations (Harley *et al.*, 2006). How such an outcome impacts trophic links and other biological processes within this community is not clear.

The latest reports from the IPCC (2007a; 2007b) state that temperature increases over the last 50 years are nearly twice those for the last 100 years, with projections that temperature will rise 2–4.5^{o}C, largely caused by a doubling of atmospheric carbon dioxide emissions. Increases in seawater surface temperature of about 1–3ºC are likely to cause more frequent coral bleaching events that cause widespread mortality unless thermal adaptation or acclimatization by corals occurs (IPCC, 2007b). However, the ability of corals to adapt or acclimatize to increasing seawater temperature is largely unknown (Berkelmans and van Oppen, 2006) and remains a research topic of paramount importance.

Consequences of coral bleaching, during which corals lose their symbiotic algae, depend on the severity and duration of the bleaching event and range from minimal affects on growth and reproduction to widespread mortality. Coral bleaching at the ecosystem level is a relatively recent phenomenon, first receiving widespread attention in 1987 when abnormally high summer seawater surface temperatures throughout the Caribbean resulted in a mass bleaching event (Williams, Goenaga, and Vicente, 1987; Williams and Bunkley-Williams, 1990). Soon after, coral reef scientists identified climate change as a major long-term threat to coral reefs (Glynn, 1991; Smith and Buddemeier, 1992). Ten years later, in 1997–1998, a mass bleaching event in association with an ENSO event caused worldwide bleaching and coral mortality (Wilkinson, 1998; 2000), and in 2005 the most devastating Caribbean-wide coral bleaching event to date occurred that, based on modeling, is highly unlikely to have occurred without anthropogenic forcing (Donner, Knutson, and Oppenheimer, 2007). Over the last 20 years, an extensive body of literature has conclusively linked anomalously high summer surface seawater temperatures as the major cause of coral bleaching (Wilkinson, 1998; 2000; Fitt *et al.*, 2001; Wilkinson, 2002; U.S. Climate Change Science Program and Subcommittee on Global Change Research, 2003; Donner *et al.*, 2005; Donner, Knutson, and Oppenheimer, 2007), with widespread agreement that continued warming—as little as 1^{o}C warmer than the average summer maxima is sufficient—will increase the severity and frequency of mass bleaching events (Smith and Buddemeier, 1992; Hoegh-Guldberg, 1999; Hughes *et al.*, 2003; Douglas, 2003; Done and Jones, 2006).

Effects of coral reef bleaching are both biological, including lost biodiversity and other ecosystem services, and economic, resulting in the decline of fisheries and tourism (Buddemeier, Kleypas, and Aronson, 2004). Coral reefs affected by mass bleaching typically take decades or longer to recover and sometimes may not recover at all. In general, coral reef decline throughout the Caribbean region has been caused by a combination of bleaching, disease, and hurricanes (Gardner *et al.*, 2003; Gardner *et al.*, 2005).

Ocean Acidification

Increased CO_2 concentrations lower oceanic pH, making it more acidic. According to the most recent IPCC report, the total inorganic carbon content of the ocean increased by 118 ($\pm$19) billion metric tons of carbon from 1750–1994 and continues to increase through absorption of excess CO_2 (Bindoff *et al.*, 2007). Furthermore, time series data for the last 20 years show a trend of decreasing pH of 0.02 pH units per decade (Bindoff *et al.*, 2007). Long-term exposures to low pH (-0.7 unit) have been shown to reduce metabolic rates, growth, and survivorship of both invertebrates and fishes (Michaelidis *et al.*, 2005; Shirayama and Thornton, 2005; Pane and Barry, 2007), but by far the greatest threat of reducing pH is to organisms that build their external skeletal material out of calcium carbonate ($CaCO_3$). Calcifying organisms such as sea urchins, cold-water corals, coralline algae, and various plankton that reside in cooler temperate waters appear to be the most threatened by acidification because CO_2 has greater solubility in cooler waters (Hoegh-Guldberg, 1999; Kleypas *et al.*, 1999; Hughes *et al.*, 2003; Feely *et al.*, 2004; Kleypas and Langdon, 2006).

The response of corals and coral reefs to ocean acidification has received substantial attention, and results show that lowering pH results in significant reductions in calcification rates in both reef-building corals and coralline algae (Kleypas *et al.*, 1999; Feely *et al.*, 2004; Orr *et al.*, 2005; Kleypas and Langdon, 2006). Declines in calcification rates of 17–35% by the year 2100 have been estimated based on projected changes in the partial pressure of CO_2 (Hoegh-Guldberg, 1999; Kleypas *et al.*, 1999; Hughes *et al.*, 2003; Orr *et al.*, 2005). Because of the greater solubility of CO_2 in cooler waters, reefs at the latitudinal margins of coral reef development (*e.g.*, Florida Keys and Hawaiian Islands) may show the most rapid and dramatic response to changing pH.

Rising Sea Level

During the last 100 years, global average sea level has risen an estimated 1–2 mm per year and is expected to accelerate due to thermal expansion of the oceans and melting ice-sheets and glaciers (Cabanes, Cazenave, and Le Provost, 2001; Albritton and Filho, 2001; Rignot and Kanagaratnam, 2006; Chen, Wilson, and Tapley, 2006; Shepherd and Wingham, 2007; Bell *et al.*, 2007; IPCC, 2007b). Rates of sea level rise at a local scale vary from -2 to 10 mm per year along U.S. coastlines (Nicholls and Leatherman, 1996; Zervas, 2001; Scavia *et al.*, 2002). Low-lying areas, especially intertidal zones, along the eastern and Gulf coasts are at the greatest risk of damage from rising sea level (Scavia *et al.*, 2002). The consequences of sea level rise include inundation of coastal areas, erosion of vulnerable shorelines, and landward shifts in species distributions.

On undeveloped coasts with relatively gentle slopes, it is thought that plant communities such as mangroves and *Spartina* salt marshes will move inland as sea level rises (Scavia *et al.*, 2002; Harley *et al.*, 2006). In contrast, coastline development will interfere with these plant migrations. As a result, wetlands may become submerged and soils may become waterlogged, resulting in plant physiological stress due to chronic and intolerable elevated salinity. Marshes, mangroves and dune plants are critical to the coastal environment because they produce and add nutrients to the coastal systems, stabilize substrates, and serve as refuges and nurseries for many species. Their depletion or loss would therefore affect nutrient flux, energy flow and essential habitat for a multitude of species, with ultimate long-term impacts on biodiversity (Scavia *et al.*, 2002; Galbraith *et al.*, 2002; Harley *et al.*, 2006). The projected 35–70% loss of barrier islands and intertidal and sandy beach habitat over the next 100 years could also drastically reduce nesting grounds for key species such as sea turtles and birds as these critical habitats disappear (Scavia *et al.*, 2002).

Climatic Variability and Ocean Circulation

Natural climatic variability resulting from ocean-atmosphere interactions such as the El Niño Southern Oscillation (ENSO), Pacific Decadal Oscillation (PDO), and North Atlantic Oscillation/Northern Hemisphere Annular Mode (NAO/NHM) result in changes in open ocean productivity, shifts in the distribution of organisms and modifications in food webs that foreshadow potential consequences of accelerated climate change (*e.g.*, Mantua *et al.*, 1997; McGowan *et al.*, 1998). These recurring patterns of ocean-atmosphere variability have very different behaviors in time. For example, whereas ENSO events persist for 6–18 months and have their major impact in the tropics, the PDO occurs over a much longer time frame of 20–30 years and has primary effects in the northern Pacific (Mantua *et al.*, 1997). Regardless of the temporal scale and region of impact, however, these natural modes of climate variability have existed historically, independent of anthropogenically driven climate change. These climate phenomena may act in tandem with (or in opposition to) human-induced alterations, with consequences that are difficult to predict (Philip and Van Oldenborgh, 2006).

Ocean-atmosphere interactions on a warming planet may also result in long-term alterations in the prevailing current and upwelling patterns (Bakun, 1990; McPhaden and Zhang, 2002; Snyder *et al.*, 2003; McGregor *et al.*, 2007). While at present there is no clear indication that ocean circulation patterns have changed (Bindoff *et al.*, 2007), modifications could have large effects within and among ecosystems through impacts on ecosystem and community connectivity in terms of both nutrients and recruits (see section 8.2.1., *Key Ecosystem Characteristics Upon Which Goals Depend*). Considering that there is evidence for warming of the Southern Ocean mode waters and Upper Circumpolar Deep Waters from 1960–2000, changes in oceanic current and upwelling patterns are likely in the future (Bindoff *et al.*, 2007). The direction that these changes will take, however, is not evident. For example, it has been hypothesized that the greater temperature differential between the land mass and ocean that will occur with climate warming will increase upwelling because of stronger alongshore winds (Bakun, 1990). In contrast, Gucinski, Lackey, and Spence (1990) proposed that warming at higher latitudes will reduce latitudinal temperature gradients resulting in decreased wind strength and less upwelling; some models show potential for Atlantic thermohaline circulation to end abruptly if high-latitude waters are no longer able to sink (Stocker and Marchal, 2000).

Storm Intensity

Whether or not storm frequency has changed over time is not clear because of large natural variability resulting from such climate drivers as the ENSO (IPCC, 2007b). However, since the mid 1970s there has been a trend toward longer storm duration and greater storm intensity (IPCC, 2007b). An increase in storm intensity generally has impacts on two fronts. First, it may increase pulses of fresh water to coastal and near-shore habitats (see below). Second, increasing storm intensity may cause physical damage to coastal ecosystems, especially those in shallow water (IPCC, 2007b).

Recent hurricanes in the southern United States have caused: extensive destruction to homes and businesses; altered near-shore water quality; scoured the ocean bottom; over-washed beaches; produced immense amounts of marine debris (wood, metals, plastics) and pollution (household hazardous wastes, pesticides, metals, oils and other toxic chemicals) from floodwaters; and damaged many mangrove, marsh, and coral reef areas (Davis *et al.*, 1994; Tilmant *et al.*, 1994; McCoy *et al.*, 1996; Lovelace and MacPherson, 1998; Baldwin *et al.*, 2001; U.S. Fish and Wildlife Service, 2005). Even 30–60 days after the storms, some areas still experienced increased turbidity, breakdown of mangrove peat soils and elevated concentrations of ammonia, dissolved phosphate, and dissolved organic carbon (Davis *et al.*, 1994; Tilmant *et al.*, 1994; Lovelace and MacPherson, 1998). In some instances, algal blooms from the high nutrients further increased the turbidity while driving down dissolved-oxygen concentrations (*i.e.*, caused eutrophication), resulting in mortalities in fish and invertebrate populations (Tilmant *et al.*, 1994; Lovelace and MacPherson, 1998). Given that most climate change models project increasing storm intensity as well as higher sea levels in many areas, it is evident that low-lying and shallow marine ecosystems such as mangroves, salt marshes, sea grasses, and coral reefs are at greatest risk of long-term damage.

Freshwater Influx

Observations indicate that changes in the amount, intensity, frequency and type of precipitation are occurring worldwide (IPCC, 2007b). Consistent with observed changes in precipitation and water transport in the atmosphere, large-scale trends in oceanic salinity have become evident for the period 1955–1998 (Bindoff *et al.*, 2007). These trends are manifested as lowered salinities at subpolar latitudes and increased salinities in shallower parts of the tropical and subtropical oceans.

In addition to altering salinity in major oceanic water masses, changes in precipitation patterns can have significant impacts in estuarine and other near-shore environments. For instance, in regions where climate change results in elevated rainfall, increased runoff may cause greater stratification of water layers within estuaries as fresh water floats out over the top of higher salinity layers (Scavia *et al.*, 2002). One consequence of this stratification may be less water column mixing and thus lower rates of nutrient exchange among water layers. Combining this stratification effect with the shorter water residence times stemming from higher inflow (Moore *et al.*, 1997) may result in significantly reduced productivity because phytoplankton populations may be flushed from the system at a rate faster than they can grow and reproduce. On the other hand, estuaries that are located in regions with lower rainfall may also show decreased productivity because of lower nutrient influx. Thus, the relationship between precipitation and marine ecosystem health is complex and difficult to predict.

Another source of fresh water is melting of polar ice (IPCC, 2007b). In the Atlantic Ocean, accelerated melting of Arctic ice and the Greenland ice sheet are predicted to continue producing more freshwater inputs that may alter oceanic circulation patterns (Dickson *et al.*, 2002; Curry, Dickson, and Yashayaev, 2003; Curry and Mauritzen, 2005; Peterson *et al.*, 2006; Greene and Pershing, 2007; Boessenkool *et al.*, 2007).

8.2.2.2 Climate Change Interactions with "Traditional" Stressors of Concern

Pollution

Marine water quality degradation and pollution stem primarily from land-based sources, with major contributions to coastal watershed and water quality deterioration falling into two broad categories: point source pollution and non-point source pollution. Point source pollution from factories, sewage treatment plants, and farms often flows into nearby waters. In contrast, marine non-point source pollution originates from coastal urban runoff where the bulk of the land is paved or covered with buildings. These impervious surfaces prevent soils from capturing runoff, resulting in the input of untreated pollutants (*e.g.*, fuels, oils, plastics, metals, insecticides, antibiotics) to coastal waters. Increased terrestrial runoff due to more intense storm events associated with climate change may increase land-based water pollution from both of these sources.

Deterioration and pollution of coastal watersheds can have far-reaching effects on marine ecosystems. As an example, the Gulf of Mexico "dead zone" that occurs each summer and extends from the Mississippi River bird-foot delta across the Louisiana shelf and onto the upper Texas coast can range from 1–125 km offshore (Rabalais, Turner, and Wiseman Jr, 2002). This mass of hypoxic (low-oxygen) water has its origins in the increased nitrate flux coincident with the exponential growth of fertilizer use that has occurred since the 1950s in the Mississippi River basin. This hypoxia results in changes in species diversity and community structure of the benthos and has impacts on trophic links that include higher order consumers in the pelagic zone (Rabalais, Turner, and Wiseman Jr, 2002).

Until recently, pollution has been the major driver of decreases in the health of marine ecosystems such as coral reefs, sea grasses, and kelp beds (Jackson *et al.*, 2001; Hughes *et al.*, 2003; Pandolfi *et al.*, 2003). Because pollution is usually more local in scope, it historically could be managed within individual MPAs; however, the addition of climate change stressors such as increased oceanic temperature, decreased pH, and greater fluctuations in salinity present greater challenges (Coe and Rogers, 1997; Carpenter *et al.*, 1998; Khamer, Bouya, and Ronneau, 2000; Burton, Jr. and Pitt, 2001; Sobel and Dahlgren, 2004; Orr *et al.*, 2005; Breitburg and Riedel, 2005; O'Connor *et al.*, 2007; IPCC, 2007b).

For example, coral bleaching from the combined stresses of climate change and local pollution (*e.g.*, high temperature and sedimentation) have already been observed (Jackson *et al.*, 2001; Hughes *et al.*, 2003; Pandolfi *et al.*, 2003). Identifying those stressors with the greatest effect is not trivial. Research in coral genomics may provide diagnostic tools for identifying stressors in coral reefs and other marine communities (*e.g.*, Edge *et al.*, 2005).

Commercial Fishing and Aquaculture

Commercial fishing has ecosystem effects on three fronts: through the physical impacts of fishing gear on habitat, over-fishing of commercial stocks and incidental take of non-targeted species. The use of trawls, seines, mollusk dredges, and other fishing gear can cause damage to living seafloor structures and alterations to geologic structures, reducing habitat complexity (Engel and Kvitek, 1998; Thrush and Dayton, 2002; Dayton, Thrush, and Coleman, 2002; Hixon and Tissot, 2007). Over-fishing is also common in the United States, with a conservative estimate of 26% of fisheries overexploited (Pauly *et al.*, 1998; National Research Council, 1999; Jackson *et al.*, 2001; Pew Ocean Commission, 2003; National Marine Fisheries Service, 2005; Lotze *et al.*, 2006). Meanwhile, non-specific fishing gear (*e.g.*, trawls, seines, dredges) causes considerable mortality of by-catch that includes invertebrates, fishes, sea turtles, marine mammals, birds, and other life stages of commercially targeted species (Condrey and Fuller, 1992; Norse, 1993; Sobel and Dahlgren, 2004; Hiddink, Jennings, and Kaiser, 2006).

Aquaculture has sometimes been introduced to augment fisheries production. Unfortunately experience shows that aquaculture can have negative environmental impacts including extensive mangrove and coastal wetland conversion to ponds, changes in hydrologic regimes, and discharge of high levels of organic matter and pollutants into coastal waters (Eng, Paw, and Guarin, 1989; Iwama, 1991; Naylor *et al.*, 2000). Furthermore, many aquacultural practices are not sustainable because farmed species consume natural resources at high rates and the intense culture environment (*e.g.*, overcrowding) creates conditions for disease outbreaks (Eng, Paw, and Guarin, 1989; Iwama, 1991; Pauly *et al.*, 2002; 2003).

Fishery populations that are overstressed and overfished exhibit greater sensitivity to climate change and other anthropogenically derived stressors than do healthy populations (Hughes *et al.*, 2005). Overfishing can reduce mean life span as well as lifetime reproductive success and larval quality, making fished species more susceptible to both short- and long-term perturbations (such as changes in prevailing current patterns) that affect recruitment success (Pauly *et al.*, 1998; Jackson *et al.*, 2001; Dayton, Thrush, and Coleman, 2002; Pauly *et al.*, 2003; Sobel and Dahlgren, 2004; Estes, 2005; Law and Stokes, 2005; Steneck and Sala, 2005; O'Connor *et al.*, 2007). Changing climatic regimes can also influence species' distributions, which are set by physiological tolerances to temperature, precipitation, dissolved oxygen, pH, and salinity. Because rates of climate change appear to exceed the capacity of many commercial species to adapt, species will shift their ranges in accordance with their physiological thresholds and may ultimately be forced to extend past the boundaries of their "known" native range, becoming invasive elements (Murawski, 1993; Walther *et al.*, 2002; Roessig *et al.*, 2004; Perry *et al.*, 2005; Harley *et al.*, 2006).

Commercial exploitation of even a single keystone species, such as a top consumer, can destabilize ecosystems by decreasing redundancy and making them more susceptible to climate change stressors (Hughes *et al.*, 2005). Examples of such ecosystem destabilization through overfishing abound, including the formerly cod-dominated system of the western North Atlantic (see Box 8.2), and the fish grazing community on Caribbean coral reefs (*e.g.*, Frank *et al.*, 2005; Mumby *et al.*, 2006).

Nonindigenous/Invasive Species

Invasive species threaten all marine and estuarine communities. Currently, an estimated 2% of extinctions in marine ecosystems are related to invasive species while 6% are the result of other factors including climate change, pollution, and disease (Dulvy, Sadovy, and Reynolds, 2003). Principal mechanisms of introduction vary and have occurred via both accidental and intentional release (Ruiz *et al.*, 2000; Carlton, 2000; Hare and Whitfield, 2003). Invasive species are often opportunistic and can force shifts in the relative abundance and distribution of native species, and cause significant changes in species richness and community structure (Sousa, 1984; Moyle, 1986; Mills, Soulé, and Doak, 1993; Baltz and Moyle, 1993; Carlton, 1996; Carlton, 2000; Marchetti, Moyle, and Levine, 2004).

Some native species, particularly rare and endangered ones with small population sizes and gene pools, are unlikely to be able to adapt quickly enough or shift their ranges rapidly enough to compensate for the changing climatic regimes proposed by current climate change models (IPCC, 2007b). These species will likely have their competitive abilities compromised and be more susceptible to displacement by invasive species. Increased seawater temperatures resulting from climate change may also allow introduced species to spawn earlier and for longer periods of the year, thus increasing their population growth rates relative to natives while simultaneously expanding their range (Carlton, 2000; McCarty, 2001; Stachowicz *et al.*, 2002; Marchetti, Moyle, and Levine, 2004). Furthermore, the same characteristics that make species successful invaders may also make them pre-adapted to respond to, and capitalize on, climate change. As one example, Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) are now widely distributed off the southeastern coast of the United States less than 10 years after being first observed off Florida (Whitfield *et al.*, 2007). One of the few factors limiting their spread is intolerance to minimum water temperatures during winter (Kimball *et al.*, 2004). Ocean warming could facilitate depth and range expansion in these species.

