7 National Estuaries

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7.1 Background and History

7.1.1 Historical Context and Enabling Legislation

This chapter focuses on meeting the challenges of management of national estuaries and estuarine ecosystem services under influence of changing climate. Our contribution is distinguished from previous reviews of estuarine responses to climate change (*e.g.*, National Coastal Assessment Group, 2000; National Assessment Synthesis Team, 2000; Scavia *et al.*, 2002; Kennedy *et al.*, 2002; Harley and Hughes, 2006) by its focus on developing adaptive management options and analyzing the characteristics of human and ecological systems that facilitate or inhibit management adaptation. The chapter is thus written mostly for an audience of natural resource and environmental managers and policy makers.

There are 28 national estuaries that comprise the U.S. National Estuarine Program, which is administered by the U.S. Environmental Protection Agency (Fig. 7.1). These estuaries span the full spectrum of estuarine ecosystem types and encompass the diversity of estuarine ecosystem services across the country. Estuaries are sometimes defined as those places where fresh and salt water meet and mix, thereby potentially excluding some largely enclosed coastal features such as marine lagoons and including, for some vigorous rivers like the Mississippi, extensive excursions into the coastal ocean. So as to match common characteristics of the 28 national estuaries, we choose an alternative, geomorphologically based definition of an estuary as a semi-enclosed body of water on the sea coast in which fresh and salt water mix (adapted from Pritchard, 1967). Such a definition includes not only those water bodies that are largely perpendicular to the coastline where rivers approach the sea, but also marine lagoons, which are largely parallel to the shoreline and experience only occasional fresh water inflow, thereby retaining high salinities most of the time. In the landward direction, we include the intertidal and supratidal shorezone to be part of the estuary and thus include marshes, swamps and mangroves, (*e.g.*, the coastal wetlands).

Figure 7.1. Organization of the NEP system (U.S. Environmental Protection Agency, 2007b).

Estuaries are notoriously idiosyncratic because of intrinsic differences among them in physical, geological, chemical, and biological conditions (Wolfe, 1986). There can also be considerable variation within an estuary. This variation exists over wide spectra of time and space (Remane and Schlieper, 1971). This high level of environmental variability in estuaries places physiological constraints on the organisms that can occupy them, generally requiring broad tolerances for varying salinity, but also for temperature and other factors. Consequently, the organisms of estuaries represent a biota that may have unusually high intrinsic capability for species-level physiological adaptation to changing salinity, temperature, and other naturally varying aspects of historic climate change. The challenge is to predict how these species will respond to accelerated rates of change and how species interactions will alter communities and ecosystems.

Estuaries possess several features that render them unusually valuable for their ecosystem services, both to nature and to humans. The biological productivity of estuaries is generally high, with substantial contributions from vascular plants of historically extensive tidal marshes and coastal wetlands as well as from seagrasses and other submerged aquatic vegetation. A large fraction of the fisheries of the coastal ocean depend on estuaries to provide nursery or even adult habitat necessary to complete the life cycle of the fish or shellfish. Similarly, many species of coastal wildlife including terrestrial and marine mammals and coastal birds depend on estuaries as essential feeding and breeding grounds. Although depicting the ecosystem services of only one estuarine habitat, the wetlands and marshes, the Millennium Ecosystem Assessment (2005) provides a table of ecosystem services that helps indicate the types and range of natural and human values that are vested in estuarine ecosystems more broadly (Box 7.1). Partly in recognition of the value of estuaries and the threats to their health, the National Estuary Program (NEP) was established by Congress in 1987 and housed within EPA (U.S. Congress, 1987) (see Fig. 7.1). After the establishment of this program, the 28 national estuaries were added over a ten-year period (Fig. 7.2).

Figure 7.2. Timeline of National Estuaries Program formation (U.S. Environmental Protection Agency, 2007a).

Estuaries represent the collection point past which runoff from the entire watershed must flow. The health and functioning of estuaries are at risk from those stressor pollutants discharged and released over the entire catchment area that reach these collection points. Degradation of estuarine habitats, water quality, and function is traceable to human modification of watersheds and the cumulative consequences are now substantial worldwide (Jackson *et al.*, 2001; Worm *et al.*, 2006; Lotze *et al.*, 2006). More recently, threats to estuaries have arisen from sources even closer to estuarine waters as human population migration and growth have targeted the coasts and especially waterfront property. Although more than half of the U.S. population now lives on the 17% of lands considered coastal, within the next 25 years human populations on the coast are expected to increase by 25% (National Coastal Assessment Group, 2000). Thus, the threats to estuarine ecosystems are not only widespread, requiring a basin-wide scope for management, but increasingly local as more people choose to occupy habitats of higher risk. The growing human occupation of estuarine shores increases the challenge of managing for climate change, because estuarine services are placed at growing risk from both direct impacts of changing climate as well as indirect consequences of human responses to personal and property risks from climate change.

7.1.2 Interpretation of National Estuary Program Goals

Under the goals of Section 320 of the Clean Water Act, each National Estuary[1] is required to develop a Comprehensive Conservation and Management Plan (CCMP). Many national estuaries have watersheds found within a single state, and therefore their CCMP is contained within one state. Other estuarine watersheds are trans-boundary and more than one state participates. Emphasis is on "integrated, watershed-based, stakeholder-oriented water resource management" (U.S. Environmental Protection Agency, 2006). These plans are produced by a full range of stakeholders within each National Estuary through a process involving (1) assessments of trends in water quality, natural resources, and uses of the estuary, (2) evaluation of appropriate data, and (3) development of pollutant loading relationships to watershed and estuarine condition. The final CCMP is approved by the governors of the states in the study area and the EPA administrator. The programs are then obligated to implement the CCMPs and monitor effectiveness of actions (U.S. Congress, 2002). Each National Estuary prepares an annual plan, approved by EPA, to guide implementation of its CCMP.

The national estuaries represent a wide variety of sizes, geomorphologies, and watershed characteristics. For example Santa Monica Bay is a relatively small open embayment or coastal lagoon, the Maryland Coastal Bays are a group of more closed lagoons, and the Albemarle-Pamlico Sound is a complex of drowned river valleys emptying into largely closed coastal lagoons. The Columbia River Estuary and the Delaware Estuary are the more traditional drowned river valleys. This diversity has largely prevented classification, grouping, and synthetic assessment of the constituent national estuaries. There is geographic separation into four regions: West Coast (six sites), Gulf of Mexico (seven sites), South Atlantic (six sites, including San Juan Bay, Puerto Rico), and Northeast (nine sites). Although the estuaries do not share easily identified geomorphic characteristics, they are recognized to share common stressors (Bricker *et al.*, 1999; Worm *et al.*, 2006; Lotze *et al.*, 2006). These stressors include "eutrophication, contamination from toxic substances and pathogens, habitat loss, altered freshwater inflows, and endangered and invasive species" (Bearden, 2001). This particular list ignores direct and indirect fishing impacts, which are important and included in many CCMPs. Even more importantly, this list fails to include the direct and indirect effects of climate change, particularly the threats posed by sea level rise.

A hallmark of the NEP is that it is largely a local program with federal support. While federal grants provide a critical source of base funding, most national estuaries have successfully raised significant local and state support, primarily to finance specific projects or activities. The individual national estuaries lack regulatory authority; thus, they depend on voluntary cooperation using various incentives plus existing federal, state, tribal, and local legislation and regulation. Their purpose is to coordinate these local efforts and promote the mechanisms to develop, implement, and monitor the CCMPs. The NEP was designed to provide funding and guidance for the 28 estuaries around the

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[1] In the National Estuary Program, individual National Estuaries are referred to as National Estuary Programs. To avoid confusion between individual estuary programs and the umbrella program, this document uses the term National Estuaries to refer to the individual programs.

country to work in a bottom-up science-based way within the complex policy-making landscape of federal, state, and local regulations. Non-regulatory strategies must complement the limited federal and even state authority or regulations. Lessons learned about how monitoring, research, communication, education, coordination, and advocacy work to achieve goals are transferable to all estuaries, not just NEP members.