Diseases

Disease outbreaks alter the structure and function of marine ecosystems by affecting the abundance and diversity of vertebrates (*e.g.*, mammals, turtles, fish), invertebrates (*e.g.*, corals, crustaceans, echinoderms, oysters) and plants (*e.g.*, seagrasses, kelp beds). Pathogen outbreaks or epidemics spread rapidly due to the lack of dispersal barriers in some parts of the ocean and the potential for long-term survival of pathogens outside the host (Harvell *et al.*, 1999; Harvell *et al.*, 2002). Many pathogens of marine taxa such as coral viruses, bacteria, and fungi are positively responsive to temperature increases within their physiological thresholds (Porter *et al.*, 2001; Kim and Harvell, 2004; Munn, 2006; Mydlarz, Jones, and Harvell, 2006; Boyett, Bourne, and Willis, 2007).

Exposure to disease compromises the ability of species to resist other anthropogenic stressors and vice versa (Harvell *et al.*, 1999; Harvell *et al.*, 2002). For example, in 1998, the most geographically extensive and severe coral bleaching ever recorded was associated with the high sea surface temperature anomalies facilitated by an ENSO event (Hoegh-Guldberg, 1999; Wilkinson *et al.*, 1999; Mydlarz, Jones, and Harvell, 2006). In some species of reef-building corals and gorgonians, this bleaching event was thought to be accelerated by opportunistic infections (Harvell *et al.*, 1999; Harvell *et al.*, 2001). Several pathogens—such as bacteria, viruses, and fungi that infect such diverse hosts as seals, abalone, and starfish—show possible onset with warmer temperatures (reviewed in Harvell *et al.*, 2002). The mechanisms for pathogenesis, however, are largely unknown. Given that exposure to multiple stressors may compromise the ability of marine species to resist infection, the most effective means of reducing disease incidence under climate change may be to minimize impacts of stressors such as pollution and overfishing.

8.2.3 Management Approaches and Sensitivity of Management Goals to Climate Change

Marine protected area programs have been identified as a critical mechanism for protecting marine biodiversity and associated ecosystem services (Ballantine, 1997; National Research Council, 2001; Palumbi, 2002; Roberts *et al.*, 2003a; Sobel and Dahlgren, 2004; Palumbi, 2004; Roberts, 2005; Salm, Done, and McLeod, 2006). MPA networks are being implemented globally to address multiple threats to the marine environment, and are generally accepted as an improvement over individual MPAs (Ballantine, 1997; Salm, Clark, and Siirila, 2000; Allison *et al.*, 2003; Roberts *et al.*, 2003a; Mora *et al.*, 2006). Networks are more effective than single MPAs at protecting the full range of habitat and community types because they spread the risk of losing a habitat or community type following a disturbance such as a climate-change impact across a larger area. Networks are better able to protect both short- and long-distance dispersers than individual MPAs and thus have more potential to achieve conservation and fishery objectives (Roberts, 1997a). Networks provide enhanced larval recruitment among adjacent MPAs that are linked by local and regional dispersal patterns, enhanced protection of critical life stages, and enhanced protection of critical processes and functions, *e.g.*, migration corridors (Gerber and Heppell, 2004). Finally, networks allow for protection of marine ecosystems at an appropriate scale. A network of MPAs could cover a large gradient of biogeographic and oceanographic conditions without the need to establish one extremely large reserve and can provide more inclusive representation of stakeholders (National Research Council, 2001; Hansen, Biringer, and Hoffman, 2003).

While MPA networks are considered a critical management tool for conserving marine biodiversity, they must be established together with other management strategies to be effective (Hughes *et al.*, 2003). MPAs are vulnerable to activities beyond their boundaries. For example, uncontrolled pollution and unsustainable fishing outside protected areas can adversely affect the species and ecosystem function within the protected area (Kaiser, 2005). Therefore, MPA networks should be established considering other forms of fisheries management (*e.g.*, catch limits and gear restrictions) (Allison, Lubchenco, and Carr, 1998; Beger, Jones, and Munday, 2003; Kaiser, 2005) and coastal management to control land-based threats such as pollution and sedimentation (Cho, 2005). In the long term, the most effective configuration would be a network of highly protected areas nested within a broader management framework (Salm, Done, and McLeod, 2006). Such a framework might include a vast multiple-use area managed for sustainable fisheries as well as protection of biodiversity, integrated with coastal management regimes where appropriate, to enable effective control of threats originating upstream and to maintain high water quality (*e.g.*, Done and Reichelt, 1998).

The National Marine Sanctuary Program has developed a set of goals (Box 8.1) to help clarify the relationship between operations at individual sanctuaries and the broad directives of the National Marine Sanctuaries Act. A subset of these goals (Goals 1, 4, 5, and 6) are relevant to resource protection and climate change. Box 8.3 expands upon Goals 1, 4, 5, and 6 to display their attendant objectives, which provide guidance for management plans that are developed by sanctuary sites (see Table 8.3). Sanctuary management plans are developed and subsequently reviewed and revised on a five-year cycle as a collaboration between sanctuary staff and local communities. After threats and stressors to resources are identified, action plans are prepared that identify activities to address them. Threats and stressors may include such things as overexploitation of natural resources, degraded water quality, and habitat damage and destruction. Sanctuary management plans are designed to address additional issues raised by local communities, such as user conflicts, needs for education and outreach, and interest in volunteer programs.

Fully protected marine reserves within national marine sanctuaries have been implemented at some sites (*e.g.*, Channel Islands and the Florida Keys; Keller and Causey, 2005) to reduce fishing pressure; the entire area of the Papahānaumokuākea Marine National Monument will become no-take within five years. These additional protective actions complement existing fishery regulations. Some sites such as Monterey Bay and the Florida Keys have Water Quality Protection Programs to address issues such as watershed pollution, vessel discharges, and, in the case of the Florida Keys, wastewater and stormwater treatment systems. Habitat damage may be addressed using waterway marking programs to reduce vessel groundings and mooring buoys to minimize anchor damage. Many of these activities are supported through education and outreach programs to inform the public, volunteer programs to help distribute information (e.g., Team OCEAN; Florida Keys National Marine Sanctuary, 2003), and law enforcement.

Sanctuary management plans are intended to be comprehensive and may take years of community involvement to develop. For example, it took over five years to develop the management plan for the Florida Keys National Marine Sanctuary (Keller and Causey, 2005), and an additional three years were required to prepare a supplemental plan for the Tortugas Ecological Reserve (Cowie-Haskell and Delaney, 2003; Delaney, 2003).

Effective management and preservation of ecosystem characteristics in the face of climate change projections is relevant to achieving NMSP Goals 1, 2, 4, and 5 (Box 8.1). The NMSP can be a leader in employing new management approaches by including stakeholders in decision-making (Sanctuary Advisory Councils and public scoping meetings at the site level). This model of public involvement should serve well as management strategies adapt under the stresses of climate change. Exporting lessons learned to the general public, managers of other MPAs, and the international community will further address NMSP Goals 2, 3, and 6.

An additional approach of the NMSP that should further efforts toward adaptive management in the context of climate change is the development of performance measures to help evaluate the success of the program (Box 8.4). Although climate change stressors are not explicitly addressed in these performance measures, attainment of a number of these measures clearly will be increasingly affected by climate change. The performance-measure approach should encourage sanctuary managers to address climate change impacts using the public processes of Sanctuary Advisory Councils and public scoping meetings. In addition, national marine sanctuaries are preparing Condition Reports (National Marine Sanctuary Program, 2007c), which provide summaries of resources, pressures on resources, current condition and trends, and management responses to pressures that threaten the integrity of the marine environment. These reports will provide opportunities for sanctuaries to evaluate climate change as a pressure and identify management responses on a site-by-site basis.

8.3 Adapting to Climate Change

MPA managers can respond to challenges of climate change at two scales: actions at individual sites and implementing MPA networks. At particular MPAs, managers can increase efforts to ameliorate existing anthropogenic stressors with a goal of reducing the overall load of multiple stressors (Breitburg and Riedel, 2005). For example, the concept of protecting or enhancing coral reef resilience has been proposed to help ameliorate negative consequences of coral bleaching (Hughes *et al.*, 2003; Hughes *et al.*, 2005; Marshall and Schuttenberg, 2006a). Under this approach, resilience is an ecosystem property that can be managed and is defined as the ability of an ecosystem to resist or absorb disturbance without significantly degrading processes that determine community structure, or if alterations occur, recovery is *not* to an alternate community state (Gunderson, 2000; Nyström, Folke, and Moberg, 2000; Hughes *et al.*, 2003). In short, managing for resilience includes dealing with causes of coral reef disturbance and decline that managers can address at local and regional levels, such as overfishing and pollution. These are the things that managers would want to do anyway, even if climate change were not a threat, because these activities help to maintain the ecological and economic value of the ecosystem.

In addition to the approach of ameliorating existing stressors such as overfishing and pollution, MPA managers can protect apparently resistant and potentially resilient areas, develop networks of MPAs, and integrate climate change into planning efforts. Specific examples of adaptation options from across these approaches are presented in Box 8.5 and elaborated upon further in the sections that follow.

8.3.1 Ameliorate Existing Stressors in Coastal Waters

Managers can increase resilience to climate change in areas of interest by managing other stressors, such as fishing, input of nutrients, sediment and pollutants, and water quality. Kelp forest ecosystems in marine reserves, where no fishing is allowed, are more resilient to ocean warming than those in areas where fishing occurs (Behrens and Lafferty, 2004). This ecological response is a result of changes in trophic structure of communities in and around the reserves. When top predators such as spiny lobster are fished, their prey, herbivorous sea urchins, increase in abundance and consume giant kelp and other algae. When kelp forests are subjected to intense grazing by these herbivores, the density of kelp is reduced, sometimes becoming an "urchin barren," particularly during ocean warming events such as ENSO cycles. In reserves, where fishing is prohibited, lobster populations were larger, urchin populations were diminished, and kelp forests persisted over a period of 20 years, including four ENSO cycles (Behrens and Lafferty, 2004).

Managing water quality has been identified as a key strategy for maintaining ecological resilience (Salm, Done, and McLeod, 2006; Marshall and Schuttenberg, 2006a). In the Florida Keys National Marine Sanctuary and the Great Barrier Reef Marine Park, water quality protection is recognized as an essential component of management (The State of Queensland and Commonwealth of Australia, 2003; Grigg *et al.*, 2005; also see the Monterey Bay National Marine Sanctuary's water quality agreements with land-based agencies: Monterey Bay National Marine Sanctuary, 2007). Strong circumstantial evidence exists linking poor water quality to increased macroalgal abundances, internal bioerosion, and susceptibility to some diseases in corals and octocorals (Fabricius and De'ath, 2004). Addressing sources of pollution, especially nutrient enrichment, which can lead to increased algal growth and reduced coral settlement, is critical to maintaining ecosystem health. In addition to controlling point-source pollution within an MPA, managers must also link their MPAs into the governance system of adjacent areas to control sources of pollution beyond the MPA boundaries. Further actions necessary to improve water quality include raising awareness of how land-based activities can adversely affect adjacent marine environments, designing policies for integrated coastal and watershed management, and developing options for advanced wastewater treatment (The Group of Experts on Scientific Aspects of Marine Environmental Protection, 2001).

Managers can build resilience to climate change into MPA management strategies by protecting marine habitats such as coral reefs and mangroves from direct threats such as pollution, sedimentation, destructive fishing, and overfishing. The healthier the marine habitat, the greater the potential will be for it to recover from a catastrophic event such as mass coral bleaching. Therefore, managers should continue to develop and implement strategies to reduce land-based pollution, decrease nutrient and sediment runoff, eliminate the use of persistent pesticides, and increase filtration of effluent to improve water quality.

Another mechanism that has been identified to maintain resilience is the management of functional groups, specifically herbivores (Hughes *et al.*, 2003; Bellwood *et al.*, 2004). Bellwood *et al.* (2004) identified three functional groups of herbivores that assist in maintaining coral reef resilience: bioeroders, grazers, and scrapers. These groups work together to break down dead coral to allow substrate for recruitment, graze macroalgae, and reduce the development of algal turfs to allow for a clean substrate for coral settlement. Algal biomass must be kept low to maintain healthy coral reefs (Sammarco, 1980; Hatcher and Larkum, 1983; Steneck and Dethier, 1994). In a recent paper by Bellwood, Hughes, and Hoey (2006), the authors identify the need to protect both the species that prevent phase shifts from coral-dominated to algal-dominated reefs and the species that help reefs recover from algal dominance. They suggest that while parrotfishes and surgeonfishes appear to play a critical role in preventing phase shifts to macroalgae, their ability to remove algae may be limited if a phase shift to macroalgae has already occurred (Bellwood, Hughes, and Hoey, 2006). In their study on the Great Barrier Reef, the phase shift reversal from macroalgal-dominated to a coral- and epilithic algal-dominated state was driven by a single batfish species (*Platax pinnatus*), not grazing by dominant parrotfishes or surgeonfishes (Bellwood, Hughes, and Hoey, 2006). This finding highlights the need to protect the full range of species to maintain resilience.

Although protecting functional groups is a critical component of resilience, understanding which groups should be protected requires a detailed knowledge of species and interactions that is not often available for all species. Therefore, managers should strive to maintain the maximum number of species in the absence of detailed data on ecological and species interactions. For example, for managing coral reefs, regional guidelines identifying key herbivores that reduce macroalgae and encourage coral reef settlement should be developed. For kelp forests, managers should identify key predators and limit fishing on those predators to reduce herbivory and promote growth of healthy kelp forests. These guidelines should be field tested at different locations to verify the recommendations.

8.3.2 Protect Apparently Resistant and Potentially Resilient Areas

Marine ecosystems that contain biologically generated habitats face potential loss of habitat structure as climate change progresses (*e.g.*, coral reefs, seagrass beds, kelp forests, and deep coral communities) (see Hoegh-Guldberg, 1999; Steneck *et al.*, 2002; Roberts, Wheeler, and Freiwald, 2006; Orth *et al.*, 2006). It is likely that climate change contributes to mass coral bleaching events (Reaser, Pomerance, and Thomas, 2000), which became recognized globally in 1997-1998 (Wilkinson, 1998; 2000) and have affected large regions in subsequent years (Wilkinson, 2002; 2004; Whelan *et al.*, In Press). The amount of live coral has declined dramatically in the Caribbean region over the past 30 years as a result of bleaching, diseases, and hurricanes (Gardner *et al.*, 2003; 2005). In the Florida Keys, fore-reef environments that formerly supported dense growths of coral are now nearly depauperate, and highest coral cover is in patch reef environments (Porter *et al.*, 2002; Lirman and Fong, 2007). Irrespective of the mechanism—resistance, resilience, or exposure to relatively low levels of past environmental stress— these patch-reef environments might be good candidates for additional protective measures due to their ability to survive climate stress.

Done (2001; see also Marshall and Schuttenberg, 2006b) presented a decision tree for identifying areas that would be suitable for MPAs under a global warming scenario. Two types of favorable outcomes included reefs that survived bleaching (*i.e.*, were resilient) and reefs that were not exposed to elevated sea surface temperatures (*e.g.*, may be located within refugia). This type of decision tree has already been adapted to aid resilient site selection for mangroves (McLeod and Salm, 2006) as well, and it could be extended further for other habitat types such as seagrass beds and kelp forests.

Because climate change impacts on marine systems are patchy (with reefs that avoid bleaching one year potentially bleaching the following year), it is essential that areas that appear to be resistant or resilient to climate change impacts be monitored and tested to ensure that they continue to provide benefits (see section 8.3.4.1 for more on monitoring and research). This allows managers to target potential refugia for MPA design now while also monitoring these areas over time so that management can be adapted as circumstances and habitats change (*i.e.*, as per an adaptive management approach).

8.3.3 Develop Networks of MPAs

The concept of systems or networks of MPAs has considerable appeal because of emergent properties (*i.e.*, representation, replication, sustainability, connectivity) (Ballantine, 1997; National Research Council, 2001; Roberts *et al.*, 2003a), spreading the risk of catastrophic habitat loss (Palumbi, 2002; Allison *et al.*, 2003), and the provision of functional wilderness areas sufficient to resist fundamental changes to entire ecosystems (Kaufman *et al.*, 2004). While MPA networks have been recognized as a valuable tool to conserve marine resources in the face of climate change, there have been a number of challenges to implementation (Pandolfi *et al.*, 2005; Mora *et al.*, 2006); nevertheless, a number of principles have been developed and are gradually being applied to aid MPA network design and implementation. These principles are described below.

8.3.3.1 Protect Critical Areas

Critical areas—areas that are biologically or ecologically significant—should be identified and included in MPAs. These critical areas include nursery grounds, spawning grounds, areas of high species diversity, areas that contain a variety of habitat types in close proximity to each other, and climate refugia (Allison, Lubchenco, and Carr, 1998; Sale *et al.*, 2005; Sadovy, 2006). Coral assemblages that demonstrate resilience to climate change may be identified and provided additional protection to ensure a secure source of recruitment to support recovery in damaged areas. Managers can analyze how assemblages have responded to past climate events to determine likely resilience to climate change impacts. For example, some coral reefs resist bleaching due to genetic characteristics or avoid bleaching due to environmental factors. Managers can fully protect those that either resist or recover quickly from mass bleaching events, as well as those that are located in areas where physical conditions (*e.g.*, currents, shading) afford them some protection from temperature anomalies. Reefs that are resistant and reefs that are located in climate refugia play a critical role in reef survival by providing a reliable source of larvae for dispersal to and recovery of affected areas (Salm and Coles, 2001). For coral reefs, indicators of potential refugia include a ratio of live to dead coral and a range of colony sizes and ages suggesting persistence over time. Refugia must be large enough to support high species richness to maximize their effectiveness as sources of recruits to replenish areas that have been damaged (Palumbi *et al.*, 1997; Bellwood and Hughes, 2001; Salm, Done, and McLeod, 2006).

8.3.3.2 Incorporate Connectivity in Planning MPA Networks

Connectivity is the natural linkage between marine habitats (Crowder *et al.*, 2000; Stewart, Noyce, and Possingham, 2003; Roberts *et al.*, 2003b), which occurs through advection by ocean currents and includes larval dispersal and movements of adults and juveniles. Connectivity is an important part of ensuring larval exchange and the replenishment of populations in areas damaged by natural or human-related agents. Salm *et al.* (2006) recommend that patterns of connectivity be identified among source and sink reefs to inform reef selection in the design of MPA networks, providing “stepping-stones” for reefs to enhance recovery following disturbance events. This principle applies to other marine systems, such as mangroves, as well. For example, healthy mangroves could be selected up-current from areas that may succumb to sea level rise, and areas could be selected that would be suitable habitat for mangroves in the future following sea level rise. These areas of healthy mangroves could provide secure sources of propagules to replenish down-current mangroves following a disturbance event.

A suspected benefit of MPAs is the dispersal of larvae to areas surrounding MPAs, but there are few data that can be used to estimate the exchange of larvae among local populations (Palumbi, 2004). Understanding larval dispersal and transport are critical to determining connectivity, and thus the design of MPAs. The size of an individual MPA should be based on the movement of adults of species of interest (Hastings and Botsford, 2003; Botsford, Micheli, and Hastings, 2003; California Department of Fish and Game, 2007a). An individual MPA should be large enough to contain the different habitats used and the daily movements of species of interest. The distance between adjacent MPAs should take into account the potential dispersal distances of larvae of fish, invertebrates, and other species of interest (California Department of Fish and Game, 2007a).

One approach in MPA design has been to establish the size of MPAs based on the spatial scale of movements of adults of heavily fished species and to space MPAs based on scales of larval dispersal (Palumbi, 2004). However, guidelines for the minimum size of MPAs and no-take reserves, and spacing between adjacent MPAs, vary dramatically depending on the goals for the MPAs (Hastings and Botsford, 2003). Friedlander *et al.* (2003) suggested that no-take zones should measure ca. 10 km^2 to ensure viable populations of a range of species in the Seaflower Biosphere Reserve, Colombia. Airamé *et al.* (2003) recommended a network of three to five no-take zones in each biogeographic region of the Channel Islands National Marine Sanctuary, comprising approximately 30–50% of the area, in order to conserve biodiversity and contribute to sustainable fisheries in the region.

Recent studies confirm that larval dispersal is more localized than previously thought, and short-lived species may require regular recruitment from oceanographically connected sites (Cowen, Paris, and Srinivasan, 2006; Steneck, 2006). Palumbi (2003) concluded that marine reserves tens of km apart may exchange larvae in a single generation. Shanks, Grantham, and Carr (2003) similarly concluded that marine reserves spaced 20 km apart would allow larvae to be carried to adjacent reserves. The Science Advisory Team to California's Marine Life Protection Act Initiative recommended spacing high protection MPAs, such as marine reserves, within 50–100 km in order to accommodate larval dispersal distances of a wide range of species of interest. Halpern *et al.* (2006) corroborated these findings using an uncertainty-modeling approach.

No-take zones measuring a minimum of 20 km in diameter will accommodate short-distance dispersers in addition to including a significant part of the local benthic fishes, thus generating fisheries benefits (Shanks, Grantham, and Carr, 2003; Fernandes *et al.*, 2005; Mora *et al.*, 2006). While this recommendation is likely to protect the majority of small benthic fish and benthic invertebrates, it is unlikely to protect large pelagic fish and large migratory species (Roberts *et al.*, 2003b; Palumbi, 2004). Recommendations to protect highly migratory and pelagic species include designing MPAs to protect predictable breeding and foraging habits, ensuring these have dynamic boundaries and extensive buffers, and establishing dynamic MPAs that are defined by the extent and location of large-scale oceanographic features such as oceanic fronts where changes in types and abundances of marine organisms often occur (Hyrenbach, Forney, and Dayton, 2000).

A system-wide approach should be taken that addresses patterns of connectivity between ecosystems like mangroves, coral reefs, and seagrass beds (Mumby *et al.*, 2004). For example, mangroves in the Caribbean enhance the biomass of coral reef fish communities because they provide essential nursery habitat. Coral reefs can protect mangroves by buffering the impacts of wave erosion, while mangroves can protect reefs and seagrass beds from siltation. Thus, connectivity between functionally linked habitats is essential for maintaining ecosystem function and resilience (Ogden and Gladfelter, 1983; Roberts, 1996; Nagelkerken *et al.*, 2000). Entire ecological units (*e.g.*, coral reefs with their associated mangroves and seagrasses) should be included in MPA design where possible. If entire biological units cannot be included, then larger areas should be chosen over smaller areas to accommodate local-scale recruitment.

Although maintaining connectivity within and between MPAs is critical for maintaining marine biodiversity, ecosystem function, and resilience, many challenges exist. For example, the same currents and pathways that allow for larval recruitment following a disturbance event can expose an ecosystem to invasive species or pollutants, which can undermine the resilience of a system (McClanahan, Polunin, and Done, 2002). Numerous challenges also exist in estimating larval dispersal patterns. Although there have been detailed studies addressing dispersal *potential* of marine species based on their larval biology (*e.g.*, Shanks, Grantham, and Carr, 2003; Kinlan and Gaines, 2003), little is known about where in the oceans larvae go and how far they travel. A single network design is unlikely to satisfy the potential dispersal ranges for all species; Roberts *et al.* (2003b) recommended an approach using various sizes and spacing of MPAs in a network to accommodate the diversity of dispersal ranges. Larval duration in the plankton also varies from minutes to years, and the more time propagules spend in the water column, the farther they tend to be dispersed (Shanks, Grantham, and Carr, 2003; Steneck, 2006). Evidence from hydrodynamic models and genetic structure data indicates that in addition to large variation of larval dispersal distances among species, the average scale of dispersal can vary widely—even within a given species—at different locations in space and time (*e.g.*, Cowen *et al.*, 2003; Sotka *et al.*, 2004; Engie and Klinger, 2007). Some information suggests long-distance dispersal is common, but other emerging information suggests that larval dispersal may be limited (Jones *et al.*, 1999; Swearer *et al.*, 1999; Warner, Swearer, and Caselle, 2000; Thorrold *et al.*, 2001; Palumbi, 2003; Paris and Cowen, 2004; Jones, Planes, and Thorrold, 2005). Additional research will be required to better understand where and how far larvae travel in various marine ecosystems.

8.3.3.3 Replicate Multiple Habitat Types in MPA Networks

Recognizing that the science underlying resilience is developing and that climate change will not affect marine species equally everywhere, an element of spreading the risk must be built into MPA design. To avoid the loss of a single habitat type, managers can protect multiple samples of the full range of marine habitat types (Hockey and Branch, 1994; Ballantine, 1997; Roberts *et al.*, 2001; Friedlander *et al.*, 2003; Roberts *et al.*, 2003b; Salm, Done, and McLeod, 2006; Wells, 2006). For example, these marine habitat types include coral reefs with varying degrees of exposure to wave energy (*e.g.*, offshore, mid-shelf, and inshore reefs), seagrass beds, and a range of mangrove communities (riverine, basin, and fringe forests in areas of varying salinity, tidal fluctuation, and sea level) (Salm, Done, and McLeod, 2006). Reflecting the current federal goal of protecting at least 30% of lifetime stock spawning potential (Ault, Bohnsack, and Meester, 1998; National Marine Fisheries Service, 2003), it has been recommended that more than 30% of appropriate habitats should be included in no-take zones (Bohnsack *et al.*, 2002). In 2004, the Great Barrier Reef Marine Park Authority increased the area of no-take zones from less than 5% to approximately 33% of the area of the Marine Park, ensuring that at least 20% of each bioregion (area of every region of biodiversity) was zoned as no-take (Day *et al.*, 2002; Fernandes *et al.*, 2005).

For both terrestrial and marine systems, species diversity often increases with habitat diversity, and species richness increases with habitat complexity; the greater the variety of habitats protected, the greater the biodiversity conserved (Friedlander *et al.*, 2003; Carr *et al.*, 2003). High species diversity may increase ecosystem resilience by ensuring sufficient redundancy to maintain ecological processes and protect against environmental disturbance (McNaughton, 1977; McClanahan, Polunin, and Done, 2002). This is particularly true in the context of additive or synergistic stressors. Maximizing habitat heterogeneity is critical for maintaining ecological health, thus MPAs should include large areas and depth gradients (Done, 2001; Hansen, Biringer, and Hoffman, 2003; Roberts *et al.*, 2003a). By protecting a representative range of habitat types and communities, MPAs have a higher potential to protect a region's biodiversity, biological connections between habitats, and ecological functions (Day *et al.*, 2002).

Replication of habitat types in multiple areas provides a further way to spread risks associated with climate change. If a habitat type is destroyed in one area, a replicate of that habitat may survive in another area to provide larvae for recovery. While the number of replicates will be determined by a balance of desired representation and practical concerns such as funding and enforcement capacity (Airamé *et al.*, 2003), generally at least three to five replicates are recommended to effectively protect a particular habitat or community type (Airamé *et al.*, 2003; Roberts *et al.*, 2003b; Fernandes *et al.*, 2005). Wherever possible, multiple samples of each habitat type should be included in MPA networks or larger management frameworks such as multiple-use MPAs or areas under rigorous integrated management regimes (Salm, Done, and McLeod, 2006). This approach has the advantage of protecting essential habitat for a wide variety of commercially valuable fish and macroinvertebrates.

While a risk-spreading approach to address the uncertainty of the impacts of climate change makes practical sense, there are challenges to adequate representation. Managers must have access to classification maps of marine habitat types/communities or local knowledge of habitat types/communities for their area to determine which representative examples should be included in MPA design. Replication of habitat types may not always be feasible due to limited monitoring and enforcement resources, conflicting needs of resource users, and existence of certain habitat types within an MPA.

8.3.4 Integrate Climate Change Into MPA Planning, Management, and Evaluation

A number of tools exist to help managers address climate impacts and build resilience into MPA design and management. Ecological changes that are common in marine reserves worldwide and guidelines for marine reserve design are summarized in an educational booklet for policymakers, managers, and educators, entitled "The Science of Marine Reserves" (Partnership for Interdisciplinary Studies of Coastal Oceans, 2005). The Reef Resilience toolkit (The Nature Conservancy and Partners, 2004) provides marine resource managers with strategies to address coral bleaching and conserve reef fish spawning aggregations, helping to build resilience into coral reef conservation programs. "A Reef Manager's Guide to Coral Bleaching" (Marshall and Schuttenberg, 2006a) provides information on the causes and consequences of coral bleaching and management strategies to help local and regional reef managers reduce this threat to coral reef ecosystems. The application of some of these strategies is discussed in a recent report by the U.S. Environmental Protection Agency, which applies resilience theory in a case study for the reefs of American Samoa and proposes climate adaptation strategies that can be leveraged with existing local management plans, processes, and mandates (U.S. Environmental Protection Agency, 2007).

In contrast, with regard to the impacts on marine organisms of reductions in ocean pH because of CO_2 emissions (Caldeira and Wickett, 2003), management strategies have not yet been developed. Adding chemicals to counter acidification is not a viable option, as it would likely be only partly effective and, if so, only at a very local scale (The Royal Society, 2005). Therefore, further research is needed on impacts of high concentrations of CO_2 in the oceans, possible acclimation or evolution of organisms in response to changes in ocean chemistry, and how management might respond (The Royal Society, 2005).

Determining management effectiveness is important for gauging the success of an MPA or network, and also can inform adaptive management strategies to address shortcomings in a particular MPA or network. To help managers improve the management of MPAs, the IUCN World Commission on Protected Areas and the World Wide Fund for Nature developed an MPA management effectiveness guidebook. This guidebook, "How is Your MPA Doing? A Guidebook of Natural and Social Indicators for Evaluating Marine Protected Area Management Effectiveness," (Pomeroy, Parks, and Watson, 2004) helps managers and other decision-makers assess management effectiveness through the selection and use of biophysical, socioeconomic, and governance indicators. The goal of the guidebook is to enhance the capability for adaptive management in MPAs. The "Framework for Measuring Success" (Parks and Salafsky, 2001) also provides a suite of tools to analyze community response to an MPA, and replicable methodologies to assess both social and ecological criteria.

National marine sanctuaries are preparing a series of Condition Reports for each site, which provide a summary of resources, pressures on those resources, current condition and trends, and management responses to the pressures (National Marine Sanctuary Program, 2007c). This information is intended to be used in reviews of management plans and to help sanctuary staff identify monitoring, characterization, and research priorities to address gaps, day-to-day information needs, and new threats.

8.3.4.1 MPA Monitoring and Research

Marine protected areas must be effectively monitored to ensure the success of marine protected area design and management. If MPA design and management are not successful, then adaptations need to be made to meet the challenges posed by anthropogenic and natural stresses. As the number of pristine areas is decreasing rapidly, establishing baseline data for marine habitats is urgent and essential. Once baseline data are established, managers should monitor to determine the effects of climate change on local resources and populations. Retrospective testing of resistance to climate change impacts is difficult, so rapid response strategies should be in place to assess ecological effects of extreme events as they occur. For coral reefs, coral bleaching patterns either disappear with time or become confounded with other causes of mortality such as predation by the crown-of-thorns starfish, disease, or multiple other stressors (Salm, Done, and McLeod, 2006). Therefore, response strategies must be implemented immediately following a mass bleaching event or other climate-related event to determine bleaching impacts. For coral reefs, bleaching and mortality responses of corals to heat stress, the recovery rates of coral communities, and the resilience/resistance of certain corals to bleaching should be monitored.

Monitoring also can be an effective way to engage community members and raise awareness of the impacts of climate change on marine systems. For example, the Reef Check program enables community volunteers to collect coral reef monitoring data to supplement other monitoring data from researchers and government agencies. Programs that engage coral reef users (such as local fishermen and tourism operators) in monitoring can help raise awareness of impacts on marine systems and can help support the need to manage for local threats. The Nature Conservancy is managing the Florida Reef Resilience Program to develop strategies to improve the condition of Florida's coral reefs and support human dimensions investigations (The Nature Conservancy, 2007). The program includes annual surveys of coral bleaching effects at reefs along the Florida Keys and the southeast Florida coast using trained divers from agencies, universities, and non-governmental organizations.

Changes in ocean chemistry (CO_2 and O_2 levels and salinity), hydrography (sea level, currents, vertical mixing, storms, and waves), and temperature should be monitored over long time scales to determine climate changes and possible climate trends. This information could then be analyzed to determine the efficacy of MPAs now and in the future. Changes in sea temperature, sea level, and ocean chemistry will change species distributions.

NOAA's Coral Reef Watch program (National Oceanic and Atmospheric Administration, 2007a) provides products that can warn managers of potential impending bleaching events. In addition, Coral Reef Watch is developing bleaching forecasts that will provide outlooks of bleaching potential months in advance. These tools can help managers prepare for bleaching events so that when the event occurs, managers can have the necessary capacity in place to respond. In addition to a number of guides to help managers address resilience, global information databases exist that consolidate climate change impacts on marine systems such as coral reefs. Reefbase (The World Fish Center, 2007) is a global information system and is the database of the Global Coral Reef Monitoring Network and the International Coral Reef Action Network (ICRAN). Coral bleaching reports, maps, photographs, and publications are freely available on the website, and bleaching reports can be submitted for inclusion in the database. Reefbase provides an essential mechanism for collecting bleaching data from around the world, thus helping researchers and managers to identify potential patterns in reef vulnerability.

8.3.4.2 Social Resilience, Stakeholder Participation, and Education and Outreach

In addition to identifying and building ecological resilience into MPA design and management, it is equally important for managers to address social resilience (*i.e.*, social, economic, and political factors that influence MPAs and networks). Social resilience is the "ability of groups or communities to cope with external stresses and disturbances as a result of social, political and environmental change" (Adger, 2000). MPAs that reinforce social resilience can provide communities with the opportunity to strengthen social relations and political stability and diversify economic options (Corrigan, 2006). A variety of management actions have been identified to reinforce social resilience (Corrigan, 2006) including: 1) provide opportunities for shared leadership roles within government and management systems (Adger *et al.*, 2005; Cinner *et al.*, 2005; McClanahan *et al.*, 2006); 2) integrate MPAs and networks into broader coastal management initiatives to increase public awareness and support of management goals (Marshall and Schuttenberg, 2006a; U.S. Environmental Protection Agency, 2007); 3) encourage local economic diversification so that communities are able to deal with environmental, economic, and social changes (Adger *et al.*, 2005; Marschke and Berkes, 2006); 4) encourage stakeholder participation and incorporate their ecological knowledge in a multi-governance system (Tompkins and Adger, 2004; Granek and Brown, 2005; Lebel *et al.*, 2006); and 5) make culturally appropriate conflict resolution mechanisms accessible to local communities (Christie, 2004; Marschke and Berkes, 2006).

Some MPA managers may feel that engaging in supporting human adaptive capacity to climate change impacts is beyond the scope of their work. However, it is important to recognize that resource use patterns will change in response to changing environmental conditions. For example, recent studies suggest that when fishers are meaningfully engaged in natural resource management decision-making processes, their confidence and social resilience to changes in resource access can be increased (Marshall, In Press). Furthermore, as management is adapted to address changing conditions, engagement with stakeholders during this process will help MPA managers build the alliances, knowledge, and influence needed to implement adaptive approaches (Schuttenberg and Marshall, 2007). For example, national marine sanctuaries have Sanctuary Advisory Councils comprised of a wide range of stakeholder representatives, which provide advice to sanctuary managers and help develop sanctuary management plans (National Marine Sanctuary Program, 2007b). Education and outreach programs can help inform the public about effects of climate change on marine ecosystems and the pressing need to ameliorate existing stressors in coastal waters.

8.4 Case Studies

This section includes three U.S. case studies along with an Australian case study for comparison. Each case study discusses existing management approaches, threats of climate change, and adaptation options. The case studies are located in Florida (Florida Keys National Marine Sanctuary), Australia (Great Barrier Reef Marine Park), Hawaii (Papahānaumokuākea Marine National Monument), and California (Channel Islands National Marine Sanctuary). These MPAs range in size, species composition, and levels of protection; no-take designations, for example, are 6% (FKNMS), 10% (CINMS), 33% (GBRMP), and 100% (PMNM).

8.4.1 Case Study: the Florida Keys National Marine Sanctuary

8.4.1.1 Introduction

The Florida Keys are a limestone island archipelago extending southwest over 320 km from the southern tip of the Florida mainland (Fig. 8.3). The Florida Keys National Marine Sanctuary (FKNMS) surrounds the Florida Reef Tract, one of the world's largest systems of coral reefs and the only bank-barrier reef in the coterminous United States. The FKNMS is bounded by and connected to Florida Bay, the Southwest Florida Continental Shelf, and the Straits of Florida and Atlantic Ocean. It is influenced by the powerful Loop Current/Florida Current/Gulf Stream system to the west and south, as well as a weaker southerly flow along the West Florida Shelf (Lee *et al.*, 2002). The combined Gulf of Mexico and tropical Atlantic biotic influences make the area one of the most diverse in North America.

The uniqueness of the marine environment and ready access from the mainland by a series of bridges and causeways draws millions of visitors to the Keys, including many from the heavily populated city of Miami and other metropolitan areas of South Florida. Also, in recent years Key West has become a major destination for cruise liners, attracting more than 500 stop-overs annually. The major industry in the Florida Keys has become tourism, including dive shops, charter fishing, and dive boats and marinas as well as hotels and restaurants. There also is an important commercial fishing industry.

National Marine Sanctuaries established at Key Largo in 1975 and Looe Key in 1981 demonstrated that measures to protect coral reefs from direct impacts could be successful using management actions such as mooring buoys, education programs, research and monitoring, restoration efforts, and proactive, interpretive law enforcement. In 1989, mounting threats to the health and ecological future of the coral reef ecosystem in the Florida Keys prompted Congress to take further protective steps. The threat of oil drilling in the mid- to late-1980s off the Florida Keys, combined with reports of deteriorating water quality throughout the region, occurred at the same time as adverse effects of coral bleaching (Causey, 2001), the Caribbean-wide die-off of the long-spined urchin (Lessios, Robertson, and Cubit, 1984), loss of living coral cover on reefs (Porter and Meier, 1992), a major seagrass die-off (Robblee *et al.*, 1991), declines in reef fish populations (Bohnsack, Harper, and McClellan, 1994; Ault, Bohnsack, and Meester, 1998), and the spread of coral diseases (Kuta and Richardson, 1996). These were already topics of major scientific concern and the focus of several scientific workshops when, in the fall of 1989, three large ships ran aground on the Florida Reef Tract within a brief 18-day period. On November 16, 1990, President Bush signed into law the Florida Keys National Marine Sanctuary and Protection Act. Specific regulations to manage the sanctuary did not go into effect until July 1997, after the final management plan (U.S. Department of Commerce, 1996) had been approved by the Secretary of Commerce and the Governor and Cabinet of the State of Florida. The FKNMS encompasses approximately 9,800 km^2 of coastal and oceanic waters surrounding the Florida Keys (Keller and Causey, 2005) (Fig. 8.3), including the Florida Reef Tract, all of the mangrove islands of the Florida Keys, extensive seagrass beds and hard-bottom areas, and hundreds of shipwrecks.

Millions of visitors come to the Keys each year. Visitors spent $1.2 billion (Leeworthy and Wiley, 2003) over 12.1 million person-days (Johns *et al.*, 2003) in the Florida Keys between June 2000 and May 2001. Over that period, visitors and residents spent 5.5 million of the person-days on natural and artificial reefs. Significantly, visitors (and residents) perceive significant declines in the quality of the marine environment of the Keys (Leeworthy, Wiley, and Hospital, 2004).

8.4.1.2 Specific Management Goals and Current Ecosystem Stressors Being Addressed

Goal and Objectives of the Florida Keys National Marine Sanctuary

The goal of the FKNMS is “To preserve and protect the physical and biological components of the South Florida estuarine and marine ecosystem to ensure its viability for the use and enjoyment of present and future generations” (U.S. Department of Commerce, 1996). The Florida Keys National Marine Sanctuary and Protection Act as well as the Sanctuary Advisory Council identified a number of objectives to achieve this goal (Box 8.6).

Coral Reef and Seagrass Protection

The management plan (U.S. Department of Commerce, 1996) established a channel and reef marking program that coordinated federal, state, and local efforts to mark channels and shallow reef areas. These markers help prevent damage from boat groundings and propeller-scarring.

A mooring buoy program is one of the most simple and effective management actions to protect sanctuary resources from direct impact by boat anchors. By installing mooring buoys in high-use areas, the sanctuary has prevented damage to coral from the thousands of anchors dropped every week in the Keys.

Marine Zoning

The management plan implemented marine zoning with five categories of zones. The relatively large "no-take" Ecological Reserve at Western Sambo (Fig. 8.3) was designed to help restore ecosystem structure and function. A second Ecological Reserve was implemented in the Tortugas region in 2001 as one of the largest no-take areas in U.S. waters (U.S. Department of Commerce, 2000; Cowie-Haskell and Delaney, 2003; Delaney, 2003). In addition to the larger Ecological Reserves, there are 18 small, no-take Sanctuary Preservation Areas (SPAs) that protect over 65% of shallow, spur and groove reef habitat. These areas displaced few commercial and recreational fishermen and resolved a user conflict with snorkeling and diving activities in the same shallow reef areas. Four small Research-Only Areas are also no-take; only scientists with permits are allowed access.