The overarching areas of concern in national estuaries can be classified as water quality, fisheries, habitat, wildlife, introduced species, biodiversity, human values, and freshwater quantity. More specifically the goals include "protection of public water supplies and the protection and propagation of a balanced, indigenous population of shellfish, fish, and wildlife, and [allowing] recreational activities, in and on water, [and requiring]…control of point and nonpoint sources of pollution to supplement existing controls of pollution" (U.S. Congress, 1987). Thus, overwhelmingly the interest has been on anthropogenic impacts and their management (Kennish, 1999).

Within recent years, each National Estuary has developed or begun to develop system-specific ecosystem status indicators. These indicators allow ongoing assessments of the success of management activities resulting from the CCMPs. However, mention of climate change is missing from almost all CCMPs and only one National Estuary (Puget Sound) has completed a planning process to assess implications of climate change for the perpetuation of ecosystem services in its system (Snover *et al.*, 2005). Managers may fail to account for the effects of climate change on the estuaries if the choices of indicators are not reconsidered in the context of changing climate. Perhaps more importantly, climate change may confound the interpretation of the indicator trend results and thus the interpretation of the effectiveness of the CCMPs.

7.2 Current Status of Management Systems

7.2.1 Key Ecosystem Characteristics on Which Goals Depend

To understand how climate drivers might affect individual national estuaries, it is useful to identify the susceptibility of characteristics of the entire management system. At a large scale is the location of the estuary on Earth (*i.e.*, its latitude and longitude). Climate varies over the globe, and expectations for change likewise differ geographically on a global scale. Expected temperature and precipitation changes and range shifts can be estimated from global-scale geographic position quite well, whereas local variation of these and other variables (*e.g.*, winds) of climate change are less predictable.

Next in scale is the airshed. This is the area capable of influencing the estuary through the contribution of quantitatively significant pollutants, especially nitrogen oxides (NO_x). For the Chesapeake Bay, this area includes Midwestern states, the source of nutrients from industrial and transportation activities. Estuaries on the Gulf and East coasts are likely to have different dependencies on their airsheds for nutrient enrichment than their western counterparts. Western estuaries are affected more by fog banks emanating from coastal waters. Climate drivers that change wind, ultraviolet radiation, and precipitation patterns are particularly important at this scale.

Next in hierarchical context is the watershed. Central to the NEP is the watershed perspective to management. Land and watershed use, population density, and regulatory effectiveness combine to determine the potential loading of pollutants, extraction of freshwater and resources, and transformation of habitat and coastline. Climate change can influence each of these factors. Changes in temperature, sea level, storminess, precipitation, and evapotranspiration patterns can alter human settlement and migration, agricultural and fisheries practices, and energy and resource use. These responses are likely to be long-term and large-scale, although their influence on estuarine dynamics may be exhibited on shorter time scales. For example, seasonal nutrient loading varies as a result of changes in tourism or crop choice. These factors largely affect the concentration of nutrients, while changes in runoff and river flow affect the discharge component of loading.

At the opposite end of the estuary is the marine environment, which also serves as an intermixing boundary susceptible to climate change. The oceans and coastal marine waters have responded or are expected to respond to climate change by changes in sea level, circulation patterns, storm intensity, salinity, temperature, and pH. Some of these factors may change little over the large scale but may be altered locally outside the mouths of estuaries. All of these factors influence the biota, with all but pH exerting additional indirect effects by modifying estuarine hydrodynamics.

Susceptibility of individual estuaries to climate change depends on a number of characteristics that act at a variety of spatial and temporal scales. All of the previously mentioned climate drivers can affect estuaries. How they do so depends on physical features such as estuarine depth, size, and balance between ocean water circulation and fresh-water inflows. Furthermore, the geomorphology and direction of longest fetch set conditions for susceptibility to storms. All of these features help determine the biological communities that reside within the estuary and how they might respond to the various components of climate change.

The way in which a specific estuary responds to climate change depends on the anthropogenic stressors acting on it. These stressors include both those that pollute and contaminate the system and those that remove or disrupt estuarine resources. Pollutants include nutrients, metals, pathogens, sediments, and organic toxicants. Invasive species are additions that disrupt communities. Extractions include uses of fresh and brackish water, sediments, and living resources within the ecosystem. Disruption of a variety of biological communities occurs through fishing, habitat destruction, damming, boat traffic, and shoreline conversion and stabilization activities.

Finally, there are the social, political, and economic contexts for susceptibility. Some of these contexts play out in ways already mentioned. But it is clear that stakeholder attitudes about estuaries and their perceptions about climate change are critical to wise management for climate change. Each stakeholder group, indeed each individual, uses estuaries in different ways and places different importance on specific ecosystem services. One aim of this report is to provide a common body of knowledge to stakeholders and to managers at higher levels (local, state, tribal, and federal governments) to inform their choices.

7.2.2 Current Stressors of Concern

Estuaries are generally stressful environments because of their strong and naturally variable gradients of salinity, temperature, and other parameters. However, estuaries are also essential feeding and reproduction grounds, and provide refuge for a wide variety of seasonal and permanent inhabitants. Throughout history, estuaries have been focal points of human settlement and resource use, and humans have added multiple stressors to estuarine ecosystems (Lotze *et al.*, 2006). We define a stressor as an anthropogenic or naturally occurring environmental factor that adversely affects individual physiology, population performance, or ecosystem function when it extends beyond its typical range of variation (Vinebrooke *et al.*, 2004). This document focuses specifically on those stressors that significantly affect the services that estuaries are managed to provide. The major stressors currently imposed on estuaries are listed in Table 7.1. Almost all current efforts to manage estuarine resources are focused on these stressors (Kennish, 1999 and the various CCMPs).

Several stressors result from modified rates of loading of naturally occurring energy and materials. Nutrient loading is perhaps the most studied and important material addition. Although essential to the primary production of any open ecosystem, too much nutrient loading can cause eutrophication, the subject of considerable concern for estuaries and the target for much management action (Nixon, 1995; Bricker *et al.*, 1999). Nutrient (especially nitrogen N) loading comes from diverse point- and non-point sources including agriculture, aquaculture, and industrial and municipal discharges, and can lead to harmful and nuisance algal blooms, loss of perennial vegetation, bottom-water hypoxia, and fish kills. Sediment delivery has also been altered by human activities. Again, sediments are important to estuarine ecosystems as a material source for the geomorphological balance in the face of sea level rise and for nutrients (especially phosphorus P) for primary production. However, land clearing, agriculture, and urban land use can increase sediment load (Howarth, Fruci, and Sherman, 1991; Cooper and Brush, 1993; Syvitski *et al.*, 2005), while dams may greatly restrict delivery and promote deltaic erosion (Syvitski *et al.*, 2005). Historically, sediment loading has increased on average 25-fold and nitrogen and phosphorus loading almost 10-fold in estuaries since 1700 (Lotze *et al.*, 2006). Because riverine loading of both nutrients and sediments depends on their concentration and river flow, modifications of river flow will further alter the amount and timing of material delivery. River flow also contributes to the energy budget through mechanical energy. River flow may be a major determinant of flushing times, salinity regime, and stratification, and thus determine community structure and resource use patterns. Modifications in river flow come from dam management decisions, land development, loss of riparian wetlands, extraction of freshwater, and surface and ground water consumption. Thermal pollution, largely from power plants, is a direct enhancement of energy with resultant local changes in metabolic rates, community structure, and species interactions.