In addition, 27 Wildlife Management Areas (WMAs) were established to address human impacts to nearshore habitats such as seagrass flats and mangrove-fringed shorelines. Most of these WMAs only allow no-motorized access. Finally, because the FKNMS Act called for the two existing sanctuaries to be subsumed by the FKNMS, a final type of marine zone, called Existing Management Areas, was used to codify both Key Largo and Looe Key NMS regulations into FKNMS regulations. This was a way to maintain the additional protective resource measures that had been in effect for the Key Largo and Looe Key NMSs since 1975 and 1981, respectively. Those areas prohibited spearfishing, marine life collecting, fish trapping, trawling, and a number of other specific activities that posed threats to coral reef resources.

Improvement of Water Quality

The FKNMS Act directed the U.S. Environmental Protection Agency to work with the State of Florida and NOAA to develop a Water Quality Protection Program (WQPP) to address water quality problems and establish corrective actions. The WQPP consists of four interrelated components: 1) corrective actions that reduce water pollution directly by using engineering methods, prohibiting or restricting certain activities, tightening existing regulations, and increasing enforcement; 2) monitoring of water quality, seagrasses, and coral reefs to provide information about status and trends in the sanctuary; 3) research to identify and understand cause-and-effect relationships involving pollutants, transport pathways, and biological communities; and 4) public education and outreach programs to increase public awareness of the sanctuary, the WQPP, and pollution sources and impacts on sanctuary resources.

Research and Monitoring

The FKNMS management plan established a research and monitoring program that focused research on specific management needs. In 2000, staff convened a panel of external peers to review the sanctuary's science program and provide recommendations for improvements (Florida Keys National Marine Sanctuary, 2007). Based on the panel's recommendation that sanctuary managers identify priority research needs, staff prepared a Comprehensive Science Plan to identify priority areas (Florida Keys National Marine Sanctuary, 2002). A second review of the science program is being conducted during 2007.

The three monitoring projects of the WQPP (Fish and Wildlife Research Institute, 2007) are developing baselines for water quality, seagrass distribution and abundance, and coral cover, diversity, and condition. Such a baseline of information is particularly important to have as the Comprehensive Everglades Restoration Plan (CERP) is implemented (U.S. Army Corps of Engineers, 2007) just north of the FKNMS. The CERP is designed so that managers can be adaptive to ecological or hydrological changes that are taking place within or emanating from the Everglades, with possible positive or negative influences on communities in the FKNMS (Keller and Causey, 2005).

Additional monitoring is done for the Marine Zone Monitoring Program, which is designed to detect changes in populations, communities, and human dimensions resulting from no-take zoning (Keller and Donahue, 2006). Coupled with environmental monitoring using data buoys (National Oceanic and Atmospheric Administration, 2006b), routine cruises (National Oceanic and Atmospheric Administration, 2007c), and remote sensing (NOAA Coast Watch Program, 2007), the FKNMS is a relatively data-rich environment for detecting presumptive climate change effects.

Education and Outreach

The management plan for the FKNMS includes an education and outreach program that lays out ways that education efforts can directly enhance the various programs to protect sanctuary resources. Public awareness and understanding are essential to achieve resource protection through cooperation and compliance with regulations.

Regulations and Enforcement

The FKNMS management plan includes regulations that have helped managers protect resources of the sanctuary while having the least amount of impact on those who enjoy and utilize sanctuary resources in a conscientious way. In order to maximize existing enforcement programs, the management plan contains an enforcement plan that has served to help focus enforcement on priority problems within the sanctuary. The program also coordinates all the enforcement agencies in the Keys. Enforcement complements education and outreach in efforts to achieve compliance with regulations.

8.4.1.3 Potential Effects of Climate Change on Management

Coral Bleaching

The potential effects of climate change on coral reefs are generally well known (*e.g.*, Smith and Buddemeier, 1992; Hoegh-Guldberg, 1999; Buddemeier, Kleypas, and Aronson, 2004; Hoegh-Guldberg, 2004; Sheppard, 2006), but the fate of individual reef systems such as the Florida Reef Tract will vary based on a combination of factors related to history, geography, and an understanding of processes that explain the patchiness of coral bleaching and subsequent mortality that occurs on reefs. Coral bleaching was first reported in the Florida Keys in 1973 (Jaap, 1979), with at least seven other episodes documented prior to 2000 (Causey, 2001) and a major bleaching event in 2005 that also affected the Caribbean (Miller *et al.*, 2006; Donner, Knutson, and Oppenheimer, 2007). Unfortunately, before-during-and-after sampling has not been conducted during major bleaching events in the Florida Keys, which makes assumptions about coral mortality caused by bleaching at best correlative. Hurricanes are an especially confounding factor when they occur during bleaching years, as they did in 1997–98 and 2005. Still, anecdotal evidence suggests that large numbers of corals were killed in 1997–98 when corals remained bleached for two consecutive years (Causey, 2001). Long-term temperature records do not exist that reveal trends of increasing surface seawater temperature for the Florida Keys, but Williams, Jackson, and Kutzbach (2007), using climate models and IPCC greenhouse gas estimates to forecast how climate zones may change in the next 100 years, identified the southeastern United States as a region with the greatest likelihood of developing novel regional climate conditions that would be associated with temperature increases of several degrees. The consequences of such changes on coral reefs in Florida will be dramatic unless significant adaptation or acclimatization occurs.

Governments and agencies have responded to the crisis of coral bleaching with detailed management plans (Westmacott *et al.*, 2000; Marshall and Schuttenberg, 2006b), workshops to develop strategies that support response efforts (Salm and Coles, 2001), and research plans (Marshall and Schuttenberg, 2006b; Puglise and Kelty, 2007). Two themes have emerged from these efforts. First, effort is needed at local and regional levels to identify and protect bleaching-resistant sites—if they exist. Second, management plans should be developed or modified in the case of the FKNMS to restore or enhance the natural resilience (Hughes *et al.*, 2003; West and Salm, 2003) of coral reefs.

Response plans to coral bleaching events depend upon increasingly accurate predictions to help guide resource assessment and monitoring programs, and the NOAA Coral Reef Watch program has increasingly accurate capability to predict the severity, timing, and geographic variability of mass bleaching events, largely using remote sensing technologies (NOAA Satellite and Information Service, 2007). Scientists and managers in Florida have not fully implemented an assessment and monitoring program that specifically addresses bleaching events, including the critical before-during-after sampling that is necessary to quantify the distribution, severity, and consequences of mass bleaching. While such monitoring programs do nothing to prevent coral bleaching, they do provide data that may identify bleaching-resistant sites that, if not already protected, can be considered high priority for management action and protection against local stressors.

Currently in Florida, status and trends monitoring has identified habitat types with higher than average coral cover and abundance, but it is unknown whether these areas are more or less prone to bleaching because only baseline assessments have been conducted (Miller *et al.*, 2005). Deeper reefs (to 35 meters) may also exhibit less evidence of mortality caused by coral bleaching (Miller *et al.*, 2001), but even less is known about these habitats—especially related to the distribution and abundance of coral diseases, which can confound assessments of factors causing mortality because the temporal scale of monitoring is sufficient to only assess disease prevalence and not incidence or mortality rates.

No-Take Protection and Zoning for Resistance or Resilience

The use of marine reserves (Sanctuary Preservation Areas, Research-Only Areas, and Ecological Reserves) in the Florida Keys National Marine Sanctuary has already been adopted as a tool to manage multiple user groups throughout the Sanctuary (U.S. Department of Commerce, 1996), and in the Dry Tortugas to enhance fisheries where positive results have been obtained after only a few years (Ault *et al.*, 2006). Potential exists to use a range of options to identify bleaching resistant reefs in the Keys, from simply identifying the best remaining sites left and using a decision matrix based on factors that may confer resilience to establish priority sites for protection, to the Bayesian approach of Wooldridge and Done (2005). Only recently have coral community data been obtained at the relevant spatial scales and across multiple habitat types (Smith *et al.*, In Press). Whatever approach is used, the results are likely to include sites with high coral cover and abundance, high diversity, connectivity related to current regimes with the potential to transport larvae, and protection from local stressors including overfishing and pollution (Done, 1999; Salm, Smith, and Llewellyn, 2001; West, 2001; Hughes *et al.*, 2003).

Interestingly, the theoretical framework that links protection against overfishing (to restore herbivores that then reduce algae that kill corals or prevent recruitment) using no-take marine reserves and the cascading effects that result and link to improved coral condition is hotly debated (Jackson *et al.*, 2001; Grigg *et al.*, 2005; Pandolfi *et al.*, 2005; Aronson and Precht, 2006). This is perhaps surprising because of the strong intuitive sense such arguments make, but reserves also protect predators, so declines in herbivorous fish might occur, as opposed to increases. Also, data from field studies provide conflicting results on the role of herbivores. Mumby *et al.* (2006) showed that increased densities of herbivorous fish in a marine reserve reduced algal growth after mass bleaching caused extensive coral mortality, but such herbivore densities do not always increase after protection is provided (Mosquera *et al.*, 2000; Graham, Evans, and Russ, 2003; Micheli *et al.*, 2004; Robertson *et al.*, 2005). Further, there is widespread belief that the mass mortality of *Diadema antillarum*—a major grazer on reefs—in 1983–1984 was a significant proximal cause of coral reef decline throughout the Caribbean. However, as reported in Aronson and Precht (2006) half the coral reef decline throughout the Caribbean reported by Gardner et al. (2003) occurred before the die-off of *D. antillarum*, and immediately after the die-off coral cover remained unchanged (Fig. 8.5) (Gardner *et al.*, 2003). Subsequent declines in cover throughout the region were due to coral bleaching (1987, 1997–1998) and disease. It is important to highlight this complexity because it emphasizes how much is unknown about basic ecological processes on coral reefs and consequently how much needs to be learned about whether no-take marine reserves work effectively to enhance resilience when disease and bleaching remain significant sources of coral mortality (Aronson and Precht, 2006).

Figure 8.5. Total observed change in coral cover (%) across the Caribbean basin over the past 25 years (Gardner *et al.*, 2003). A. Coral cover (%) 1977-2001. Annual estimates (▲) are weighted means with 95% bootstrap confidence intervals. Also shown are unweighted estimates (●), unweighted mean coral cover with the Florida Keys Coral Reef Monitoring Project (1996-2001) omitted (x), and the number of studies each year (○). B. Year-on-year rate of change (mean $\Delta N \pm SE$) in coral cover (%) for all sites reporting two consecutive years of data 1975-2000 (●) and the number of studies for each two-year period (○).

In the Florida Keys, marine protected areas date to 1960 for John Pennekamp Coral Reef State Park, 1975 for the Key Largo National Marine Sanctuary, 1981 for Looe Key National Marine Sanctuary, and 1990 for expansion of these sites to include 2,800 square nautical miles of coastal waters that are now designated as the Florida Keys National Marine Sanctuary. The Tortugas

Ecological Reserve was added in 2001, and six years later a 46-square-mile Research Natural Area was also established within Dry Tortugas National Park (National Park Service, 2007). While spatial resolution among habitat types from Miami to the Dry Tortugas is not as extensive as in the Great Barrier Reef, work similar to Wooldridge and Done (2005) should be evaluated for application to the Florida Keys. For example, a combination of retrospective sea-surface temperature studies using NOAA Coral Reef Watch products, combined with *in situ* temperature data, water quality monitoring data (e.g. Boyer and Briceño, 2006), and detailed site characterizations (Miller, Swanson, and Chiappone, 2002) might help identify bleaching-resistant sites (if temporally- and spatially-relevant sampling is conducted before, during, and after a bleaching event), identify candidate sites for protection based on resilience criteria, and in general validate the concept of marine reserve networks in the region as a management response to coral bleaching threats.

Geographic Range Extensions of Coral Reefs in Florida

Coral reefs in south Florida represent the northern geographic limit of reef development in the United States. It is reasonable to assume that some northward expansion of either the whole reef community or individual species may occur as a result of warming climate. Indeed, such a northward expansion may already be in progress, but caution is necessary before assigning too much significance to what might be an anomalous event. Specifically, *Acropora cervicornis* was discovered growing in large thickets off Fort Lauderdale in 1998 (Vargas-Ángel, Thomas, and Hoke, 2003) and *A. palmata* was discovered off Pompano Beach in northern Broward county (Precht and Aronson, 2004). It is possible that these populations—over 50 km northward of their previously known northern limit—are a result of recent climate warming known to have occurred in the western Atlantic (Hoegh-Guldberg, 1999; Levitus *et al.*, 2000; Barnett, Pierce, and Schnur, 2001). It is also possible that these reefs represent a remnant population or a chance recruitment event based on a short-term but favorable set of circumstances that will disappear with the next hurricane, cold front, disease epidemic, or bleaching event. Still, the presence of these acroporid reefs is suggestive of what might happen as climate warms. Interestingly, the presence of these northern acroporid populations matches the previous northern extension of reef development in the region during the middle Holocene (Lighty, Macintyre, and Stuckenrath, 1978), when sea surface temperatures were warmer. Reefs up to 10 m thick grew off Palm Beach County in the middle Holocene (Lighty, Macintyre, and Stuckenrath, 1978) and when temperatures started to cool 5,000 years before present reef development moved south to its current location (Precht and Aronson, 2004).

Despite these northern extensions in the geographic distributions of corals seen in the fossil record, predicting future geographic expansions in Florida is complicated by factors other than temperature that influence coral reefs, including light, carbonate saturation state, pollution, disease (Buddemeier, Kleypas, and Aronson, 2004), and a shift from a carbonate to siliciclastic sedimentary regime along with increasing nutrient concentrations as latitude increases up the east coast of Florida (Precht and Aronson, 2004). One thing, however, is certain: geographic shifts of reefs in Florida that result from global warming will not mitigate existing factors that today cause widespread local and regional coral reef decline (Precht and Aronson, 2004). Further, if we assume that the reefs of the mid-Holocene were in better condition than today's reefs, they may not prove to be a good analogue for predicting the future geographic trajectory of today's reefs. Because corals in Florida are already severely impacted by disease, bleaching, pollution, and overfishing, expansion at best will be severely limited compared to what might occur if the ecosystem were intact.

At the global scale and across deep geological time, range extensions to higher latitudes occurred for hard corals that survived the Cretaceous warming period (Kiessling, 2001; Kleypas, 2006), and some coral species today that are found in the Red Sea and Persian Gulf can survive under much greater temperature ranges than they experience throughout the Indo-Pacific (Coles and Fadlallah, 1991). Both of these examples, however, probably reflect long-term adaptation by natural selection and not short-term acclimatization (Kleypas, 2006). At shorter times scales (decades), corals that survive rapid climate warming may be those that are able to quickly colonize and survive at higher latitudes where maximum summer temperatures may be reduced compared to their previous geographic range. An alternative to migration is the situation where corals adapt to increasing temperatures at ecological time scales (decades), and there is some evidence to suggest that this might occur (Guzmán and Cortés, 2001; Podestá and Glynn, 2001). However, the ability to predict if corals will acclimate is complicated because absolute values and adaptive potential are likely to vary across species (Ware, 1997; Hughes *et al.*, 2003; Kleypas, 2006). Acclimation without range expansion is a topic of great significance related to coral bleaching.

Another question related to the potential for coral reef migration to higher latitudes in Florida is related to understanding factors that currently limit expansion northward. Cold-water temperature tolerances for individual corals are not well known; however, their present-day global distribution generally follows the 18 $^{\circ}$C monthly minimum seawater isotherm (Kleypas, McManus, and Mendez, 1999; Kleypas, Buddemeier, and Gattuso, 2001; Buddemeier, Kleypas, and Aronson, 2004). South Florida is located between the 18 and 20°C isotherm and is thus significantly affected by severe winter cold fronts, especially for corals in shallow water (Jones, 1977; Burns, 1985; Walker, Rouse, and Huh, 1987). Well documented coral die-offs due to cold water fronts have occurred repeatedly throughout the Florida Keys (Davis, 1982; Porter, Battey, and Smith, 1982; Walker *et al.*, 1982; Roberts, Rouse, and Walker, 1983; Shinn, 1989); and as far south as the Dry Tortugas (Porter, Battey, and Smith, 1982; Jaap and Hallock, 1990; Jaap and Sargent, 1994). Porter and Tougas (2001) documented a decreasing trend in generic coral diversity along the east coast of Florida, but a number of coral species extend well beyond the 18°C isotherm with at least two species surviving as far north as North Carolina, likely due to the influence of the Gulf Stream. Thus, climate warming that has the potential to influence the impact of winter cold fronts may influence the range expansion of corals in Florida.

Finally, the above examples have focused mostly on the acroporid corals, which represent only two species out of more than forty that are found regionally (Jaap, 1984). Obviously, when considering range expansion of the total reef system, and not just two coral species, models designed to optimize or anticipate management actions that conserve existing habitat or predict future locations for habitat protection are likely to be exceedingly complicated. In Florida, assuming that reefs remain in sufficiently good condition to act as seed populations for range expansion, one management action to anticipate the effects of climate change would be to protect habitats similar to those that thrived during the middle Holocene when coral reefs flourished north of their current distribution (Lighty, Macintyre, and Stuckenrath, 1978). However, existing declines in the acroporids throughout Florida and the Caribbean (Gardner *et al.*, 2003; Precht and

Miller, 2006) suggest that at least for these two species, the major framework building species in the region, expansion will not occur unless factors such as disease and coral bleaching are mitigated.

8.4.2 Case Study: The Great Barrier Reef Marine Park

8.4.2.1 Introduction

The Great Barrier Reef (GBR) is a maze of reefs and islands spanning an area of 348,000 km^2 off the Queensland coast in northeast Australia (Fig. 8.6). It spans 14 degrees of latitude, making it the largest coral reef ecosystem in the world and one of the richest in biological diversity. The GBR supports 1,500 species of fish, 350 species of hard corals, more than 4,000 species of mollusks, 500 species of algae, six of the world's seven species of marine turtles, 24 species of seabirds, more than 30 species of whales and dolphins, and the dugong. The GBR was chosen as a case study because it is a large marine protected area that has moderate representation of no-take areas (33%) and has been under a management regime since 1975.

Figure 8.6. Map of the Great Barrier Reef Marine Park showing the adjacent catchment in Queensland. Modified from Haynes (2001) and courtesy of the Great Barrier Reef Marine Park Authority.

The GBR already appears to have been affected by climate change. The first reports of coral bleaching in the GBR appeared in the literature in the 1980s (Oliver, 1985) and have continued to increase in frequency since then (Hoegh-Guldberg, 1999; Done *et al.*, 2003). Coral-coring work done at the Australian Institute of Marine Science detected the earliest growth hiatus associated with mass coral bleaching in 1998 (Lough, 2007). There have been nine bleaching events on the GBR, with three major events in the last decade correlating with elevated sea temperatures and causing damage to parts of the reef. These early signs of climate change, and the extensive research and monitoring data that are available for the GBR, make it a suitable case study for this report.

The conservation values of the GBR are recognized in its status as a World Heritage Area (listed in 1981), and its resources are protected within the Great Barrier Reef Marine Park. The enactment of the Great Barrier Reef Marine Park Act in 1975 established the legal framework for protecting these values. The goal of the legislation is "…*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.*"

8.4.2.2 Managing the Great Barrier Reef Marine Park

The Great Barrier Reef Marine Park Authority has management strategies in place to address current stresses on the GBR. Stressors include terrestrial inputs of sediment, nutrients, and pesticides from coastal catchments; fisheries extraction; tourism and recreational activities; and changes to coastal hydrology as a result of coastal development and climate change. Sustainability of the environmental and social values of the Great Barrier Reef depend largely (and in most cases, entirely) on a healthy, self-perpetuating ecosystem. Reducing pressures on this system has been a focus of management activities over the last decade.

The Great Barrier Reef Marine Park was rezoned in 2003 to increase the area of highly protected no-take zones to 33%, with at least 20% protected in each habitat bioregion. These no-take areas aim to conserve biodiversity, increasing the potential of maintaining an intact ecosystem, with larger no-take areas including more representative habitats (Day *et al.*, 2002; Day *et al.*, 2004).

Current Approaches to Management

There are 26 major catchments that drain into the GBR (Fig. 8.6) covering an area of 425,964 km^2. Cropping (primarily of sugar cane), grazing, heavy industry and urban settlement are the main land uses. The *Reef Water Quality Protection Plan* (The State of Queensland and Commonwealth of Australia, 2003) is a joint state and federal initiative that aims to *halt and reverse the decline in the quality of water entering the Reef by 2013*. Under this initiative, diffuse sources of pollution are targeted through a range of voluntary and incentive-driven strategies to address water quality entering the GBR from activities in the catchments.

Important commercial fisheries in the GBR include trawling that mainly targets prawns and reef-based hook-and-line that targets coral trout and sweetlip emperor, inshore fin fish, and three crab fisheries (spanner, blue, and mud). None of these fisheries is considered overexploited; however, there is considerable unused (latent) effort in both the commercial and recreational sectors. Commercial fisheries contribute A$251 million to the Australian economy (Access Economics and Vecchia, 2007). Fisheries management is undertaken by the Queensland Government and includes a range of measures such as limited entry, management plans, catch and effort limits, permits, and industry accreditation. Recreational activities (including fishing) contribute A$623 million per annum to the region (Access Economics and Vecchia, 2007), and recreational fishing is subject to size and bag limits for many species.

Over 1 million tourists visit the GBR annually, contributing A$6.1 billion to the Australian economy (Access Economics and Vecchia, 2007). The Great Barrier Reef Marine Park Authority manages tourism using permits, zoning, and other planning tools such as management plans and site plans (Smith *et al.*, 2004). Visitation is concentrated in the Cairns and Whitsunday Island areas, and an eco-certification program encourages best practices and sustainable tourism (Skeat, 2003).

As one of the fastest growing regions in Australia, the GBR coast is being extensively developed through the addition of tourist resorts, urban subdivisions, marinas, and major infrastructure such as roads and sewage treatment plants. All levels of government regulate coastal development depending on the scale and potential impacts of the development. Local government uses local planning schemes and permits, state government uses the Integrated Planning Act (1997), and in the case of significant developments, the federal government uses the Environment Protection and Biodiversity Conservation Act (1999) to assess the environmental impacts of proposals. These efforts have resulted in an increase in biodiversity protection, a multi-stakeholder agreement to address water quality, and a well-managed, multiple-use marine protected area.

Vulnerability of the Great Barrier Reef to Climate Change

Despite these landmark initiatives, the ability of the ecosystem to sustain provision of goods and services is under renewed threat from climate change (Wilkinson, 2004). Climate change is rapidly emerging as one of the most significant challenges facing the GBR and its management. While MPA managers cannot directly control climate, and climate change cannot be fully averted, there is an urgent need to identify possibilities for reducing climate-induced stresses on the GBR (Marshall and Schuttenberg, 2006b). The GBR Climate Change Response Program has undertaken an assessment of the vulnerability of the GBR to climate change and is developing strategies to enhance ecosystem resilience, sustain regional communities and industries that rely on the GBR, and provide supportive policy and collaborations.

The Climate Change Response Program used regional GBR climate projections to assess the vulnerability of species, habitats, and key processes to climate change. Some relevant projections emerged. Regional GBR sea temperatures have increased by 0.4^{o}C since 1850 and are projected to increase by a further 1–3^{o}C above present temperatures by 2100 (Fig. 8.7). Sea level rise is projected to be 30–60 cm by 2100, and ocean chemistry is projected to decrease in pH by 0.4–0.5 units by 2100 (Lough, 2007). There is less certainty about: changes to tropical cyclones, with a 5–12% increase in wind speed projected; rainfall and river flow, with projected increases in intensity of droughts and rainfall events; and ENSOs, which will continue to be a source of high interannual variability (Lough, 2007).

Figure 8.7. Sea surface temperature (SST) projections for the Great Barrier Reef (GBR) (Lough, 2007).

Coral Bleaching

The key threats to the GBR ecosystem from climate change manifest in impacts to all components of the ecosystem, from species to populations to habitats and key processes. Although coral reefs represent only 6% of the Great Barrier Reef, they are an iconic component of the system and support a diversity of life. Unusually warm summers caused significant coral bleaching events in the GBR in 1998, 2002, and 2006. More than 50% of reefs were affected by bleaching in the summers of 1998 and 2002, following persistent high sea temperatures throughout the GBR. Fortunately, temperatures cooled soon enough to avoid catastrophic impacts, yet approximately 5% of reefs suffered long-term damage in each year. Stressful temperatures were confined to the southern parts of the GBR in the summer of 2006 and persisted long enough to cause over 40% of the corals to die. Future warming of the world's oceans is projected to increase the frequency and severity of coral bleaching events, making further damage to the GBR inevitable (Hoegh-Guldberg *et al.*, 2007). Continued monitoring efforts—such as those proposed in the GBR Coral Bleaching Response Plan—will be essential for understanding this ecosystem change.

Impacts to Species

Mass mortalities of seabirds and failures of nesting (death of all chicks) have been observed at several key seabird rookeries during anomalously warm summers on the GBR (coinciding with mass coral bleaching). New research is showing that provisioning failure, resulting when adults have to travel too far to find food for their chicks, causes these deaths (Congdon *et al.*, 2007).

This is thought to be due to decreased availability of food fish caused by changes in circulation patterns (location and depth of cool water bodies preferred by these fish). Marine turtles are also at risk from climate change, with increasing air temperatures projected to alter the gender ratio of turtle hatchlings; during periods of extremely high temperatures in the past, complete nesting failures have been observed. Sea level rise also poses a threat to seabirds and turtles, as nesting islands and beaches become inundated and suitability of alternative beaches is reduced by coastal development.

Fish, shark, and ray populations will be most affected by reductions in reef habitat, with resultant decreases in diversity and abundance and changes in community composition (Munday *et al.*, 2007; Chin *et al.*, 2007). Conversely, small increases in sea temperature may benefit larval fish by accelerating embryonic and larval growth and enhancing larval swimming ability. This shows that climate change will not affect all organisms equally, and some populations or groups (such as macroalgae) may in fact benefit by increasing their range or growth rate. However, this will change the distributions of species as they migrate southward or offshore. This in turn would likely result in population explosions of fast growing, ‘weed-like’ species to the detriment of other species, thereby reducing species diversity. As species and habitats decline, so too does the productivity of the system and its ability to respond to future change.

Impacts to Key Processes

The reef matrix itself is at risk from climate change through loss of coral—not only from coral bleaching but also physical damage from more intense storms and cyclones and reduced coral calcification rates as ocean pH decreases. This is critical from the perspective of the structural integrity of the GBR as well as the services reefs provide to other organisms, such as habitat and food.

Primary productivity, through changes to microbial, plankton, and seagrass communities, is likely to be affected as changes in the carbon cycle occur. Changes in rainfall patterns, runoff, and sea temperature also are likely to change plankton, seagrass, and microbial communities. These changes reduce trophic efficiency, which decreases food quality and quantity for higher trophic levels with a resultant decline in abundance of animals at higher trophic levels. Productivity is also likely to be sensitive to changes in ocean circulation as nutrient transport patterns change, thereby reducing nutrient availability and primary production.

Connectivity is at risk from changes to ocean circulation patterns and ENSO; as ocean currents and upwelling are affected, so too will be the hydrological cycles that transport material latitudinally and across the shelf. Connectivity will also be affected by coastal changes such as sea level rise and altered rainfall regimes, which are likely to have the most influence on coastal connectivity between estuaries and the inshore lagoon of the GBR. As temperature-induced stratification reduces wind-driven upwelling, offshore hydrological cycles are affected, potentially reducing connectivity between offshore reefs. All these changes could interact to affect the survival and dispersal patterns of larvae between reefs.

As biodiversity and connectivity are lost, the system becomes less complex, which initiates a cascade of events that results in long-term change. Simplified systems are generally less resilient and therefore less able to absorb shocks and disturbances while continuing to maintain their original levels of function. Reducing biodiversity and connectivity reduces the number of components and networks that can buffer against poor water quality, overfishing, and climate change. Maintaining a healthy ecosystem requires that ecological processes be preserved and that there is sufficient biodiversity to respond to changes. Larger marine protected areas that include representative habitats and protect biodiversity and connectivity will be more resilient to climate change into the future (Roberts *et al.*, 2006).

8.4.2.3 Adapting Management to Climate Change

In the face of these potential climate change impacts, the GBR Climate Change Response Program developed a Climate Change Action Plan in 2006. The action plan has five main objectives:

1. Address climate change knowledge gaps
2. Communicate with and educate communities about climate change implications for the GBR
3. Support greenhouse gas emissions mitigation strategies in the GBR region
4. Enhance resilience of the GBR ecosystem to climate change
5. Support GBR communities and industries to adapt to climate change

Key strategies within the action plan include assessing the vulnerability of the GBR ecological and social systems to climate change; developing an agency-wide communication strategy for climate change; facilitating greenhouse gas emissions reductions using the Reef Guardian incentive project; undertaking resilience mapping for the entire GBR and reviewing management arrangements in light of the relative resilience of areas of the GBR; and working with industries to promote industry-led initiatives to address climate change.

Addressing information gaps

The Great Barrier Reef Marine Park Authority (GBRMPA) has been working with scientists to assess the vulnerability of the different components of the GBR ecosystem, industries, and communities to climate change. A resultant publication identifies the key vulnerabilities for all components of the ecosystem, from plankton to corals to marine mammals, and makes management recommendations that aim to maximize the ability of the system to resist or adapt to climate changes (Johnson and Marshall, 2007). Examples of management recommendations include addressing water quality in inshore areas where primary productivity is high (*e.g.*, areas with extensive seagrass meadows or with critical plankton aggregations). Another example is conserving landward areas for migration of mangroves and wetlands as sea level rises, including possible land acquisitions and removal of barrier structures. Finally, protecting sites of specific importance from coral bleaching through shading or water mixing in summer months is an option. Reducing other impacts on critical habitats or species is also recommended (*e.g.*, improving shark fisheries management, reducing disturbance of seabird nesting sites during breeding season, reducing boat traffic and entanglement of marine mammals, protecting key turtle nesting beaches, enhancing resilience of coral reefs by improving water quality, protecting herbivores, and managing other destructive activities such as anchoring and snorkeling). These recommendations will be used to review existing management strategies and incorporate climate change considerations where needed.

Raising Awareness and Changing Behavior

The Climate Change Response Program developed a communication strategy in 2004 that aims to increase public awareness of the implications of climate change for the GBR. This strategy is being amended to include all GBRMPA activities and ensure that all groups consistently present key climate change messages. This is particularly important for groups that are addressing those factors that confer resilience to the ecosystem, such as water quality and fisheries. The key messages of the agency-wide communication strategy are that climate change is real, climate change is happening now, climate change is affecting the GBR, the GBRMPA is working to address climate change, and individuals' actions can make a difference.

The Reef Guardian program is a partnership with schools and local governments in GBR catchments. The program is voluntary and provides resources for schools and councils to incorporate sustainability initiatives into their everyday business. A sustainability and climate change syllabus has been developed for primary schools and will teach students about climate change and the implications for the GBR, as well as provide greenhouse gas emission reductions projects for the schools. The local council participants have been provided with similar information, and in order to be a recognized Reef Guardian, a council must implement a minimum number of sustainability modules. This partnership currently has 180 schools and is incrementally working toward having 20 local councils participating by 2010.

Toward Resilience-Based Management

One of the most significant strategies that coral reef managers can employ in the face of climate change is to enhance the resilience of the ecosystem (West *et al.*, 2006). Working with researchers, the Climate Change Response Program has identified resilience factors that include water quality, coral cover, community composition, larval supply, recruitment success, herbivory, disease, and effective management. These will be used to identify areas of the GBR that have high resilience to climate change and should be protected from other stresses, as well as areas that have low resilience and may require active management to enhance their resilience. Recognized research institutes have provided essential science that has formed the basis of this project and will continue collaborations between GBRMPA and researchers. Ultimately, it is hoped that this information can be used to review existing management regimes (such as planning and permit tools) to protect areas with high resilience as source sites and actively work in areas with low resilience to improve their condition.

Partnering with Stakeholders

The GBRMPA has been working with the GBR tourism industry to facilitate development of the GBR Tourism and Climate Change Action Strategy. This initiative was the result of a workshop with representative tourism operators that generated the GBR Tourism and Climate Change Action Group. This industry-led group has developed the action strategy to identify how climate change will affect the industry, how the industry can respond, and what options are available for the industry to become climate sustainable. The marine tourism industry considers reef-based activities particularly susceptible to the effects of climate change. Loss of coral from bleaching and changes to the abundance and location of fish, marine mammals, and other iconic species are likely to have the greatest impact on the industry. Increasing intensity of cyclones and storms will affect trip scheduling, industry seasonality, tourism infrastructure (particularly on islands), and future tourism industry development. Potential strategies for adapting to climate change include product diversification, new marketing initiatives, and targeting eco-accredited programs.

Managing Uncertainty

A critical component of all these strategies is the ability to manage flexibly and respond to change rapidly. This is important to enable managers to shift focus as new information becomes available or climate impact events occur. In reviewing existing management regimes, there will be a focus on ways of making management more flexible and drawing on management tools as they are needed. This type of adaptive management is essential for addressing the uncertain and shifting climate change impacts on the GBR. Given the scale of the issue and the fact that the cause and many of the solutions lie outside the jurisdiction of GBRMPA managers, effective partnerships with other levels of government and stakeholders to work cooperatively on climate change have been developed and will continue to be integral to adapting management to the climate change challenge.

8.4.3 Case Study: The Papahānaumokuākea (Northwestern Hawaiian Islands) Marine National Monument

8.4.3.1 Introduction

The Hawaiian Islands are one of the most isolated archipelagos in the world and stretch for over 2,500 km, from the island of Hawaii in the southeast to Kure Atoll (the world's highest-latitude atoll) in the northwest (Grigg, 1982; 1988; Friedlander *et al.*, 2005). Beginning at Nihoa and Mokumanamana Islands (~7 and 10 million years old, respectively) and extending to Midway and Kure Atolls (~28 million years old), the Northwestern Hawaiian Islands (NWHI) represent the older portion of the emergent archipelago (Grigg, 1988). The majority of the islets, shoals, and atolls are low-lying and remain uninhabited, although Midway, Kure, Laysan Island, and French Frigate Shoals have all been occupied for extended periods over the last century by various government agencies (Shallenberger, 2006). Because of their location in the central Pacific, the NWHI are influenced by large-wave events resulting from extratropical storms passing across the North Pacific each winter that have a profound influence on the geology and biology of the region (Grigg, 1998; Dollar and Grigg, 2004; Jokiel *et al.*, 2004; Friedlander *et al.*, 2005).

Ecosystem Structure

With coral reefs around the world in decline (Jackson *et al.*, 2001; Bellwood *et al.*, 2004; Pandolfi *et al.*, 2005), it is extremely rare to be able to examine a coral reef ecosystem that is relatively free of human influence and consisting of a wide range of healthy coral reef habitats. The remoteness and limited reef fishing and other human activities that have occurred in the NWHI have resulted in minimal anthropogenic impacts (Friedlander and DeMartini, 2002; Friedlander *et al.*, 2005). The NWHI therefore provide a unique opportunity to assess how a "natural" coral reef ecosystem functions in the absence of major localized human intervention.

One of the most striking and unique components of the NWHI ecosystem is the abundance and dominance of large apex predators such as sharks and jacks (Friedlander and DeMartini, 2002; DeMartini, Friedlander, and Holzwarth, 2005). These predators exert a strong top-down control on the ecosystem (DeMartini, Friedlander, and Holzwarth, 2005; DeMartini and Friedlander,

2006) and have been depleted in most other locations around the world (Myers and Worm, 2003; 2005). Differences in fish biomass between the main Hawaiian Islands (MHI) and NWHI represent both near-extirpation of apex predators and heavy exploitation of lower-trophic-level fishes on shallow reefs of the MHI (Friedlander and DeMartini, 2002; DeMartini and Friedlander, 2006).

The geographic isolation of the Hawaiian Islands has resulted in some of the highest endemism of any tropical marine ecosystem on earth (Jokiel, 1987; Kay and Palumbi, 1987; Randall, 1998) (Fig. 8.8). Some of these endemics are a dominant component of the community, resulting in a unique ecosystem that has extremely high conservation value (DeMartini and Friedlander, 2004; Maragos *et al.*, 2004). With species loss in the sea accelerating, the irreplaceability of these species makes Hawaii an important biodiversity hotspot (Roberts *et al.*, 2002; Allen, 2002; DeMartini and Friedlander, 2006). The coral assemblage in the NWHI contains a large number of endemics (~30%), including at least seven species of acroporid corals (Maragos *et al.*, 2004). Acroporids are the dominant reef-building corals in the Indo-Pacific, but are absent from the MHI (Grigg, 1981; Grigg, Wells, and Wallace, 1981). Kure Atoll is the world's most northern atoll and is referred to as the Darwin Point, where coral growth, subsidence, and erosion balance one another (Grigg, 1982).

Figure 8.8. Endemic species from the Hawaiian Islands. A. Masked angelfish, *Genicanthus personatus* (Photo: J. Watt), B. Rice coral, *Montipora capitata,* and finger coral, *Porites compressa* (photo: C. Hunter), C. Hawaiian hermit crab, *Calcinus laurentae* (photo: S. Godwin), D. Red alga, *Acrosymphtyon brainardii* (photo: P. Vroom).

The NWHI represent important habitat for a number of threatened and endangered species. The Hawaiian monk seal is one of the most critically endangered marine mammals in the United States (1,300 individuals) and depends almost entirely on the islands of the NWHI for breeding and the surrounding reefs for sustenance (Antonelis *et al.*, 2006). Over 90% of all sub-adult and adult Hawaiian green sea turtles found throughout Hawaii inhabit the NWHI (Balazs and Chaloupka, 2006). Additionally, seabird colonies in the NWHI constitute one of the largest and most important assemblages of seabirds in the world (Friedlander *et al.*, 2005).

In contrast to the MHI, the reefs of the NWHI are relatively free of major human influences. The few alien species known from the NWHI are restricted to the anthropogenic habitats of Midway Atoll and French Frigate Shoals (Friedlander *et al.*, 2005). Disease levels in corals in the NWHI were much lower than those reported from other locations in the Indo-Pacific (Aeby, 2006).

Existing Stressors

Although limited in scale, a number of past and present human activities have negatively affected the NWHI. Marine debris is currently one of the largest threats to the reefs of the NWHI (Boland *et al.*, 2006; Dameron *et al.*, 2007). Marine debris has caused entanglement of a number of protected species and damage to benthic habitats and is a potential vector for invasive species in the NWHI (Dameron *et al.*, 2007). An extensive debris removal effort between 1999 and 2003 has now surpassed the accumulation rate, resulting in a reduction in overall accumulation levels

(Boland *et al.*, 2006). However, much of this debris originates thousands of kilometers away in the north Pacific, making the solution to the problem both a national and international issue. Other direct human stresses such as pollution, coastal development, and ship groundings, have had negative consequences in localized areas but have been limited to a small number of locations.

The NWHI are influenced by a dynamic environment that includes large annual water temperature fluctuations, seasonally high wave energy, and strong inter-annual and inter-decadal variations in ocean productivity (Polovina *et al.*, 1994; Grigg, 1998; Polovina *et al.*, 2001; Friedlander *et al.*, 2005). As a result of these influences, natural stressors play an important role in the structure of the NWHI ecosystem. Large swell events generated every winter commonly produce waves up to 10–12 m in vertical height and between 15–20 m about once every decade (Grigg *et al.*, 2007). This limits the growth and abundance of coral communities, particularly on the north and western sides of all the islands. The best-developed reefs on all the islands exist either in the lagoons or off southwestern exposures (Grigg, 1982).

Summer sea surface temperatures (SSTs) along the island chain are generally similar, peaking at about 28ºC; however, winter SSTs are much cooler at the northern end of the chain, dipping down to 17ºC in some years (Grigg, 1982; Grigg *et al.*, 2007). This represents a 10ºC intra-annual difference at the northern end of the chain, while that at the southern end of the NWHI is only half as great: 5ºC (22–27ºC). Compared with most reef ecosystems around the globe, the annual fluctuations of SST of about 10°C at these northerly atolls is extremely high. Cooler water temperatures to the north restrict the growth and distribution of a number of coral species (Grigg, 1982). In addition, the biogeographic distribution of many fish species in the NWHI is influenced by differences in water temperatures along the archipelago (DeMartini and Friedlander, 2004; Mundy, 2005).

Climate Sensitivity

The NWHI ecosystem is sensitive to natural climate variability at a number of spatial and temporal scales. The Pacific Decadal Oscillation (PDO) results in changes in ocean productivity at large spatial and long temporal scales and has been attributed to changes in monk seal pup survival, sea bird fledging success, and spiny lobster recruitment in the NWHI (Polovina *et al.*, 1994; Polovina, Mitchem, and Evans, 1995). Inter-annual variation in the Transition Zone Chlorophyll Front is also known to affect the distribution and survival of a number of species in the NWHI (Polovina *et al.*, 1994; Polovina *et al.*, 2001).