Human activities also cause or enhance the delivery of materials and organisms that are not normally part of the natural systems. Pathogen loading compromises the use of estuarine resources, causing shellfish bed closures and beach closures (*e.g.*, Health Ecological and Economic Dimensions of Global Change Program, 1998), human health advisories, and diseases to estuarine organisms themselves. Other anthropogenic contributions include the discharge and ongoing legacy of organic wastes and persistent organic pollutants (*e.g.*, DDT, dioxin, PCBs, petroleum) (Kennish, 1999). The toxicity of some of the persistent organic pollutants has been recognized for decades, dating to the publication of *Silent Spring* by Rachel Carson (Carson, 1962). More recently, the potential importance of other endocrine disrupting chemicals is causing concern (Cropper, 2005). Added to these organic pollutants are metals entering estuaries from direct dumping, riverine waters, sediments, and atmospheric deposition. Moreover, biodegradable organic wastes contribute to eutrophication and dissolved oxygen deficits (Nixon, 1995). Finally, the introduction and spread of non-indigenous species are enhanced by globalization and shipping, intentional decisions for commerce or other human use, and unintentional actions (Mooney and Hobbs, 2000). For those locations that have been surveyed, the number of known numbers of resident non-indigenous species ranges from about 60 to about 200 species per estuary in the United States (Ruiz *et al.*, 1997; Lotze *et al.*, 2006), and are likely the result of an increasing rate of invasions over the last 300 years (Lotze *et al.*, 2006).

Use and development in and around estuaries alter wetland and subtidal habitats directly. Wetland destruction has occurred during much of human history as a result of the perceptions of wetlands as wastelands and the value of waterfront land. For example, 12 estuaries around the world have lost an average of more than 65% of wetland area (with a range of 20–95%) over the last 300 years (Lotze *et al.*, 2006). Wetland habitat loss from development continues despite changes in perceptions about wetland value and regulations intended to protect wetlands. Coastal wetlands represent a diverse assortment of hyrdogeomorphic classes (Brinson, 1993; Christian *et al.*, 2000), both sea-level controlled (*e.g.*, marshes and mangroves), non-sea-level controlled (*e.g.*, swamps, fens, bogs, and pocosins) and subtidal (*e.g.*, submerged aquatic vegetation (SAV), seagrass, and macroalgal) habitats. Supratidal and intertidal wetlands are subject to land use change, dredging and filling, and changes in water quality. Subtidal habitats are particularly susceptible to not only these impacts but also activities within the water. For example, SAV loss also occurs from bottom-disturbing fishing practices and eutrophication. Oyster reef habitat destruction occurs from direct exploitation and bottom disturbance from fishing practices (*e.g.*, trawling). For 12 study sites around the world, both seagrass meadows and oyster reefs have experienced substantial losses over the last 300 years (about >65% and 80%, respectively) (Lotze *et al.*, 2006). Together with the loss of wetlands, these changes have resulted in great reductions of essential nursery habitats, important filtering functions (nutrient cycling and storage), as well as coastal protection (barriers and floodplains) in estuaries (Worm *et al.*, 2006; Lotze *et al.*, 2006).

Another important anthropogenic stressor in estuaries is the extraction of living and non-living material that alters estuarine ecosystem structure and functioning. Historically, estuaries provided a wide variety of resources used and valued by humans as sources of food, fur, feathers, fertilizer, and other purposes (Lotze *et al.*, 2006). Since the 19th century, however, the ecological service of estuaries receiving greatest management attention has been their support of fisheries. Pollution, damming, and habitat destruction affect fisheries. Recently more emphasis has been placed on overfishing as a negative impact, not only on the target species but also on the community and food web structure (*e.g.*, Dayton, Thrush, and Coleman, 2002). Large apex predators have been greatly reduced from many if not most estuarine and coastal ecosystems (Lotze *et al.*, 2006). The absence of these large consumers (including marine mammals, birds, reptiles, and larger fish) translates through the food web, creating ecosystem states that are distinct from those of the past (*e.g.*, Jackson *et al.*, 2001; Lotze *et al.*, 2006; Myers *et al.*, 2007). Ongoing fishing pressure targets species lower and lower in the food chain, including detritivorous and herbivorous invertebrates and marine plants, further altering ecosystem structure and functioning and undermining habitat integrity and filtering functions (Pauly *et al.*, 1998; Worm *et al.*, 2006; Lotze *et al.*, 2006). Management goals to stabilize current or restore former ecosystem states are jeopardized if large consumers are not also recovered (Jackson *et al.*, 2001).

It is rare that an estuary is subject to only one of these stressors. Management decisions must consider not only stressors acting independently but also interacting with each other (Breitburg, Seitzinger, and Sanders, 1999; Lotze *et al.*, 2006). Multiple stressors can interact and cause responses that cannot be anticipated from our understanding of each one separately. For example, Lenihan and Peterson (1998) demonstrate that habitat damage from oyster dredging and the stress of bottom-water hypoxia interact to affect oyster survival. Tall oyster reefs, both those that remain and those that have been rebuilt, project above hypoxic bottom waters and therefore allow oyster survival in the upper wind-mixed layers even as water quality further deteriorates. Unfortunately, management of fisheries and water quality is done by different agencies, inhibiting the integrated approach that such interacting stressors demand.

Interactive effects of multiple stressors are likely to be common and important because of both the interdependence of physiological rate processes within individuals and the interdependence of ecological interactions within communities and ecosystems (Breitburg and Riedel, 2005). Individual stressors fundamentally change the playing field upon which additional stressors act by selecting for tolerant species while also changing the abundance, distribution, or interactions of predators, prey, parasites, hosts, and structural foundation species (*e.g.*, organisms such as bivalves and corals that create physical structures upon which other species depend). These direct and indirect effects can be common when stressors occur simultaneously, but they also occur from exposure to stressors in sequence. Across hierarchical levels from individuals through ecosystems, the recovery period from a particular stressor can extend beyond the period of exposure, thus influencing responses to subsequent stressors. For example, Peterson and Black (1988) demonstrated that bivalves that were already stressed from living under crowded conditions exhibited higher mortality rates after experimental application of the stress of sedimentation. Moreover, effects of stressors on indirect interactions within populations and communities can extend the spatial scale of stressor effects and delay recovery (Peterson *et al.*, 2003), increasing the potential for interactions with additional stressors.

For example, years after the Exxon Valdez oil spill, female harlequin ducks (1) exposed to lingering oil during feeding on benthic invertebrates in contaminated sediments and (2) exhibiting activation of detoxification enzymes suffered lower survivorship over winter. Winter is a period of energetic stress to these small-bodied ducks (Peterson *et al.*, 2003). On longer time scales, heritable adaptations that increase tolerance to one class of stressors may enhance susceptibility to others (Meyer and Di Giulio, 2003).

One hallmark of the NEP is the recognition that management actions need to take account of the complexity of the larger watershed and the potentially diverse socioeconomic demands and objectives within them. The NEP tracks habitat restoration and protection efforts with annual updates from the component estuaries (U.S. Environmental Protection Agency, 2007c).

7.2.3 Legislative Mandates Guiding Management of Stressors

Because of the intrinsically wide range of estuarine resources and diversity of human activities that influence those estuarine resources, management of estuarine services is achieved via numerous legislative acts at the federal level. Many of these acts possess state counterparts, and local laws—especially land use planning and zoning—also play roles in management of estuarine services. This web of legal authorities and guiding legislation is an historical legacy, reflective of prevailing management that compartmentalized responsibilities into multiple agencies and programs.

The presentation here of applicable federal legislative acts is long, yet incomplete, and does not attempt to list state and local laws. One motivation in providing this spectrum of applicable legislation is to illustrate the challenges involved for estuaries in the integration of management authorities that is urged under the umbrella of ecosystem-based management by the U.S. Commission on Ocean Policy.

7.2.3.1 Basin-Wide Management of Water Quality

As one of the tools to meet the goal of “restoration and maintenance of the chemical, physical, and biological integrity of the Nation’s waters” under §402 of the Federal Water Pollution Control Act (U.S. Congress, 2002), any entity that discharges pollutants into a navigable body of water must possess an National Pollutant Discharge Elimination System (NPDES) permit. This includes public facilities such as wastewater treatment plants, public and private industrial facilities, and all other point sources. While EPA was the original administrator of the program, many states have now assumed this function. All states have approved State NPDES Permit Programs except Alaska, The District of Columbia, Idaho, Massachusetts, New Hampshire, New Mexico, and the territories and trusts (American Samoa, Guam, Johnston Atoll, Midway Island, Northern Marianas, Puerto Rico, the Trust Territories and Wake Island). All those without approved State NPDES Permit Programs are administered directly by EPA. The only unapproved states with estuaries (disregarding the trusts and territories) are then the District of Columbia, Massachusetts and New Hampshire. As of 1987, NPDES permits were also required for some storm water discharges, beginning with larger urbanized entities and recently extending to some medium-sized units of government who own or operate municipal storm water discharge facilities.

Although the content, style, and length of any given NPDES permit for point-source discharge will be slightly different depending on where and when it is written, all permits contain certain core components mandated by the Clean Water Act, including testing, monitoring, and self reporting. NPDES permits are renewed every five years, and monitoring and/or reporting requirements may change. These changes are determined by the local Regional Water Quality Control Boards or the State Water Resources Control Board through their research and monitoring efforts.

7.2.3.2 Habitat Conservation under Federal (Essential Fish Habitat) and State Fishery Management Plans

As administered under NOAA, the Magnuson Fishery Conservation and Management Act of 1976 (amended as the Sustainable Fisheries Act (SFA) in 1996 [P.L. 94-265] and reauthorized as Magnuson-Stevens Fishery Conservation and Management Reauthorization Act (MSA) of 2006 [P.L. 109-479]) established eight regional fishery management councils that are responsible for managing fishery resources within the federal 200-mile zone bordering coastal states. Management is implemented through the establishment and regulation of Fishery Management Plans (FMPs). In addition to “conservation and management of the fishery resources of the United States…to prevent overfishing, rebuild overfished stocks and insure conservation,” the Act also mandates the facilitation of long-term protection of *essential fish habitats*, which are defined as “those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity” (U.S. Congress, 1996). The Act states “One of the greatest long-term threats to the viability of commercial and recreational fisheries is the continuing loss of marine, estuarine, and other aquatic habitats.” It emphasizes that habitat considerations “should receive increased attention for the conservation and management of fishery resources of the United States” (U.S. Congress, 1996) and “to promote the protection of essential fish habitat in the review of projects conducted under Federal permits, licenses, or other authorities that affect or have the potential to affect such habitat” (U.S. Congress, 1996).

FMPs prepared by the councils (or by the Secretary of Commerce/NOAA) must describe and identify essential fish habitat to minimize adverse effects on such habitat caused by fishing. In addition, they must identify other actions to encourage the conservation and enhancement of essential fish habitat and include management measures in the plan to conserve habitats, “considering the variety of ecological factors affecting fishery populations” (U.S. Congress, 1987).

Because managed species use a variety of estuarine/coastal habitats throughout their life histories, few are considered to be “dependent” on a single, specific habitat type (except, for example, larger juvenile and adult snappers and groupers on ocean hard bottoms) or region. As a result, federal FMPs do not comprehensively cover species’ habitats that are not specifically targeted within their region. In addition, the only estuarine-dependent fish stocks under federal management authority are migratory stocks, such as red drum and shrimp, so estuarine habitats are not a key focus for essential fish habitat. However, many states also have FMPs in place or in preparation for target fisheries under their jurisdiction (the non-migratory inshore species) and participate with the regional councils under the SFA/MSA.

Thus, threats to marshes and other estuarine systems that constitute essential fish habitat or state-protected fisheries habitat should include all potential stressors, whether natural or anthropogenic, such as climate change and sea level rise. Although essential fish habitats have been codified for many fisheries, and science and management studies have focused on the status and trends of fisheries-habitat interactions, most management consideration has targeted stresses caused by different types of fishing gear. Because few fisheries take place in emergent marshes, the essential fish habitat efforts have not provided much protection to this important habitat. Seagrass and oyster reef habitats have been targeted for additional management concern because of the federal essential fish habitat provisions. State protections of fishery habitat vary, but generally include salt marsh and other habitats. Nearly two decades ago, EPA projected extensive loss of coastal marshes and wetlands from sea level rise by 2100, with an elimination of 6,441 square miles (65%) of marshes in the continental United States associated with a probable rise of 1m (Park *et al.*, 1989).

7.2.3.3 Estuarine Ecosystem Restoration Programs

While comprehensive planning of coastal restoration is inconsistent at the national level, a number of national, regional, and local programs are coordinated to the extent that stressors are either the target of restoration or addressed as constraints to restoration. These programs tend to be oriented toward rehabilitation of injuries done by individual stressors, such as eutrophication or contaminants, or toward restoration of ecosystems that have not been so extensively modified that their loss or degradation is not irreversible. Federal programs that authorize restoration of estuaries include:

Estuary Restoration Act of 2000 (P.L. 106-457, Title I)

Probably the most prominent federal program that involves non-regulatory restoration in the nation's estuaries is the Estuary Restoration Act of 2000 (ERA). The ERA promotes estuarine habitat restoration through coordinating federal and non-federal restoration activities and more efficient financing of restoration projects. It authorizes a program under which the Secretary of the Army through the Corps of Engineers (USACE) may carry out projects and provide technical assistance to meet the restoration goal. The purpose of the Act is to promote the restoration of estuarine habitat; to develop a national Estuary Habitat Restoration Strategy for creating and maintaining effective partnerships within the federal government and with the private sector; to provide federal assistance for and promote efficient financing of estuary habitat restoration projects; and to develop and enhance monitoring, data sharing, and research capabilities. Guidance provided by an Estuary Habitat Restoration Council consisting of representatives of NOAA, EPA, USFWS, and USACE includes soliciting, evaluating, reviewing, and recommending project proposals for funding; developing the national strategy; reviewing the effectiveness of the strategy; and providing advice on development of databases, monitoring standards, and reports required under the Act. The Interagency Council implementing the ERA published a strategy in December of 2002 with the goal of restoring one million acres of estuarine habitat by the year 2010. Progress toward the goal is being tracked via NOAA's National Estuaries Restoration Inventory.

Although the guiding principles that contributed to the development of this legislation argued for the "need to learn more about the effects of sea level rise, sedimentation, and a host of other variables to help set appropriate goals and success indicators for restoration projects in their dynamic natural environments," climate change is not explicitly addressed in the ERA (U.S. Congress, 2000). Similarly, the Council's Estuarine Habitat Restoration Strategy, published in 2002, neglects to explicitly mention climate change or sea level rise.