Because of their high latitude location in the central Pacific, the NWHI were thought to be one of the last places in the world to experience coral bleaching (Hoegh-Guldberg, 1999). Hawaiian reefs were unaffected by the 1998 mass bleaching event that affected much of the Indo-Pacific region (Hoegh-Guldberg, 1999; Reaser, Pomerance, and Thomas, 2000; Jokiel and Brown, 2004). The first documented bleaching event in the MHI was reported in 1996 (Jokiel and Brown, 2004). The NWHI were affected by mass coral bleaching in 2002 and again in 2004 (Aeby *et al.*, 2003; Kenyon *et al.*, 2006). Bleaching was most acute at the three northern-most atolls (Pearl and Hermes, Midway, and Kure) and was most severe on backreef habitats (Kenyon and Brainard, 2006). Of the three coral genera that predominate at these atolls, *Montipora* and *Pocillopora* spp. were most affected by bleaching, with lesser incidences observed in *Porites*

(Kenyon and Brainard, 2006). The occurrence of two mass bleaching episodes in three years lends credence to the projection of increased frequency of bleaching with climate change.

SST data derived from both remotely sensed satellite observations (Fig. 8.9a) as well as in situ Coral Reef Early Warning System (CREWS) buoys suggest that prolonged, elevated SSTs combined with a prolonged period of anomalously light wind speed led to decreased wind and wave mixing of the upper ocean (Hoeke et al., 2006) (Fig. 8.9b). The reefs to the southeast of the archipelago show smaller positive temperature anomalies compared with the reefs towards the northwest. Research and monitoring efforts should target this pattern to better understand dispersal, bleaching, and other events that might be affected by it.

Figure 8.9. a) NOAA Pathfinder SST anomaly composite during summer 2002 period of NWHI elevated temperatures, July 28–August 29. b) NASA/JPL Quikscat winds (wind stress overlayed by wind vector arrows) composite during summer 2002 period of increasing SSTs, July 16–August 13. The Hawaii Exclusive Economic Zone (EEZ) is indicated with a heavy black line; all island shorelines in the archipelago are also plotted (adapted from Hoeke et al., 2006).

Potential Impacts of Climate Change

Climate change may increase the intensity of storm events as well as result in changes in ocean temperature, circulation patterns, and water chemistry (Cabanes, Cazenave, and Le Provost, 2001; Houghton *et al.*, 2001; Caldeira and Wickett, 2003). Warmer temperatures in Hawaii have been shown to cause bleaching mortality (Jokiel and Coles, 1990) and negatively affect fertilization and development of corals (Krupp, Hollingsworth, and Peterka, 2006). Annual spawning of some species in Hawaii occurs at temperatures near the upper limit for reproduction (Krupp, Hollingsworth, and Peterka, 2006), so increases in ocean temperature related to climate change may have a profound effect on coral populations by causing reproductive failure. The rate and scale at which bleaching has been increasing in recent decades (Glynn, 1993) points to the likelihood of future bleaching events in Hawaii (Jokiel and Coles, 1990).

Coral disease is currently low in the NWHI (Aeby, 2006), but increases in the frequency and intensity of bleaching events will stress corals and make them more susceptible to disease (Harvell *et al.*, 1999; Harvell *et al.*, 2002). Acroporid corals are prone to bleaching and disease (Willis, Page, and Dinsdale, 2004) and are restricted in range and habitat within the Hawaiian Archipelago to a few core reefs in the NWHI (Grigg, 1981; Grigg, Wells, and Wallace, 1981; Maragos *et al.*, 2004). This combination could lead to the extinction of this genus from Hawaii if mortality associated with climate change becomes severe.

Most of the emergent land in the NWHI is low-lying, highly vulnerable to inundation from storm waves, and therefore vulnerable to sea-level rise (Baker, Littnan, and Johnston, 2006). The limited amount of emergent land in the NWHI is critical habitat for the endangered Hawaiian monk seal (Antonelis *et al.*, 2006), the threatened green sea turtle (Balazs and Chaloupka, 2006), and numerous terrestrial organisms and land birds that are found nowhere else on Earth (Rauzon, 2001). The emergent land in the NWHI may shrink by as much as 65% with a 48 cm rise in sea level (Baker, Littnan, and Johnston, 2006). Efforts such as translocation or habitat alteration might be necessary if these species are to be saved from extinction.

At the northern end of the chain, lower coral diversity is linked to lower winter temperatures and lower annual solar radiation (Grigg, 1982). Increases in ocean temperature could therefore change the distribution of corals and other organisms that might currently be limited by lower temperatures. Many shallow-water fish species that are adapted to warmer water are restricted from occurring in the NWHI by winter temperatures that can be as much as 7°C cooler than the MHI (Mundy, 2005). Conversely, some shallow-water species are adapted to cooler water and can be found in deeper waters at the southern end of the archipelago. This phenomenon—known as tropical submergence—is exemplified by species such as the yellowfin soldierfish (*Myripristis chrysonemus*), the endemic Hawaiian grouper (*Epinephelus quernus*), and the masked angelfish (*Genicanthus personatus*), which are found in shallower water at Midway and/or Kure atolls, but are restricted to deeper depths in the MHI (Randall *et al.*, 1993; DeMartini and Friedlander, 2004; Mundy, 2005).

Level/Degree of Management

Administrative jurisdiction over the islands and marine waters is shared by NOAA/NMSP, U.S. Fish and Wildlife Service, and the State of Hawaii. Eight of the 10 NWHI (except Kure and Midway Atolls) have been protected by what is now the Hawaiian Islands National Wildlife Refuge (HINWR) established by President Theodore Roosevelt in 1909. The Northwestern Hawaiian Islands Coral Reef Ecosystem Reserve was created by Executive Orders 13178 and 13196 in December 2000 and amended by Executive Order 13196 in January of 2001 to include the marine waters and submerged lands extending 1,200 nautical miles long and 100 nautical miles wide from Nihoa Island to Kure Atoll.

In June 2006, nearly 140,000 square miles of the marine environment in the NWHI was designated as the Papahānaumokuākea (Northwestern Hawaiian Islands) Marine National Monument (PMNM). This action provided immediate and permanent protection for the resources of the NWHI and established a management structure that requires extensive collaboration and coordination among the three primary co-trustee agencies: the State of Hawaii, the U.S. Fish and Wildlife Service, and NOAA.

Proclamation 8031 states that the monument will:

- Preserve access for Native Hawaiian cultural activities;
- Provide for carefully regulated educational and scientific activities;
- Enhance visitation in a special area around Midway Island;
- Prohibit unauthorized access to the monument;
- Phase out commercial fishing over a five-year period; and
- Ban other types of resource extraction and dumping of waste.

Preservation areas have been established in the PMNM in sensitive areas around all the emergent reefs, islands, and atolls. In the future, all vessels issued permits to operate in the PMNM will be required to carry approved Vessel Monitoring Systems (VMS).

Program of Monitoring and Research

Long-term monitoring relevant to climate change has been conducted in the NWHI dating back to the 1970s by a variety of agencies (Griggs, 2006). Since 2000, a collaborative interagency monitoring program led by the Coral Reef Ecosystem Division (CRED) of the NOAA Pacific Islands Science Center has conducted integrated assessment and monitoring of coral reef ecosystems in the NWHI and throughout the U.S. Pacific (Wadell, 2005; Friedlander *et al.*, 2005). In conjunction with various state, federal, and academic partners, this program has integrated ecological studies with environmental data to develop a comprehensive ecosystem-based program of assessment and monitoring of U.S. Pacific coral reef ecosystems.

Ocean currents are measured and monitored in the NWHI using shipboard acoustic Doppler current profilers (ADCP), Surface Velocity Program (SVP) current drifters, and APEX profiling drifters (Friedlander *et al.*, 2005; Firing and Brainard, 2006). Spatial maps of ocean currents in the vicinity of the NWHI are also computed from satellite observations of sea surface height from the TOPEX-Poseidon and JASON altimetric satellites (Polovina, Kleiber, and Kobayashi, 1999). Moored ADCPs have been deployed by CRED at several locations to examine temporal variability of ocean currents over submerged banks and reef habitats in the NWHI.

Because of the significant influence of temperature on coral reef ecosystem health, observations of temperature in the NWHI are collected by a wide array of instruments and platforms, including satellite remote sensing (AVHRR) of SST (Smith and Reynolds, 2004), moored surface buoys and subsurface temperature recorders, closely spaced shallow water conductivity-temperature-depth profiles (CTD casts) in nearshore reef habitats, broadly spaced shipboard deep water CTD casts to depths of 500 m, and satellite-tracked SVP drifters. These data are integrated in the Coral Reef Ecosystem Integrated Observing System (CREIOS) as described below.

8.4.3.2 Managing the Papahānaumokuākea Marine National Monument

Current Approaches to Management and How Climate Change is Being Addressed

Over the past several years, the NOAA Coral Reef Conservation Program has established the Coral Reef Ecosystem Integrated Observing System (CREIOS), which is a cross-cutting collaboration between four NOAA Line Offices (NMFS, OAR, NESDIS, and NOS) focused on mapping, monitoring, and observing ecological and environmental conditions of U.S. coral reefs. At present, the ocean observing system in the NWHI consists of surface buoys measuring SST, salinity, wind, atmospheric pressure, and air temperature (enhanced systems also measure ultraviolet-B (UV-B) and photosynthetically available radiation); surface SST buoys; subsurface Ocean Data Platforms measuring ocean current profiles, wave energy and direction, temperature and salinity; subsurface current meters measuring bottom currents and temperature; and subsurface temperature recorders. Many of the surface platforms provide near real-time data telemetry to the Pacific Islands Fisheries Science Center and subsequent distribution via the World Wide Web. Time series data from subsurface instruments (without telemetry) are typically available every 12 to 24 months, after the instrument has been recovered and the dataset uploaded. Information about available datasets such as geo-location, depth, data format, and other metadata are available for both surface and subsurface instruments at the NOAA Coral Reef Information System (CoRIS) website (National Oceanic and Atmospheric Administration, 2007b).

Another component of CREIOS is Coral Reef Watch (NESDIS, Office of Research and Applications) which uses remote sensing, computational algorithms, and artificial intelligence tools in the near real-time monitoring, modeling, and reporting of physical environmental conditions that adversely influence coral reef ecosystems. Satellite remotely sensed data products include near real-time identification of bleaching "hotspots" and identification of low-wind (doldrums) areas over the world's oceans. The CRED long-term moored observing stations are part of the Coral Reef Early Warning System (CREWS) network initiated by the NOAA Coral Health and Monitoring Program, which provides access to near real-time meteorological and oceanographic data from major U.S. coral reef areas. The CREWS buoys deployed by CRED in the NWHI record and telemeter data pertaining to sea-surface temperature, salinity, wind speed and direction, air temperature, barometric pressure, UV-B, and photosynthetically available radiation (Kenyon *et al.*, 2006; NOAA National Marine Fisheries Service, 2007).

Information from CREIOS serves to alert resource managers and researchers to environmental events considered significant to the health of the surrounding coral reef ecosystem, allowing managers to implement response measures in a timely manner, and allowing researchers to increase spatial or temporal sampling resolution, if warranted. Response measures might include focused monitoring to determine the extent and duration of the event and management actions could include limiting access to these areas until recovery is observed. Information from the Coral Reef Watch Program in summer 2002 indicated conditions favorable for bleaching and resulted in assessments focused on potential bleaching areas during the subsequent research cruise.

Potential for Altering or Supplementing Current Management Practices to Enable Adaptation to Climate Change

To more fully address concerns about the ecological impacts of climate change on coral reef ecosystems and the effect of reef ecosystems on climate change, a number of agencies have proposed a collaborative effort to establish a state-of-the-art ocean observing system to monitor the key parameters of climate change impacting reef ecosystems of the Pacific and Western Atlantic/Caribbean. This proposed system includes:

- Expanding the existing array of oceanographic platforms across the remainder of the U.S. Pacific Islands
- Installing pCO_2 and UV-B sensors to examine long-term changes in carbon cycling and UV radiation
- Establish long-term records of coral reef environmental variability to examine past climate changes using paleoclimatic records of SST and other parameters from coral skeletons. This will allow us to determine if current and future SST stresses are unusual, or part of natural climatic variability.
- Develop/expand integrated *in situ* and satellite based bleaching mapping system
- Continue the development of the Coral Reef Early Warning System, which can be used to develop timely research activities to determine the extent and duration of any climate event and management actions that can potentially be implemented to mitigate these events.

In order to better understanding the impact of sea-level rise on low-lying emergent areas in the NWHI, data are needed on hydrodynamic and geological characteristics of the region. Detailed information on elevation, bathymetry, waves, wind, tide, etc. is needed to develop predictive models of shoreline change relative to climate change. One possible management measure to counter loss of habitat for monk seals and turtles in the NWHI due to sea level rise might be beach nourishment (Baker, Littnan, and Johnston, 2006). Given the small size of the islets in the NWHI, local sand resources might be sufficient to mitigate sea level rise, but a great deal of research and planning would be required given the remoteness and sensitive nature of the ecosystem (Baker, Littnan, and Johnston, 2006).

8.4.3.3 Conclusions

The nearly pristine condition of the NWHI results in one of the last large-scale, intact, predator-dominated reef ecosystems remaining in the world (Friedlander and DeMartini, 2002; Pandolfi *et al.*, 2005). Top predators can regulate the structure of the entire community and have the potential to buffer some of the ecological effects of climate change (Sala, 2006). Intact ecosystems such as the NWHI are hypothesized to be more resistant and resilient to stressors, including climate change (West and Salm, 2003). Owing to its irreplaceable assemblage of organisms, it possesses extremely high conservation value. The Papahānaumokuākea Marine National Monument is the largest marine protected area (MPA) in the world and provides a unique opportunity to examine the effects of climate change on a nearly intact large-scale marine ecosystem.

8.4.4 Case Study: the Channel Islands National Marine Sanctuary

8.4.4.1 Introduction

Ecosystem Structure

Designated in 1980, the Channel Islands National Marine Sanctuary (CINMS) consists of an area of approximately 1,243 nm^2 of coastal and ocean waters and submerged lands off the southern coast of California (Fig. 8.10). CINMS extends 6 nm offshore from the five northern Channel Islands, including San Miguel, Santa Cruz, Santa Rosa, Anacapa, and Santa Barbara islands. The primary objective of the sanctuary is to conserve, protect, and enhance the biodiversity, ecological integrity, and cultural legacy of marine resources surrounding the Channel Islands for current and future generations. State and federal agencies with overlapping jurisdiction in the CINMS, including the California Department of Fish and Game, the Channel Islands National Park, and the National Marine Fisheries Service, are working together to manage impacts of human activities on marine ecosystems.

Figure 8.10. Map of the Channel Islands National Marine Sanctuary showing the location of existing state and proposed federal marine reserves and marine conservation areas (Channel Islands National Marine Sanctuary, 2007).

The Channel Islands are distributed across a biogeographic boundary between cool temperate waters of the Californian Current and warm temperate waters of the Davidson Current (or California Countercurrent). The California Current is characterized by coastal upwelling of cool, nutrient-rich waters that contribute to high biological productivity. Intertidal communities around San Miguel, Santa Rosa, and part of Santa Cruz islands are characteristic of the cool temperate region, whereas those around Catalina, San Clemente, Anacapa, and Santa Barbara islands are associated with the warm temperate region (Murray and Littler, 1981). Fish communities around the Channel Islands also show a distinctive grouping based on association with western islands (influenced strongly by the California Current) and eastern islands (influenced by the Davidson Current). Rockfish (*Sebastes* spp.), embiotocid species, and pile perch occur more in western islands while Island kelpfish (*Alloclinus holderi*), opaleye (*Girella nigricans*), garibaldi (*Hypsypops rubicundus*), blacksmith (*Chromis punctipinnis*), and kelp bass (*Paralabrax clathratus*) occur more often in the eastern islands (Halpern and Cottenie, 2007).

From Monterey Bay to Baja California, including the Channel Islands, giant kelp (*Macrocystis pyrifera*) is the dominant habitat-forming alga. Giant kelp grows in dense stands on hard rocky substrate at depths of 2–30 m (Foster and Schiel, 1985). Kelp is among the fastest growing of all algae, adding an average of 27 cm/day (in spring) and a maximum of 61 cm/day and reaching lengths of 60 m (200 ft). Giant kelp forests support a diverse community of associated species including marine invertebrates, fishes, marine mammals and seabirds (Graham, 2004). Kelp stocks and fronds may support thousands of invertebrates including amphipods, decapods, polychaetes, and ophiuroids. Some invertebrates such as sea urchins (*Strongylocentrotus* spp.) and abalone (*Haliotis* spp.) rely on bits of drifting kelp as their primary source of food. Fish in the kelp forest community specialize in life at different depths: kelp, black and yellow, and gopher rockfish are found at the base of kelp stocks, while olive, yellowtail, and black rockfish swim in mid-water. Drifting kelp mats at the sea surface provide cover for young fishes that are vulnerable to predation. Marine mammals and seabirds are attracted to abundant fish and invertebrate populations (which serve as their primary prey) associated with kelp forests. Because of their high diversity, California kelp forests are thought to be more resistant and resilient to disturbance than kelp forests elsewhere (Steneck *et al.*, 2002).

Stressors on Marine Ecosystems in the Channel Islands

Kelp forest communities are vulnerable to an array of stressors caused by human activities and natural environmental variation. Using data gathered by the Channel Islands National Park over a period of 20 years, Halpern and Cottenie (2007) documented overall declines in abundance of giant kelp communities over time. These declines were linked with commercial and recreational fishing in the Channel Islands. Fishing reduces density and average individual size of targeted populations and, consequently, targeted species are more vulnerable to the effects of natural environmental variation. Fishing also has cascading effects through the marine food web. In areas of the Channel Islands where lobster (*Panulirus interruptus*) and other top predators were fished, purple sea urchin (*Strongylocentrotus purpuratus*) populations were more abundant, overgrazing stands of giant kelp and other algae and resulting in barren reefs devoid of kelp and its associated species (Behrens and Lafferty, 2004).

Kelp forest communities also respond to natural environmental variations, such as increased storm activity, ocean warming, and shifts in winds associated with ENSO events (Dayton *et al.*, 1992; Ladah, Zertuche-Gonzalez, and Hernandez-Carmona, 1999; Edwards, 2004). Storm activity, which is known to increase during periods of ocean warming, damages kelp stocks and rips kelp holdfasts from their rocky substrate (Dayton *et al.*, 1992; 1999). In addition to the physical damage from storms, kelp growth may be suppressed by lower levels of nutrients due to relaxation of coastal wind activity and reduction of upwelling during ENSO events. Giant kelp forests were decimated during the intense ENSO event of 1982–83 and did not recover to their previous extent for almost two decades. Several other ENSO events, in 1992–93 and 1997–98 also diminished kelp growth. The effects of these ENSO events may have been compounded by a shift (Pacific Decadal Oscillation) in 1977 to a period of slightly warmer waters in the northeastern Pacific Ocean.

Dramatic declines of giant kelp communities are likely the consequence of cumulative impacts of human activities and natural environmental variation. Giant kelp forests in one marine reserve (where fishing has been prohibited since 1978) were more resilient to ocean warming, shifts in winds, and increased storm activity associated with ENSO (Behrens and Lafferty, 2004). Giant kelp forests in the reserve persisted over a period of 20 years, including several intense ENSO events. Kelp forests at all study sites outside of the reserve were overgrazed by dense populations of sea urchins, and their growth was further inhibited by warmer water, increased storm activity, and lower levels of nutrients, leading to periodic die-backs to a barren reef state. These observations suggest that marine reserves can be used as a management tool to increase resilience of kelp forest communities.

Current Management of the Channel Islands

In 1999, the CINMS and the California Department of Fish and Game (CDFG) developed a partnership and public process (modeled after the Florida Keys National Marine Sanctuary) to consider the use of fully protected marine reserves to protect natural biological communities (Box 8.7). The cooperating agencies engaged a working group of stakeholders through the Sanctuary Advisory Council to evaluate the problem and develop potential solutions. The "Marine Reserves Working Group" developed a problem statement acknowledging that human activities and natural ecological changes contributed to the decline of marine communities in southern California. The working group determined that marine reserves should be established to protect marine habitats and species, to achieve sustainable fisheries and maintain long-term socioeconomic viability, and to protect cultural heritage. The stakeholders, working with marine scientists and economists, created a range of options for marine reserves to meet these goals. Subsequently, the CINMS and CDFG used the two most widely supported options to craft compromise solution that addressed the interests of a broad array of stakeholders.

In 2003, the CDFG established a network of 10 fully protected marine reserves and two conservation areas that allow limited commercial and recreational fishing (Fig. 8.10). The total area protected was 102 nm^2, approximately 10% of sanctuary waters. The marine reserves and conservation areas included a variety of representative marine habitats characteristic of the region, such as rocky intertidal habitats, sandy beaches, kelp forests, seagrass beds, soft bottom habitats, submerged rocky substrate, and submarine canyons. In 2006, the Pacific Fisheries Management Council designated Essential Fish Habitat to protect benthic communities from bottom contact fishing gear within and adjacent to the state marine protected areas, up to 6 nm offshore. In the same year, the CINMS released a Draft Environmental Impact Statement proposing complementary marine reserves and a marine conservation area extending into federal waters (Fig. 8.10). The Essential Fish Habitat designated by the Council and the marine protected areas proposed by the sanctuary increase the total area of protected marine zones to 19% of the CINMS.