National Estuary Program and National Monitoring Program (EPA)

The National Estuary Program (NEP), administered under Section 320 of the 1987 amended Clean Water Act, focuses on point- and non-point source pollution in targeted, high-priority estuarine waters. Under the NEP, EPA assists state, regional, and local governments, landowners, and community organizations in developing a Comprehensive Conservation and Management Plan (CCMP) for each estuary. The CCMP characterizes the resources in the watershed and estuary and identifies specific actions to restore water quality, habitats, and other designated beneficial uses. Each of the 28 national estuaries has developed a CCMP to meet the goals of Section 320. Because the primary goal of the National Estuary Program is maintenance or restoration of water quality in estuaries, the CCMPs tend to focus on source control or treatment of pollution. Estuarine habitat restoration and protection is tracked by EPA's National Estuaries Program, with annual updates using information provided by the constituent national estuaries (U.S. Environmental Protection Agency, 2007c). While climate change is not considered a direct stressor, it is gradually being addressed in individual CCMPs in the context of potential increased nutrient loading from watersheds under future increased precipitation. For instance, the Hudson River Estuary Program has initiated with other partners an ongoing dialogue about how climate change constitutes a future stressor of concern to the estuary and its communities (New York State Department of Environmental Conservation, 2006). The Puget Sound and Sarasota Bay Estuary Programs have been the most proactive relative to anticipating a range of climate change challenges, although these assessments have been completed only recently.

7.2.3.4 State Sedimentation and Erosion Control, Shoreline Buffers, and Other Shoreline Management Programs Involving Public Trust Management of Tidelands and Submerged Lands

Protection from shoreline erosion has a long legal history, as far back as the tenets of property law established under the court of Roman Emperor Justinian (Spyres, 1999). In general, property law protection of tidelands held in public trust (most of the U.S. coastline) is conveyed either as the *law of erosion* (public ownership migrates inland when shores erode) or the *public trust doctrine* (the state holds tidelands in trust for the people unless it decides otherwise). Shoreline planners in many states (*e.g.,* Texas, Maine, Rhode Island, South Carolina, and Massachusetts) use these laws to plan for natural shoreline dynamics, including policies and tools such as "rolling easements" (*i.e.,* as the sea rises, the public's easement "rolls" inland; owners are obligated to remove structures if and when they are threatened by an advancing shoreline), setbacks (*i.e.*, prohibitions against development of certain areas at a set distance from the shoreward property line), prohibition of future shoreline armoring, and direct purchase of land that will allow wetlands or beaches to shift naturally (Spyres, 1999; IPCC, 2001). Some states are beginning to prohibit new structures in areas likely to be eroded in the next 30-60 years (*e.g.*, North Carolina through its Coastal Resources Commission).

7.2.3.5 Species Recovery under Federal Endangered Species Act

Recovery plans for aquatic species that are threatened or endangered under the Endangered Species Act (ESA) (U.S. Congress, 1973) may be contingent on implicit assumptions about habitat conditions in the coastal zone. However, explicit accounting for impacts and strategic designing of recovery efforts to consider climate variability and change is rare. A recent analysis of current ESA recovery plans indicates that of 101 plans that mention climate change, global warming, or related terms, only 60 actually discuss these topics, and only 47 identify climate change or its effects as a threat, possible threat, or factor in the species’ decline (Jimerfield, Waage, and Snape, 2007). Strategies and approaches that specifically address climate include monitoring for metapopulation variability that could link climate variation to extinction/recolonization probabilities or to unpredictable changes in existing or proposed future habitat. For example, the NOAA recovery plan for the Hawaiian monk seal (*Monachus schauinslandi*) suggests that habitat loss that has already been observed could be exacerbated by “…sea level rise over the longer term [that] may threaten a large portion of the resting and pupping habitat…” (National Marine Fisheries Service, 2006).

Climate variability and change will undoubtedly involve an even more consequential response by diadromous fishes and macroinvertebrates that require extensive, high-quality juvenile or adult transitional habitats during migrations between ocean and estuarine or freshwater aquatic systems. For example, in the Pacific Northwest and Alaska, sea level rise and shifts in timing and magnitude of snowmelt-derived riverine runoff may be particularly exacerbated by climate variability and change. Consequently, the recovery plans for threatened or endangered Pacific salmon (*e.g.*, juvenile, “ocean-type” Chinook [*Oncorhynchus tshawytscha*] and summer chum [*O. keta*] salmon) may need to account for their extreme sensitivity to climate-induced changes in environmental conditions of their estuarine wetland habitats during different life stages of the fish.

7.2.3.6 Wetland Protection Rules Requiring Avoidance, Minimization, and Mitigation for Unavoidable Impacts

Federal jurisdiction of waters of the United States began in 1899 with the Rivers and Harbors Act of 1899 and wetlands were included in that definition with the passing of the Clean Water Act of 1977 (CWA). This jurisdiction does not extend beyond the wetland/upland boundary. However, many state environmental laws, such as those of New York (*e.g.*, New York State, 1992) and New Jersey, require permits for alterations in adjacent upland areas in addition to protecting the wetland itself. While not originally intended for the purpose of increasing climate change preparedness, many of these regulations could facilitate adaptation to sea level rise (Tartig *et al.*, 2000).

The U.S. Army Corps of Engineers regulates dredging, the discharge of dredged or fill material, and construction of structures in waterways and wetlands through Section 404 of the CWA (codified generally as 33 U.S.C. §1251; 1977), the provisions of which have been amended progressively through 1987. Although not explicitly required within the language of the amended law, the CWA provides the Corps with the implicit authority to require that dredge or fill activities avoid or minimize wetland impacts (Committee on Mitigating Wetland Losses, National Research Council, 2001). The Corps and EPA developed criteria (Section 404(b)(1) guidelines) that over the years (latest, 1980) have defined mitigation as both minimization of wetland impacts and compensation for wetland losses. Thus, mitigation has been loosely interpreted to include a range of actions from wetland restoration and enhancement to creation of wetlands where they have never occurred (U.S. Congress, 1980). However, a 1990 memorandum of agreement (MOA) between the Corps and EPA established that mitigation must be applied sequentially. In other words, an applicant must first avoid wetland impacts to the extent practicable, then minimize unavoidable impacts, and finally—only after these two options are reasonably rejected—compensate for any remaining impacts through restoration, enhancement, creation, or in exceptional cases, preservation (Committee on Mitigating Wetland Losses, National Research Council, 2001). The Corps now grants permits for shoreline development that include armoring of the present shoreline, which guarantees future loss of wetlands as sea level rises, thereby violating the requirement for mitigation in the application of this authority (Titus, 2000).

7.2.3.7 Compensatory Restoration Requirements for Habitat and Natural Resource Injuries from Oil Spills or Discharges of Pollutants

Federal legislation requires compensatory restoration of estuarine habitats and natural resources after environmental incidents such as spills of oil or other toxicants (*e.g.*, Fonseca, Julius, and Kenworthy, 2000). For example, the Oil Pollution Act of 1990 specifies the procedures that federal agencies are required to follow to assess injury from pollution events and to conduct quantitatively matching restoration actions so the responsible parties replace the lost ecosystem services. Similar federal legislation, such as the Comprehensive Environmental Response, Compensation, and Liability Act, also specifies formation of natural resource trustees composed equally of state and federal agencies to oversee the injury assessments, pursue funding from the responsible party(ies) sufficient to achieve restoration, and then to design and implement the restoration. The process of restoration typically involves rehabilitation of biogenic habitats such as salt marshes, seagrass beds, or oyster reefs. The modeling done to insure that the restoration will provide ecosystem services equal to the injuries may need to be modified to reflect impacts of global climate change because services from habitat restorations are assumed to extend for years and even decades in these computations.

7.2.3.8 Federal Legislation Controlling Location of Ballast Water Release to Limit Introduction of Non-Indigenous Marine and Estuarine Species

One of the more troubling implications of climate change for estuaries is the probability of expanded distributions of non-indigenous species with the potential of progressively warmer waters in temperate zones. Ballast water discharged from ships in harbors after transiting from foreign ports (and domestic estuaries with extensive species invasions, such as San Francisco Bay) is one of the major sources of aquatic nuisance species. The primary federal legislation regulating ballast water discharge of invasive species is the National Invasive Species Act of 1996, which required the Coast Guard to establish national voluntary ballast water management guidelines. Because of a lack of compliance under the initial nationwide self-policing program that began in 1998, the voluntary program became mandatory in 2004. All vessels equipped with ballast water tanks that enter or operate within U.S. waters must now adhere to a national mandatory ballast water management program and maintain a ballast water management plan. Ballast water discharge may fall under the scope of the Clean Water Act, which adjudication may resolve.

7.2.3.9 Flood Zone Regulations

Tidal flood surge plains will likely be the estuarine regions most susceptible to climate change forcings, with consequent effects on human infrastructure, especially as development pressures continue to increase along the nation's coastal zone. Before the more recent projections of (higher) sea level rise rates, the Federal Emergency Management Agency (Federal Emergency Management Agency, 1991) estimated that existing development in the U.S. Coastal Zone would experience a 36%–58% increase in annual damages for a 0.3-meter rise in sea level, and a 102%–200% percent increase for a 1-meter rise. While state and local governments regulate building and other human activities in existing flood hazard zones, FEMA provides planning assistance by designating Special Flood Hazard Areas and establishing federal flood insurance rates according to the risk level.

7.2.3.10 Native American Treaty Rights

More than 565 federally recognized governments of American Indian and other indigenous peoples of Alaska, Hawaii, and the Pacific and Caribbean islands carry unique status as "domestic dependent nations" through treaties, Executive Orders, tribal legislation, acts of Congress, and decisions of the federal courts (National Assessment Synthesis Team, 2000). While climate variability and change are likely to impinge on all of these tribal entities, the impacts will perhaps be most strongly felt on the large coastal Native reservations, which are integrally linked to tourism, human health, rights to water and other natural resources, subsistence economies, and cultural resources. While these Native peoples have persisted through thousands of years of changes in their local environment, including minor ice ages, externally driven climate change will likely be more disruptive of their long, intimate association with their environments. In some cases, climatic changes are already affecting Natives such as those in Alaska who are experiencing melting of permafrost and the dissolution of marginal sea ice, altering their traditional subsistence-based economies and culture.

Where climate variability and change intersect with resource management of shared natural resources, Natives' treaty status may provide them with additional responsibility and influence. For example, on the basis of the "Boldt II decision," treaty tribes in

Washington State have treaty-based environmental rights that make them legal participants in natural resource and environmental decision making, including salmon and shellfish habitat protection and restoration (Brown, 1993; 1994).

7.2.4 Sensitivity of Management Goals to Climate Change

7.2.4.1 Climate Change and Changing Stressors of Estuarine Ecosystems

Many estuarine properties are expected to be altered by climate change. Global-scale modeling has rarely focused on explicit predictions for estuaries because realistic estuarine modeling would require very high spatial and temporal resolution. It is, however, reasonable to assume that estuaries will be forced by the same climate forcing that affects the coastal and marginal oceans. With increases in atmospheric CO_2, models project increases in oceanic temperature and stratification, decreases in convective overturning, decreases in salinity in mid- and high latitudes, longer growing seasons in mid- and high latitudes, and increases in cloud cover (Table 7.2). Such changes will necessarily force significant alterations in the physics, chemistry, and biology of estuaries. In particular, climate change may have significant impacts on those factors that are included in the definition of an estuary (Box 7.2). For example, climate-driven alterations to geomorphology will affect every physical, chemical, biological, and social function of estuaries.

The 2007 IPCC report provides a summary of the results of multiple credible models of climate change, providing various ranges of estimated change by year 2100. Whereas these predictions carry varying degrees of uncertainty and in some cases fail to include processes of likely significance in the modeling because of high scientific uncertainty, these predictions of rates of change over the next century help ground our scenario building for consequences of climate change on estuarine dynamics and on ability to attain management goals. The best estimates of average global temperature rise in the surface atmosphere vary from a low scenario of 1.1–2.9°C and a high scenario of 2.4–6.4°C. Scenarios of sea level rise range from a low prediction of 0.18–0.38 m to a high prediction of 0.26–0.59 m by 2100. The modeled sea level does not, however, include enhanced contributions from shifts of the Greenland and Antarctic ice shelves and could therefore be a serious underestimate. The future temperatures for Greenland reach levels inferred to have existed in the last interglacial period 125,000 years ago, when paleoclimate information suggests reductions of polar ice extent and a 4–6 m rise in sea level. The IPCC projects growing acidification of the ocean with reductions in pH of between 0.14 and 0.35 units over the next century. In our report, so as to standardize our framework for climate change across responses, we discuss a short term of two to three decades and also project the consequences of a 1 m rise in sea level. This increase may not occur within the next century, but if ice sheet shifts add to the present rate of sea level rise, a 1 m increase may occur sooner than the IPCC (2007) projections.

Climate change may also modify existing stressors (described in Section 7.2.2) and create new ones not discussed above. For example, the nutrient, sediment, pathogen, and contaminant stressors usually carried downstream with freshwater runoff will change in proportion to that runoff. If runoff increases, it can be expected to deliver more deleterious material to estuaries, leading to increased eutrophication via nutrients, smothering of benthic fauna via sediment loading, decreased photosynthesis via sediment turbidity, decreased health and reproductive success via a wide spectrum of toxins, and increased disease via pathogens. In contrast, "novel" stressors created by climate change include increased temperatures, shifts in the timing of seasonal warming and cooling, and the acidification caused by increased CO_2 (Box 7.3).

Importantly, there are likely to be interactions among existing and novel stressors, between those factors that define estuaries and stressors, and between stressors and existing management strategies. As noted above (Section 7.2.2), interactions among the multiple stressors posed by climate change are likely to pose considerable challenges. Nonetheless, it is important for successful natural resource management and conservation that managers, researchers, and policy makers consider the myriad stressors to which natural systems are exposed. Importantly, interactions among multiple stressors can change not only the magnitude of stressor effects, but also the patterns of variability and predictability on which management strategies rely (Breitburg *et al.*, 1998; Breitburg *et al.*, 1999; Worm *et al.*, 2006). Enhancing ecosystem resilience by establishing better controls on current stressors would limit the strength of interactions with climate change.