In 2008, data from relevant monitoring programs will be prepared for a review by the California Fish and Game Commission of the first five years of monitoring the Channel Islands state marine reserves. Expectations are that species that were targeted by commercial or recreational fisheries will increase in density and size within marine reserves (Halpern, 2003). Some species are expected to decline if their predators or competitors increase in abundance.

Potential Effects of Climate Change on Ecosystems in the Channel Islands region

Coastal SST has increased steadily (by approximately 2^{o}C) since 1950 and is expected to increase further in the coming centuries (IPCC, 2007b). Water temperature affects metabolism and growth (Bayne, Thompson, and Widdows, 1973; Phillips, 2005), feeding behavior (Petraitis, 1992; Sanford, 1999; 2002), reproduction (Hutchins, 1947; Philippart *et al.*, 2003), and rates of larval development (Hoegh-Guldberg and Pearse, 1995; Anil, Desai, and Khandeparker, 2001; Luppi, Spivak, and Bas, 2003; O'Connor *et al.*, 2007) of intertidal and subtidal animals. Shifts in species ranges already have occurred in California with the steady increase of coastal sea surface temperature. The range boundary of *Kelletia kelletii* has shifted north from the late 1970s to the 2000s (Herrlinger, 1981; Zacherl, Gaines, and Lonhart, 2003). Southern species of anthozoans, barnacles, and gastropods increased in Monterey Bay, while northern species of anthozoans and limpets decreased between the 1930s (Hewatt, 1937) and the 1990s (Barry *et al.*, 1995; Sagarin *et al.*, 1999). Holbrook, Schmitt, and Stephens, Jr. (1997) documented an increase of 150% in southern species of kelp forest fish in southern California, and a decrease of 50% in northern species since the 1970s.

Increased ocean temperatures have been linked with outbreaks of marine disease (Hofmann *et al.*, 1999). Populations of black abalone (*Haliotis cracherodii*) in the Channel Islands and north along the California coast to Cambria suffered mass mortalities from “withering syndrome” caused by the intracellular prokaryote *Xenohaliotis californiensis*, between 1986 and 2001. Healthy populations of black abalone persist north of Cambria, where cool waters suppress the disease. Samples of red abalone (*Haliotis rufescens*) from populations around San Miguel Island in 2006 indicated that approximately 58% of the population carries *X. californiensis*, but the red abalone population persists in a thermal refuge within which temperatures are low enough to suppress the expression of the disease. The disease may be expressed during prolonged periods of warming (*e.g.*, over 18^{o}C for several days) associated with ENSO or other warm-water events. In 1992, an ENSO year, an urchin-specific bacterial disease entered the Channel Islands region and spread through dense populations of purple sea urchin (*Strongylocentrotus pupuratus*). Sites located in a marine reserve where fishing was prohibited had more lobster (which prey on urchins), smaller populations of urchins, persistent forests of giant kelp, and a near absence of the disease (Lafferty and Kushner, 2000). During several warm-water events, including the ENSO of 1997–98, scientists observed and documented declines of sea star populations at the Channel Islands due to epidemics of “wasting disease,” which disintegrates the animals.

Increased temperature is expected to lead to numerous changes in currents and upwelling activity. As the sea surface warms, thermal stratification will intensify and become more stable, leading to reduced upwelling of cool, nutrient-rich water (Soto, 2001; Field *et al.*, 2001). Reduced upwelling will lead to a decline in primary productivity (McGowan *et al.*, 1998), suppression of kelp growth, and cascading effects through the marine food web.

Introductions of non-native species (such as the European green crab *Carcinus maenas* on the U.S. West Coast) are associated with rising temperatures and altered currents associated with ENSO events (Yamada *et al.*, 2005). The Sanctuary Advisory Council identified non-indigenous species as an emerging issue in the revised Sanctuary Management Plan (U.S. Department of Commerce, 2006). The sanctuary participated in the removal of a non-indigenous alga (*Undaria pinnatifida*) from the Santa Barbara Harbor, but the sanctuary does not support systematic monitoring or removal of non-indigenous species. Introduction of non-indigenous species can disrupt native communities, potentially leading to shifts in community structure.

Sea level may rise up to three feet in the next 100 years, depending on the concentrations of greenhouse gases during this period (Cayan *et al.*, 2006; IPCC, 2007b). Projections of sea level rise around the Channel Islands indicate little encroachment of seawater onto land due to steep rocky cliffs that form the margins of the islands; however, projections of sea level rise indicate potential saltwater intrusion into low-lying coastal areas such as the Santa Barbara Harbor (where the CINMS Headquarters is located) and the Channel Islands Harbor (where the sanctuary's southern office is located). Changes in sea level may affect the type of coastal ecosystem (Hoffman, 2003). Graham, Dayton, and Erlandson (2003) suggested that sea level rise transformed the Southern California Bight from a productive rocky coast to a less productive sandy coast more than 18,000 years ago.

The severity of storm events is likely to increase with climate change (Houghton *et al.*, 2001). As described above, storm activity damages kelp stocks and pulls kelp holdfasts from the substrate (Dayton *et al.*, 1992; 1999). Frequent and intense storm activity during the 1982–83 ENSO event decimated populations of giant kelp that once formed extensive beds attached to massive old kelp holdfasts in sandy areas along the mainland coast. Since the old kelp holdfasts were displaced from the mainland coast, young kelp plants have been unable to attach to the sandy substrate and the coastal kelp forests have not returned. At the Channel Islands, kelp forests that were destroyed during the same ENSO event have slowly returned to the rocky reefs around the Channel Islands, particularly following a Pacific Decadal Oscillation to cooler waters in 1998.

A Shared Vision for the Channel Islands

The CINMS manager and staff work closely with the Sanctuary Advisory Council to identify and resolve resource management issues. As noted above, the Sanctuary Advisory Council consists of representatives from local, state, and federal agencies, which share jurisdiction of resources within the Channel Islands region, and stakeholders with interests in those resources. The Sanctuary Advisory Council offers a unique opportunity to focus attention of regional agencies and stakeholders on the potential threats associated with climate change and to develop a shared vision for how to respond.

The Sanctuary Management Plan (U.S. Department of Commerce, 2006) describes a strategy to work in a coordinated, complementary, and comprehensive manner with other authorities that share similar or overlapping mandates, jurisdiction, objectives, and/or interests. The sanctuary is poised to take a leading role to bring together the relevant agencies and stakeholders to discuss the issue of climate change. The sanctuary can initiate an effort to develop regional plans to adapt to a modified landscape and seascape predicted from climate change models, and mitigate the negative impacts of climate change.

8.4.4.2 Management of the Channel Islands National Marine Sanctuary

The Sanctuary Management Plan (U.S. Department of Commerce, 2006) for the CINMS mentions but does not fully address the issue of climate change, with one exception in the strategy for offshore water quality monitoring. The strategy is to better evaluate and understand impacts on water quality from oceanographic and climatic changes and human activities. The proposed actions include continued vessel and staff support for monitoring projects related to water quality. To evaluate the potential impacts of climate change, the sanctuary staff could expand monitoring of—or collaborate with researchers who are monitoring—ocean water temperature, currents, dissolved oxygen, and pH at different depths.

The Sanctuary Management Plan (U.S. Department of Commerce, 2006) describes a strategy to identify, assess, and respond to emerging issues. The plan explicitly identifies noise pollution, non-indigenous species, and marine mammal strikes as emerging issues. Other emerging issues that are not addressed by the management plan, but should be, include ocean warming, sea level rise, shifts in ocean circulation, ocean acidification, spread of disease, and shifts in species ranges.

The Sanctuary Management Plan (U.S. Department of Commerce, 2006) outlined a potential response to emerging issues through consultation with the Sanctuary Advisory Council and local, state, or federal agencies with a leading or shared authority for addressing the issue. With the elevated level of certainty associated with climate change projections (IPCC, 2007b), it is appropriate to bring the topic of climate change to the Sanctuary Advisory Council and begin working with local, state, and federal agencies that share authority in the region to plan for potential impacts of climate change. Regional agency managers may consider and develop strategies to respond to the potential impacts of:

- Ocean warming (contributing to potential shifts in species ranges, changes in metabolic and physiological processes, and accelerated spread of disease);
- Ocean acidification (leading to breakdown of calcareous accretions in corals and shells);
- Shifts in ocean circulation (leading to changes in upwelling activity and possible formation of low oxygen zones); and
- Sea level rise (shifting jurisdictional boundaries, displacing terrestrial and intertidal organisms, leading to salt-water inundation of coastal marshes, lagoons and estuaries, and increasing coastal flood events).

Monitoring and Research in the Channel Islands Region

Monitoring and research are critical for detecting and understanding the effects of climate and ocean change. The Sanctuary Management Plan (U.S. Department of Commerce, 2006) outlines strategies for monitoring and research in the coming years, but the plan does not address climate and ocean change specifically. The current strategies for monitoring and research can be refocused slightly to capture important information about climate and ocean change.

Monitoring of algae, invertebrates, and fishes is needed within and around marine reserves to detect differences between protected and targeted populations in their responses to climate change. One hypothesis is that populations within marine reserves will be more resilient to the effects of climate change than those that are altered by fishing and other extractive uses. In addition, scientists have determined that local environmental variation causes different populations to respond in different ways to ocean warming (e.g., Helmuth *et al.*, 2006). For example, a population of red abalone at San Miguel Island lives in a “thermal refuge” where waters are cooled by upwelling, preventing spread of disease that is carried in the population. Sustained ocean warming is likely to increase thermal stress of individuals in this population and accelerate the spread of disease through affected populations. Monitoring can be used to detect such changes at individual, population, and regional levels. The CINMS has the capacity to support subtidal monitoring activities from the *RV Shearwater*, aerial surveys of kelp canopy from the sanctuary aircraft, and collaborative research projects with scientists and fishermen.

In addition to the ecological monitoring in marine reserves, it will be critical to monitor environmental variables, including ocean water temperature, sea level, currents, dissolved oxygen, and pH at different depths. Any change in these variables should trigger more intensive monitoring to evaluate the ecological impacts of ocean warming, sea level rise, shifts in current patterns, low oxygen, and increased acidification. The sanctuary could benefit from partnerships with scientists who are monitoring ocean changes and who have the capability of ramping up research activities in response to observed changes. For example, before 2002, scientists at Oregon State University, Corvallis, routinely monitored temperature and salinity at stationary moorings off the coast of Oregon. When they detected low oxygen during routine monitoring in 2002, the scientists intensified their monitoring efforts by increasing the number of temperature and salinity sensors and adding oxygen sensors (which transmit data on a daily basis) near the seafloor at a number of locations along the coast. In this way, the scientists can quantify the scope and duration of hypoxic events, which have recurred off the coast of Oregon during the past five years (Barth *et al.*, 2007).

The Sanctuary Management Plan (U.S. Department of Commerce, 2006) describes the need for analysis and evaluation of information from sanctuary monitoring and research. Working with local educational institutions and the National Center for Ecological Analysis and Synthesis, the sanctuary could develop the capacity to catalog and analyze spatial data (maps) that characterize the coastline of the sanctuary and the extent of kelp canopy within the sanctuary, among other types of information. To detect the ecological impacts of climate change, the information from sanctuary monitoring and research should be reviewed at regular intervals (at least annually) by collaborating scientists (such as the Sanctuary Advisory Council’s Research Activities Panel), sanctuary staff, and the sanctuary manager. The annual review should compare data from the current year with previous years, from areas inside marine reserves and in surrounding, fished areas. Ecological changes should be placed within the context of El Niño-Southern Oscillation and La Niña cycles and shifts associated with the Pacific Decadal Oscillation. Changes in fisheries or other management regulations also should be considered as part of the evaluation. Any significant shifts away from predictable trends should trigger further evaluation of the data in an effort to understand local and regional ecosystem dynamics and any possible links to climate change.

Communication in the Channel Islands Region

Public awareness and understanding are paramount in the discussion about how to adapt to climate change. The education and outreach strategies described in the Sanctuary Management

Plan (U.S. Department of Commerce, 2006) do not focus on the issue of climate change but, with a slight shift in focus, the existing strategies can be used to increase public awareness and understanding of the causes and impacts of climate change on ocean ecosystems. Key strategies are to educate teachers, students, volunteers, and the public using an array of tools, including workshops, public lectures, the sanctuary website and weather kiosks, and a sanctuary publication and brochure, among others. Opportunities to focus the sanctuary education program's activities and products on the issue of climate change include the following:

- Integrate information about climate change into volunteer Sanctuary Naturalist Corps and adult education programs;
- Update the sanctuary website and weather kiosks with information about causes and impacts of climate change;
- Produce a special issue of the sanctuary publication, *Alolkoy*, about the current scientific understanding of climate change and potential impacts on sanctuary resources;
- Develop a brochure about climate change to help members of the community identify opportunities to reduce their contributions to greenhouse gases and other stressors that exacerbate the problem of climate change;
- Expand the sanctuary's Ocean Etiquette program (National Marine Sanctuary Program, 2007d) to include consideration and mitigation of individual activities that contribute to climate change;
- Host a teacher workshop on the subject of climate change;
- Prepare web-based curriculum with classroom exercises and opportunities for experiential learning about climate change; and
- Partner with local scientists who study climate change to give public lectures and engage students in monitoring climate change.

8.4.5 Conclusions About Case Studies

The Great Barrier Reef Marine Park has been examined along with the National Marine Sanctuary case studies because it is an example of an MPA that has a relatively highly developed climate change program in place. A Coral Bleaching Response Plan is part of its Climate Change Response Program, which is linked to a Representative Areas Program and a Water Quality Protection Plan in a comprehensive approach to support the resilience of the coral reef ecosystem. In contrast, the Florida Keys National Marine Sanctuary is only now developing a bleaching response plan. The Florida Reef Resilience Program, under the leadership of The Nature Conservancy, is implementing a quantitative assessment of coral reefs before and after bleaching events. The recently established Papahānaumokuākea (Northwestern Hawaiian Islands) Marine National Monument is the largest MPA in the world and provides a unique opportunity to examine the effects of climate change on a nearly intact large-scale marine ecosystem. These three MPAs consist of coral reef ecosystems, which have experienced coral bleaching events over the past two decades.

The Sanctuary Management Plan for the Channel Islands National Marine Sanctuary mentions, but does not fully address, the issue of climate change. The Plan describes a strategy to identify, assess, and respond to emerging issues through consultation with the Sanctuary Advisory Council and local, state, or federal agencies. Emerging issues that are not yet addressed by the management plan include ocean warming, sea level rise, shifts in ocean circulation, ocean acidification, spread of disease, and shifts in species ranges.

Barriers to implementation of adaptation options in MPAs include lack of resources, varying degrees of interest in and concern about climate change impacts, and a need for basic research on marine ecosystems and climate change impacts. National Marine Sanctuary Program staff are hard-pressed to maintain existing management programs, which do not yet include explicit focus on effects of climate change. While the Program's strategic plan does not address climate change, the Program has recently formed a Climate Change Working Group that will be developing recommendations. Although there is considerable research on physical impacts of climate change in marine systems, research on biological effects and ecological consequences is not as well developed.

Opportunities with regard to implementation of adaptation options in MPAs include a growing public concern about the marine environment, recommendations of two ocean commissions, and an increasing dedication of marine scientists to conduct research that is relevant to MPA management. References to climate change as well as MPAs permeate both the Pew Oceans Commission and U.S. Commission on Ocean Policy reports on the state of the oceans. Both commissions held extensive public meetings, and their findings reflect changing public perceptions and attitudes about protecting marine resources from threats of climate change. The interests of the marine science community have also evolved, with a shift from "basic" to "applied" research over recent decades. Attitudes of MPA managers have changed as well, with a growing recognition of the need to better understand ecological processes in order to implement science-based adaptive management.

8.5 Conclusions

Adaptive management of MPAs in the context of climate change includes the concept that intact marine ecosystems are more resistant and resilient to change than are degraded systems (Harley *et al.*, 2006). Marine reserves develop fully functional communities when populations of heavily fished species recover and less-altered abundance patterns and size structures accrue. Implementing networks of MPAs, including large areas of the ocean, will help "spread the risk" posed by climate change by protecting multiple replicates of the full range of habitats and communities within ecosystems (Soto, 2001; Palumbi, 2003; Halpern, 2003; Halpern and Warner, 2003; Roberts *et al.*, 2003b; Palumbi, 2004; Kaufman *et al.*, 2004; Salm, Done, and McLeod, 2006).

The most effective configuration of MPAs would be a network of highly protected areas nested within a broader management framework (Botsford, 2005; Hilborn, Micheli, and De Leo, 2006; Almany *et al.*, 2007). As part of this configuration, areas that are ecologically and physically significant and connected by currents should be identified and included as a way of enhancing resilience in the context of climate change. Critical areas to consider include nursery grounds, spawning grounds, areas of high species diversity, areas that contain a variety of habitat types in close proximity, and potential climate refugia. At the site level, managers can build resilience to climate change by protecting marine habitats from direct anthropogenic threats such as pollution, sedimentation, destructive fishing and overfishing. The healthier the marine habitat, the greater the potential will be for resistance to—and recovery from—climate-related disturbances.

In designing networks, managers should consider information on areas that may represent potential refugia from climate change impacts as well as information on connectivity (current patterns that support larval replenishment and recovery) among sites that vary in their sensitivities to climate change. Protection of seascapes creates areas sufficiently large to resist basic changes to the entire ecosystem (Kaufman *et al.*, 2004). Large reserves may benefit individual species by enabling them to spend entire adult phases of their life cycle without being captured and killed, with concomitant increases in reproductive output (Sobel and Dahlgren, 2004) and quality (Berkeley, Chapman, and Sogard, 2004).

A key issue for MPA managers concerns achieving the goals and objectives of a local-scale management plan in the context of larger-scale stressors from atmospheric, terrestrial, and marine sources (Jameson, Tupper, and Ridley, 2002). Another issue concerns maintaining a focus on immediate, devastating effects of overexploitation, coastal pollution, and nonindigenous species as climate change impacts increase in magnitude or frequency over time (Paine, 1993). Within sites, managers can increase resilience to climate change by managing other anthropogenic stressors that also degrade ecosystems, such as fishing and overexploitation; inputs of nutrients, sediments, and pollutants; and habitat damage and destruction. Efforts by MPA managers to enhance resilience and resistance of marine communities may at least "buy some time" against threats of climate change by slowing the rate of decline caused by other, more manageable stressors (Hansen, Biringer, and Hoffman, 2003; Hoffman, 2003; Marshall and Schuttenberg, 2006b).

Resilience is also affected by trophic linkages, which are key characteristics maintaining ecosystem integrity. An approach that has been identified to maintain resilience is the management of functional groups, specifically herbivores. In one instance on the Great Barrier Reef, recovery from an algae-dominated to a coral-dominated state was driven by a single batfish species rather than grazing by dominant parrotfishes or surgeonfishes that normally keep algae in check on reefs (Bellwood, Hughes, and Hoey, 2006). This finding highlights the need to protect the full range of species to maintain resilience and the need for further research on key species and ecological processes.

The challenges of climate change require creative solutions and collaboration among a variety of stakeholders to generate the necessary finances and support to respond to climate change stress. Global, regional, and local partnerships across a range of sectors such as agriculture, tourism, water resource management, conservation, and infrastructure development can help alleviate the financial burdens of responding to climate change in MPAs. Finally, effective implementation of the above strategies in support of ecological resilience will only be possible in the presence of human social resilience.

8.6 References

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8.7 Acknowledgements

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8.8 Boxes

Box 8.1. Draft Goals of the National Marine Sanctuary Program, 2005-2015

Goal 1. Identify, designate, and manage sanctuaries to maintain the natural biological communities in sanctuaries and to protect and, where appropriate, restore and enhance natural habitats, populations, and ecological processes, through innovative, coordinated and community-based measures and techniques.

Goal 2. Build and strengthen the nation-wide system of marine sanctuaries, maintain and enhance the role of the NMSP's system in larger MPA networks and help provide both national and international leadership for MPA management and marine resource stewardship.

Goal 3. Enhance nation-wide public awareness, understanding, and appreciation of marine and Great Lakes ecosystems and maritime heritage resources through outreach, education, and interpretation efforts.

Goal 4. Investigate and enhance the understanding of ecosystem processes through continued scientific research, monitoring, and characterization to support ecosystem-based management in sanctuaries and throughout U.S. waters.