7.2.4.2 Impacts to and Responses of the Ecosystem

7.2.4.2.1 Temperature Effects on Species Distributions

Because species distributions are determined in part by physiological tolerances of climatic extremes, ecologists expect that species will respond to climate warming by shifting distributions towards the poles so long as dispersal and resources allow such shifts (Walther *et al.*, 2002). In fact, a wide array of species is already responding to climate warming worldwide (Walther *et al.*, 2002; Parmesan and Yohe, 2003; Root *et al.*, 2003; Parmesan and Galbraith, 2004; Parmesan, 2006). Global meta-analyses of 99 species of birds, butterflies, and alpine herbs demonstrate that terrestrial species are migrating poleward at a rate of 6.1 km per decade (Parmesan and Yohe, 2003). Moreover, 81% of 920 species from a variety of habitats showed distributional changes consistent with recent climate warming (Parmesan and Yohe, 2003). In marine systems, warm water species of zooplankton, intertidal invertebrates, and fish have migrated into areas previously too 'cool' to support growth (Barry *et al.*, 1995; Southward, Hawkins, and Burrows, 1995; Walther *et al.*, 2002; Southward *et al.*, 2004). Some copepod species have shifted hundreds to 1,000 kilometers northward (Beaugrand *et al.*, 2002), and the range of the oyster parasite *Perkinsus marinus* expands in warm years and contracts in response to cold winters (Mydlarz, Jones, and Harvell, 2006). Its range expanded 500 kilometers from Chesapeake Bay to Maine during one year—1991—in response to above-average winter temperatures (Ford, 1996) .

It is important to keep in mind that each species responds individualistically to warming: ecological communities do not move poleward as a unit (Parmesan and Yohe, 2003; Parmesan, 2006). This pattern was first demonstrated by paleoecological studies tracking the poleward expansions of individual species of plants following Pleistocene glaciation (*e.g.*, Davis, 1983; Guenette, Lauck, and Clark, 1998) and has since been extended to animals in phylogeographic studies (*e.g.*, Turgeon *et al.*, 2005). Climate warming is therefore likely to create new mixes of foundation species, predators, prey, and competitors. For example, “invading” species may move poleward faster than “resident” species retreat, potentially creating short-term increases in species richness (Walther *et al.*, 2002). Competitive, plant-herbivore, predator-prey, and parasite-host interactions can be disrupted by shifts in the distribution, abundance, or phenology of one or more of the interacting species (Walther *et al.*, 2002; Parmesan, 2006). Not surprisingly, therefore, it is difficult, if not impossible, to predict how community dynamics and ecosystem functioning will change in response to species shifts (Walther *et al.*, 2002).

Evidence from studies that have monitored changes in marine biota over the last three decades has shown that in coastal waters, the response of annual temperature cycles to climate change is both seasonally and regionally asymmetric. Along the mid-Atlantic East Coast, maximal summer temperatures are close to 30°C. When greenhouse gas forcing provides more heat to the surface waters in summer, they do not get warmer; instead the additional heat increases evaporation and is transferred to the atmosphere as a latent heat flux. Consequently maximum summer temperatures have not changed in the mid-Atlantic regions, but the minimum winter temperatures are now dramatically higher, by as much as 1-6°C (Parker Jr. and Dixon, 1998). In the reef fish community off North Carolina, the reduction over 30 years in winter kill during the coldest months made it possible for two new (to the area) families and 29 new species of tropical fishes to become permanent residents on the reef (Parker Jr. and Dixon, 1998). In addition, the 28 species of tropical reef fishes that have been present on the site for the entire three decades increased in abundance. An increase in fish-cleaning symbiosis was especially noticeable. Over the 30-year study period, no new temperate species became permanent residents and, while no temperate species dropped out of the community, the temperate species that was most abundant at the start of the study decreased in abundance by a factor of 22. This kind of seasonal asymmetry in temperature change expands the range of tropical species to the north, but so far has not changed the southern limit of temperate species—although it has reduced the biomass of temperate species that were previously abundant.

On the West Coast, changes in the species composition of a rocky intertidal community showed that between the 1930s and 1990s most species’ ranges shifted poleward (Barry *et al.*, 1995). The abundance of eight of nine southern species increased and the abundance of five of eight northern species decreased. Annual mean ocean temperatures at the central California coastal site increased by 0.75°C during the past 60 years, but more importantly the monthly mean maximum temperatures during the warmest month of year were 2.2°C warmer. On the West Coast, summer conditions are relatively cool and foggy due to strong coastal upwelling that produces water temperatures from 15–20°C. For intertidal organisms adapted to these relatively cool summer temperatures a 2°C increase in monthly mean temperature during the warmest month of the year was enough to decrease survival of northern species and increase the survival of southern species. It is clear that climate change has already altered the species composition and abundance of marine fauna, but is equally clear that the physical and biological response of organisms to warming in marine waters is extremely complex.

These effects of temperature on species distributions have influenced and will continue to influence fish and wildlife populations, and will modify habitat provided by organisms such as mangroves, requiring many site-specific adaptive modifications in management.

7.2.4.2.2 Temperature Effects on Risks of Disease and Parasitism

Not only will species' distributions change, but scientists expect that higher temperatures are likely to lead to increased risks of parasitism and disease, due to changes in parasites and pathogens as well as host responses (Harvell *et al.*, 2002; Hakalahti, Karvonen, and Valtonen, 2006). For example, temperature has the potential to alter parasite survival and development rates (Harvell *et al.*, 2002), geographic ranges (Harvell *et al.*, 2002; Poulin, 2005; Parmesan, 2006), transmission among hosts (Harvell *et al.*, 2002; Poulin, 2005), and local abundances (Poulin, 2005). In particular, shortened or less-severe winters are expected to increase potential parasite population growth rates (Hakalahti, Karvonen, and Valtonen, 2006). On the host side, increased temperatures can alter host susceptibility (Harvell *et al.*, 2002) by compromising physiological functioning and host immunity (Mydlarz, Jones, and Harvell, 2006). Animals engaged in partnerships with obligate algal symbionts, such as anemones, sponges, and corals, are at particular risk for problems if temperatures alter the relationship between partners (Mydlarz, Jones, and Harvell, 2006).

Reports of marine diseases in corals, turtles, mollusks, marine mammals, and echinoderms have increased sharply over the past three decades, especially in the Caribbean (Harvell *et al.*, 2002; Ward and Lafferty, 2004). For example, temperature-dependent growth of opportunistic microbes has been documented in corals (Ritchie, 2006). Poulin and Mouritsen (2006) documented a striking increase in cercarial production by trematodes in response to increased temperature, with potentially large effects on the intertidal community (Poulin and Mouritsen, 2006). Geographic range expansion of pathogens with broad host ranges is of particular concern because of the potential to affect a broad array of host species (Dobson and Foufopoulos, 2001; Lafferty and Gerber, 2002).

Importantly, however, we cannot predict the effects of climate change on disease and parasitism based solely on temperature (Lafferty, Porter, and Ford, 2004). Temperature is likely to interact with a variety of other stressors to affect parasitism and disease rates (Lafferty, Porter, and Ford, 2004), including excess nutrients (Harvell *et al.*, 2004), chemical pollutants such as metals and organochlorines (Harvell *et al.*, 2004; Mydlarz, Jones, and Harvell, 2006), and hypoxia (Mydlarz, Jones, and Harvell, 2006). For example, the 2002 die-off of corals and sponges in Florida Bay co-occurred with a red tide (*Karenia brevis)* driven by high nutrient conditions (Harvell *et al.*, 2004). Moreover, not all parasites will respond positively to increased temperature; some may decline (Harvell *et al.*, 2002; Roy, Guesewell, and Harte, 2004) and others may be kept in check by other factors (Harvell *et al.*, 2002; Hall *et al.*, 2006). This suggests that generalizations may not always be possible; idiosyncratic species responses may require that we consider effects on a species-by-species, or place-by-place basis, as with the species distributions discussed earlier.

Such changes in risk of parasitism and disease will influence populations of fish and wildlife, and can affect habitat that is provided by organisms like corals, thereby affecting management.