Goal 5. Facilitate human use in sanctuaries to the extent such uses are compatible with the primary mandate of resource protection, through innovative public participation and interagency cooperative arrangements.

Goal 6. Work with the international community to strengthen global protection of marine resources, investigate and employ appropriate new management approaches, and disseminate NMSP experience and techniques.

Goal 7. Build, maintain, and enhance an operational capability and infrastructure that efficiently and effectively support the attainment of the NMSP's mission and goals.

Box 8.2 The Western North Atlantic Food Web

Marine carnivores of the western North Atlantic were both more abundant and larger in the past. In Maine, archaeological evidence indicates that coastal people subsisted on Atlantic cod for at least 4,000 years (Steneck, 1997; Jackson *et al.*, 2001). Prey species such as lobsters and crabs were absent from excavated middens in the region, perhaps because large predators had eaten them (Steneck, Vavrinec, and Leland, 2004; Lotze *et al.*, 2006).

Today cod are ecologically extinct from western North Atlantic coastal zones due to overfishing. The abundant lobsters and sea urchins that had formerly been the prey of apex predators became the primary target of local fisheries. By 1993, the value of sea urchins harvested in Maine for their roe was second only to lobsters. As sea urchin populations declined, so too did communitywide rates of herbivory (Steneck, 1997). In less than a decade, sea urchins became so rare that they could no longer be found over large areas of the coast (Andrew *et al.*, 2002; Steneck, Vavrinec, and Leland, 2004).

These and other instances of "fishing down food webs" in the Gulf of Maine have resulted in hundreds of kilometers of coast now having dangerously low biological and economic diversity. Today bloodworms used for bait are worth more to Maine's economy than cod (see Figure below). The trophic level dysfunction (sensu Steneck, Vavrinec, and Leland, 2004) of both apex predators and herbivores leave a coastal zone suited for crabs and especially lobsters -- the latter attaining staggering population densities exceeding one per square meter along much of the coast of Maine (Steneck and Wilson, 2001). The economic value of lobsters is high, accounting for nearly 80% of the total value of Maine's fisheries as of 2004 (see Figure below). The remaining 42 harvested species account for the remaining 20%. If a disease such as the one that recently decimated Rhode Island's lobster stocks (Glenn and Pugh, 2006) infects lobsters in the Gulf of Maine, there will be serious socio-economic implications for the fishing industry. Prospects for such a disease outbreak may increase because of climate-induced changes in the environment such as temperature increases that favor pathogen growth (Harvell *et al.*, 1999; 2002).

* Note: This figure is provisional, based on securing permission to reprint.

Box 8.3. Draft Objectives of the Goals of the National Marine Sanctuary Program That Are Relevant to Resource Protection and Climate Change (Goals 1, 4, 5, and 6 from Box 8.1)[1]

Goal 1: Protect Resources.

Objective 1. Prepare sanctuary-specific management plans and regional and national programs and policies that utilize all program capacities to protect and manage resources.

Objective 2. Conduct and maintain routine contingency planning, emergency response, damage assessment, and restoration activities to preserve and restore the integrity of sanctuary ecosystems.

Objective 3. Develop and maintain enforcement programs and partnerships to maximize protection of sanctuary resources.

Objective 4. Review and evaluate the NMSP's effectiveness at site, regional, and national levels, through both internal and external mechanisms.

Objective 5. Anticipate, characterize, and mitigate threats to resources.

Objective 6. Assess and predict changes in the NMSP's operating, natural, and social environments and evolve sanctuary management strategies to address them, through management plan reviews, reauthorizations, and program regulatory review.

Objective 7. Designate new sanctuaries, as appropriate, to ensure the nation's marine ecosystems and networks achieve national expectations for sustainability.

Goal 4: Sanctuary Science.

Objective 1. Expand observing systems and monitoring efforts within and near national marine sanctuaries to fill important gaps in the knowledge and understanding of the ocean and Great Lakes ecosystems.

Objective 2. Support directed research activities that support management decision making on challenges and opportunities facing sanctuary ecosystems, processes, and resources.

Objective 3. Develop comprehensive characterization products of ocean and Great Lakes ecosystems, processes, and resources.

Goal 5: Facilitate Compatible Use.

Objective 1. Work closely with partners, interested parties, community members, stakeholders, and government agencies to assess and manage human use of sanctuary resources.

Objective 2. Create, operate, and support community-based sanctuary advisory councils to assist and advise sites and the overall program in the management of their resources, and to serve as liaisons to the community.

Objective 3. Consult and coordinate with federal agencies and other partners conducting activities in or near sanctuaries.

Objective 4. Use other tools such as policy development, permitting, and regulatory review and improvement to help guide human use of sanctuary resources.

Goal 6: Improve International Work.

Objective 1. Develop multilateral program relationships to interact with, share knowledge and experience with, and learn from international partners to improve the NMSP's management capacity, and bring new experiences to MPA management in the U.S.

Objective 2. Investigate the use of international legal conventions and other instruments to help protect sanctuary resources, including those that are transboundary or shared.

Objective 3. Cooperate to the extent possible with global research initiatives in order to improve the overall understanding of the ocean.

Objective 4. Make NMSP education and awareness programs accessible through international efforts to increase the global population's awareness of ocean issues.

[1]Additional goals of the NMSP are in Box 8.1.

Box 8.4. Draft Natural Resource Performance Measures of the National Marine Sanctuary Program

2015: 12 sites with water quality being maintained or improved.

2015: 12 sites with habitat being maintained or improved.

2015: 12 sites with living marine resources being maintained or improved.

2010: 100% of the System is adequately characterized.

2010: six sites are achieving or maintaining an optimal management rating on the NMSP Report Card.

2007: 100% of NMSP permits are handled in a timely fashion and correctly.

2010: 100% of sites with zones in place are assessing them for effectiveness.

Box 8.5. Adaptation Options for Resource Managers

Marine Protected Areas:
Adaptation Options for Resource Managers

- ✓ Identify ecological connections between and among marine ecosystems (*e.g.*, mangroves, coral reefs, and seagrass beds) and use them to inform management decisions (*e.g.*, preserve areas resistant to bleaching upcurrent from other areas that succumb to bleaching).
- ✓ Manage functional species groups necessary to maintaining the health of reefs and other ecosystems.
- ✓ Protect areas observed to be resistant to climate change effects to ensure a secure source of recruitment to support recovery in damaged areas.
- ✓ Design MPAs to include dynamic boundaries to protect predictable breeding and foraging habits and extensive buffers to protect migratory and pelagic species.
- ✓ Create buffer zones to accommodate ecosystem shifts in response to sea level rise and temperature change.
- ✓ Monitor ecosystems and have rapid-response strategies prepared to deal with disturbances.
- ✓ Manage human stressors such as fishing and inputs of nutrients, sediments, and pollutants within MPAs.
- ✓ Create buffer zones between intensive human activity and fully-protected marine reserves.
- ✓ Identify, protect, and restore areas observed to be resistant to climate change effects or to recover quickly from climate-induced disturbances.
- ✓ Replicate habitat types in multiple areas to spread risks associated with climate change.
- ✓ Maximize habitat heterogeneity and consider protecting larger areas to preserve biodiversity, biological connections among habitats, and ecological functions.
- ✓ Include entire ecological units (*e.g.*, coral reefs with their associated mangroves and seagrasses) in MPA design to maintain ecosystem function and resilience.
- ✓ Ensure that the full breadth of habitat types is protected (*e.g.*, fringing reef, fore reef, back reef, patch reef).

Box 8.6. Goal and Objectives of the Florida Keys National Marine Sanctuary (U.S. Department of Commerce, 1996)

Goal:
To preserve and protect the physical and biological components of the South Florida estuarine and marine ecosystem to ensure its viability for the use and enjoyment of present and future generations.

Objectives Required by the FKNMS Act:

- **Objective 1.** Facilitate all public and private uses of the Sanctuary consistent with the primary objective of resource protection.
- **Objective 2.** Consider temporal and geographic zoning to ensure protection of Sanctuary resources.
- **Objective 3.** Incorporate regulations necessary to enforce the Water Quality Protection Program.
- **Objective 4.** Identify needs for research and establish a long-term ecological monitoring program.
- **Objective 5.** Identify alternative sources of funding needed to fully implement the management plan's provisions and supplement appropriations authorized under the FKNMS and National Marine Sanctuaries Acts.
- **Objective 6.** Ensure coordination and cooperation between Sanctuary managers and other federal, state, and local authorities with jurisdiction within or adjacent to the Sanctuary.
- **Objective 7.** Promote education among users of the Sanctuary about coral reef conservation and navigational safety.
- **Objective 8.** Incorporate the existing Looe Key and Key Largo National Marine Sanctuaries into the Florida Keys National Marine Sanctuary.

Objectives Developed by the FKNMS Sanctuary Advisory Council:

- **Objective 1.** Encourage all agencies and institutions to adopt an ecosystem and cooperative approach to accomplish the following objectives, including the provision of mechanisms to address impacts affecting Sanctuary resources, but originating outside the boundaries of the Sanctuary.
- **Objective 2.** Provide a management system that is in harmony with an environment whose long-term ecological, economic, and sociological principles are understood, and which will allow appropriate sustainable uses**.**
- **Objective 3.** Manage the Florida Keys National Marine Sanctuary for the natural diversity of healthy species, populations, and communities.
- **Objective 4.** Reach every single user of and visitor to the FKNMS with information appropriate to his or her activities.
- **Objective 5.** Recognize the importance of cultural and historical resources, and managing these resources for reasonable, appropriate use and enjoyment.

Box 8.7. Timeline for Establishment of Marine Reserves in the Channel Islands National Marine Sanctuary (CINMS)

- 1998: Sportfishing group initiates discussions about marine reserves in the Channel Islands National Marine Sanctuary
- 1999: California Department of Fish and Game and NOAA develop partnership and initiate community-based Marine Reserves Working Group process
- 2001: Working Group recommendations delivered to California Department of Fish and Game and NOAA
- 2003: California Fish and Game Commission established 10 state marine reserves and 2 state marine conservation areas established in state waters of the CINMS
- 2006: Pacific Fisheries Management Council designated Essential Fish Habitat and Habitat of Areas of Particular Concern in adjacent federal waters of the CINMS prohibiting bottom fishing
- 2006: Sanctuary released Draft Environmental Impact Statement to propose marine reserves in federal waters of the CINMS.
- 2007: Pending - NOAA will release Final Environmental Impact Statement and final rule to complete the marine reserves in federal waters
- 2007: Pending - California Fish and Game Commission will take regulatory action to close gaps between state and federal marine protected areas

8.9 Tables

Table 8.1. Types of federal marine protected and marine managed areas, administration, and legislative mandates. MPAs are primarily intended to protect or conserve marine life and habitat, and are a subset of marine managed areas (MMAs), which protect, conserve, or otherwise manage a variety of resources and uses including living marine resources, cultural and historical resources, and recreational opportunities (California Department of Fish and Game, 2007b).

Type of MPA/MMA	Number of Sites	Administration	Mandate
National Marine Sanctuary	13	NOAA/National Marine Sanctuary Program	National Marine Sanctuaries Act
Fishery Management Areas	216	NOAA/National Marine Fisheries Service	Magnuson-Stevens Act, Endangered Species Act, Marine Mammal Protection Act
National Estuarine Research Reserve[1]	27	NOAA/Office of Ocean and Coastal Resource Management	Coastal Zone Management Act
National Park	42	National Park Service	NPS Organic Act
National Monument[2]	3	National Park Service[2]	NPS Organic Act[2]
National Wildlife Refuge	109	U.S. Fish and Wildlife Service	National Wildlife Refuge System Administration Act

[1]The National Estuarine Research Reserve System is a state partnership program.
[2]The Papahānaumokuākea Marine National Monument is included here. It is co-managed by NOAA/National Marine Sanctuary Program and National Marine Fisheries Service, the U.S. Fish and Wildlife Service, and the State of Hawaii and was established by Presidential Proclamation 8031.

Table 8.2. Type, number, area, and no-take area of federal marine managed areas (MMAs) and areas of Exclusive Economic Zones (EEZs) by region in U.S. waters (National Oceanic and Atmospheric Administration, 2006a).

Federal Marine Managed Areas (MMAs) in U.S. Waters (0-200 nm)

Region	Type of MMA	Number	Total Area (km²)**	Total Area No Take (km²)	% Area No Take	Area of EEZ in Region (km²)
New England						197,227
	NP	0	0	0	0%	
	NWR	1	30	0	0%	
	NMS	1	2,190	0	0%	
	FMA	30	212,930	0	0%	
	NERR*	1	27	0	0%	
Mid Atlantic						218,151
	NP	3	36,472	0	0%	
	NWR	22	15	0	0%	
	NMS	0	0	0	0%	
	FMA	9	686,379	0	0%	
	NERR*	5	460	0	0%	
South Atlantic						525,627
	NP	8	1,421	119	8%	
	NWR	19	3,705	564	15%	
	NMS	3	9,853	591	6%	
	FMA	11	974,243	349	<0.1 %	
	NERR*	5	928	0	0%	
Caribbean						212,371
	NP	2	27	1	2%	
	NWR	0	0	0	0%	
	NM***	2	128	76	59%	
	NMS	0	0	0	0%	
	FMA	6	168	55	33%	
	NERR*	1	7	0	0%	
Gulf of Mexico						695,381
	NP	4	4,612	0	0%	
	NWR	24	2,375	2	<0.1%	
	NMS	1	146	0	0%	
	FMA	7	368,446	0	0%	
	NERR*	5	2,195	0	0%	
West Coast						823,866
	NP	6	595	0	0%	
	NWR	15	226	16	7%	
	NMS	5	30,519	257	1%	
	FMA	56	386,869	0	0%	
	NERR*	5	57	0	0%	
Alaska						3,710,774
	NP	3	29,795	0	0%	
	NWR	3	212,620	0	0%	
	NMS	0	0	0	0%	
	FMA	17	1,326,177	0	0%	
	NERR*	1	931	0	0%	
Pacific Islands						3,869,806
	NP	4	21	< 1	<1%	
	NWR	10	281	158	56%	
	NM***	1	352,754	352,754	100%	
	NMS	3	3,556	1	<1%	
	FMA	6	1,467,614	0	0%	
	NERR*	0	0	0	0%	
National Total						10,413,230
	NP	**42**	**72,943**	**120**	**0.16%**	
	NWR	**109**	**219,252**	**740**	**0.34%**	
	NM	**3**	**352,882**	**352,882**	**100%**	
	NMS	**13**	**46,264**	**591**	**1.3%**	
	FMA	**216**	**5,422,826**	**488**	**0.01%**	
	NERR*	**27**	**4,606**	**0**	**0.00%**	
	TOTAL	**410**	**6,118,773**	**354,820**	5.8%	

ALL FEDERAL MMAS†

New England: Maine to Connecticut, Mid Atlantic: New York to Virginia, South Atlantic: North Carolina to Florida. NP: National Parks, NWR: National Wildlife Refuges, NMS: National Marine Sanctuaries, FMA: Fishery Management Areas, NERR: National Estuarine Research Reserves, and NM: National Monuments.

* NERRs are state/federal partnership sites.

** Total area includes only those sites for which data are available.

*** The Northwestern Hawaiian Islands Marine National Monument is scheduled to become a no-take area in five years when all fishing is phased out. This site has been included in the no-take category and will be the largest no-take MPA in the United States.

† This total is corrected for overlapping jurisdictions of Federal MMAs.

Table 8.3. Sites in the National Marine Sanctuary Program. Regions: PC = Pacific Coast, PI = Pacific Islands, SE = Southeast Atlantic, Gulf of Mexico, and Caribbean, NE = Northeast (National Marine Sanctuary Program, 2006c).

Site	Location	Region	Year Designated	Size (km²)	Yr of First Mgt Plan	Status of Mgt Plan Revision
Channel Islands	CA	PC	1980	4,263	1983	2007 planned publication
Cordell Bank	CA	PC	1989	1,362	1989	Central CA Joint Mgt Plan Review[1]
Fagatele Bay	Amer. Samoa	PI	1986	0.66	1984	Ongoing
Florida Keys	FL	SE	1990	9,844	1996	2007 planned publication
Flower Garden Banks	TX	SE	1992	2.0	In preparation	
Gray's Reef	GA	SE	1981	58	1983	Published 2006
Gulf of the Farallones	CA	PC	1981	3,252	1983	Central CA Joint Mgt Plan Review
Hawaiian Islands HW[2]	HI	PI	1992	3,548	1997	Published 2002
Monitor[3]	NC	NE	1975	4.1	1997[4]	
Monterey Bay	CA	PC	1992	13,784	1992	Central CA Joint Mgt Plan Review
Olympic Coast	WA	PC	1994	8,573	1994	Ongoing
Papahānaumokuākea MNM[5]	HI	PI	2006	~360,000	In preparation	
Stellwagen Bank	MA	NE	1992	2,188	1993	2007 planned publication
Thunder Bay[3]	MI	NE	2000	1,160	1999	Ongoing
Key Largo[6]	FL		1975	353		
Looe Key[6]	FL		1981	18		

[1]The Central California Joint Management Plan Review is a coordinated process to obtain public comments on draft management plans, proposed rules, and draft environmental impact statements for the three Central California Sanctuaries.

[2]HW = humpback whale.

[3]The Monitor and Thunder Bay NMSs were designated for protection of maritime heritage resources (2007a; National Marine Sanctuary Program, 2007e).

[4]This plan is actually a comprehensive, long-range preservation plan for the Civil War ironclad U.S.S. *Moonitor*.

[5]The Papahānaumokuākea Marine National Monument is co-managed by NOAA/National Marine Sanctuary Program and National Marine Fisheries Service, U.S. Fish and Wildlife Service, and the State of Hawaii.

[6]The Key Largo and Looe Key NMSs were subsumed within the Florida Keys NMS as Existing Management Areas.

8.10 Figures

Figure 8.1. Locations of the 14 MPAs that compose the National Marine Sanctuary System (National Marine Sanctuary Program, 2006c).

Figure 8.2. Timeline of the designation of the national marine sanctuaries in the National Marine Sanctuary Program (National Marine Sanctuary Program, 2006a).

Figure 8.3. Map of the Florida Keys National Marine Sanctuary. The 1990 designation did not include the Tortugas Ecological Reserve located at the western end of the sanctuary, which was implemented in 2001. The Key Largo NMS corresponded to the Existing Management Area (EMA) just offshore of the John Pennekamp Coral Reef State Park; the Looe Key NMS corresponded to the EMA surrounding the Looe Key Sanctuary Preservation Area and Research Only Area (National Oceanic and Atmospheric Administration, 2007d).

Figure 8.4. Organizational chart of the National Marine Sanctuary Program (NOAA National Ocean Service, 2006).

Figure 8.5. Total observed change in coral cover (%) across the Caribbean basin over the past 25 years (Gardner *et al.*, 2003). A. Coral cover (%) 1977-2001. Annual estimates (▲) are weighted means with 95% bootstrap confidence intervals. Also shown are unweighted estimates (●), unweighted mean coral cover with the Florida Keys Coral Reef Monitoring Project (1996-2001) omitted (x), and the number of studies each year (○). B. Year-on-year rate of change (mean $\Delta N \pm$ SE) in coral cover (%) for all sites reporting two consecutive years of data 1975-2000 (●) and the number of studies for each two-year period (○).

Figure 8.6. Map of the Great Barrier Reef Marine Park showing the adjacent catchment in Queensland. Modified from Haynes (2001) and courtesy of the Great Barrier Reef Marine Park Authority.

Figure 8.7. Sea surface temperature (SST) projections for the Great Barrier Reef (GBR) (Lough, 2007).

Figure 8.8. Endemic species from the Hawaiian Islands. A. Masked angelfish, *Genicanthus personatus* (Photo: J. Watt), B. Rice coral, *Montipora capitata*, and finger coral, *Porites compressa* (photo: C. Hunter), C. Hawaiian hermit crab, *Calcinus laurentae* (photo: S. Godwin), D. Red alga, *Acrosymphtyon brainardii* (photo: P. Vroom).

Figure 8.9. a) NOAA Pathfinder SST anomaly composite during summer 2002 period of NWHI elevated temperatures, July 28–August 29. b) NASA/JPL Quikscat winds (wind stress overlayed by wind vector arrows) composite during summer 2002 period of increasing SSTs, July 16–August 13. The Hawaii Exclusive Economic Zone (EEZ) is indicated with a heavy black line; all island shorelines in the archipelago are also plotted (adapted from Hoeke et al., 2006).

Figure 8.10. Map of the Channel Islands National Marine Sanctuary showing the location of existing state and proposed federal marine reserves and marine conservation areas (Channel Islands National Marine Sanctuary, 2007).