7.2.4.2.3 Effects of Shoreline Stabilization on Estuaries and their Services

Estuarine shorelines along much of the U.S. coast have been affected by human activities. These activities have exacerbated both water- and land-based stressors on the estuarine land-water interface. Real and perceived threats from global sea level rise, increased intensity of tropical storms, waves from boat wakes, and changes in delivery of and erosion by stream flows have contributed to greater numbers of actions taken to stabilize estuarine shorelines using a variety of techniques. Shoreline stabilization can affect the physical (bathymetry, wave environment, light regime, sediment dynamics) and ecological (habitat, primary production, food web support, filtration capacity) attributes of the land-water interface in estuaries. Collectively, these physical and ecological attributes determine the degree to which ecosystem services are delivered by these systems (Levin *et al.*, 2001). Shoreline stabilization on the estuarine shoreline has only recently begun to receive significant attention (Committee on Mitigating Shore Erosion along Sheltered Coasts, National Research Council, 2006).

Surprisingly little is known about the effects of estuarine shoreline stabilization structures on adjacent habitats (Committee on Mitigating Shore Erosion along Sheltered Coasts, National Research Council, 2006). Marsh communities at similar elevations with and without bulkheads behind them were found to be indistinguishable in a study in Great Bay Estuary in New Hampshire (Bozek and Burdick, 2005). However, this study also reported that bulkheads eliminated the up-slope vegetative transition zone. This loss is relevant for both current function of the marsh and also future ability of the marsh to respond to rising sea level. In several systems within Chesapeake Bay, Seitz and colleagues (2006) identified a link between the hardening of estuarine shorelines with bulkheads or rip-rap and the presence of infaunal prey and predators. This study illustrated the indirect effects that can result from shoreline stabilization, and found them to be on par with some of the obvious direct effects. Loss of ecological function in the estuarine land-water margin as a result of shoreline stabilization is a critical concern. However, the complete loss of the structured habitats (SAV, salt marsh) seaward of shoreline stabilization structures as sea level rises is a more dire threat. In addition, the intertidal sand and mud flats, which provide important foraging grounds for shorebirds and nektonic fishes and crustaceans, will be readily eliminated as sea level rises and bulkheads and other engineered shoreline stabilization structures prevent the landward migration of the shoreline habitats. Absent the ability to migrate landward, even habitats such as marshes, which can induce accretion by organic production and sediment trapping, appear to have reduced opportunity to sustain themselves as water level rises (Titus, 1998).

These effects of shoreline stabilization interacting with sea level rise will influence salt marsh and other intertidal and shallow-water estuarine habitats, with consequences for water quality, fish and wildlife production, and human values, inducing need for management adaptation.

7.2.4.2.4 Effects of Climate Change on Marsh Trapping of Sediments and Geomorphologic Resiliency

Coastal wetlands have been relatively sustained, and even expanded, under historic eustatic sea level rise. Marsh surfaces naturally subside due to soil compaction, other geologic (subsidence) processes, and anthropogenic extraction of fluids such as groundwater and oil. However, marsh surfaces (marsh plain) also build vertically due to the combined effect of surface sediment deposition and subsurface accumulation of live and dead plant roots and decaying plant roots and rhizomes. Both of these processes are controlled by tidal-fluvial hydrology that controls delivery of sediments, nutrients, and organic matter to the marsh as well as the oxygen content of the soil. Local landscape setting (wave energy) and disturbance regime (storm frequency and intensity) are also factors over long term. Thus, the relative sea level (the simultaneous effect of eustatic sea level rise and local marsh subsidence) can be relatively stable under a moderate rate of sea level rise because marsh elevation increases at the same rate as the sea level is rising (*e.g.*, Reed, 1995; Callaway, Nyman, and DeLaune, 1996; Morris *et al.*, 2002). Whether a marsh can maintain this equilibrium with mean sea level and sustain characteristic vegetation and associated attributes and functions is uncertain. It will depend on the interaction of complex factors, including sediment pore space, mineral matter deposition, initial elevation, rate of sea level rise, delivery rates of sediments in stream and tidal flows, and the production rate of below-ground organic matter (U.S. Climate Change Science Program; In Press).

Thus, changes in sediment and nutrient delivery and eustatic sea level rise are likely to be the key factors affecting geomorphic resiliency of coastal wetlands. Sediment delivery may be the critical factor: estuaries and coastal zones that presently have high rates of sediment loading, such as those on the southeast and northwest coasts, may be able to persist up to thresholds of 1.2 cm per year that are optimal for marsh primary production (Morris *et al.*, 2002). If sea level rise exceeds that rate, then marsh surface elevation decreases below the optimum for primary production. However, increased precipitation and storm intensities commensurate with many future climate scenarios (*e.g.*, in the Pacific Northwest) would also likely increase sediment delivery but also erode sediments where flows are intensified. The large-scale responses to changes in sediment delivery to estuarine and coastal marshes have not been effectively addressed by most hydrodynamic models incorporating sediment transport. SAP 4.1 elucidates potential impacts by providing maps depicting the wetland losses in the mid-Atlantic states that are anticipated under various rates of sea level rise (U.S. Climate Change Science Program; In Press). Such changes in sediment and nutrient delivery to the estuary will threaten the geomorphologic resilience of salt marsh habitat, thereby altering water quality and fish and wildlife production; these changes imply the need for management adaptation.

7.2.4.2.5 Effects of Sea Level Rise and Storm Disturbance on Coastal Barrier Deconstruction

Two important consequences of climate change are accelerated sea level rise and increased frequency of high-intensity storms. Sea level rise and intense storms work alone and in combination to alter the hydrogeomorphology of coastal ecosystems and their resultant services. Furthermore, the extent to which they act on ecosystems is dependent on human alterations to these ecosystems. Perhaps the best known example of the current interaction of sea level rise, storm intensity, and human activity is the coast of the Gulf of Mexico around the Mississippi River. Relative sea level rise of the Louisiana coast is one of the highest in the world, in large part as a result of human activities, and this has caused significant losses of wetlands (Boesch, 1994; González and Törnqvist, 2006; Day, Jr. *et al.*, 2007). The consequences of intense storms (*i.e.*, Hurricanes Katrina and Rita) on coastal ecosystems of the Gulf of Mexico, human dominated and natural, are now legend (Kates *et al.*, 2006). New Orleans and other cities were devastated by these storms. Wetland loss was dramatic with sharp alterations to community structure (Turner *et al.*, 2006; U.S. Geological Survey, 2007). Barrier islands were eroded, overwashed, and breached with severe impacts to both human lives and infrastructure. The impacts of these storms are linked to the damaged conditions and decreased area of the wetlands and their historical loss (Day, Jr. *et al.*, 2007). Now reconstruction of New Orleans and other cities has begun and plans are being offered for the replenishment and protection of wetlands and barrier islands (U.S. Army Corps of Engineers, In Press; Day, Jr. *et al.*, 2007; Coastal Protection and Restoration Authority of Louisiana, 2007).

Although the impacts of the hurricanes of 2005 and the influence of relative sea level rise on their impacts were the most costly to the United States, they are not the only examples of how storms and sea level rise influence hydrogeomorphology. Sea level rise and erosion, fostered by storms, have caused estuarine islands to disappear and led to significant changes in shorelines (Hayden *et al.*, 1995; Riggs and Ames, 2003). Barrier island shape and position are dynamic, dependent on these two processes. These processes are natural and have occurred throughout the Holocene; what is relatively new are the ways in which human values are in conflict with these processes and how humans either promote or inhibit them.

Wetlands can maintain themselves in the face of sea level rise by accretion. This accretion is supported by both sedimentation and organic matter accumulation (Chmura *et al.*, 2003). The ability to accrete makes it difficult to assess the true consequences of sea level rise on landscape pattern and resultant area of wetlands, especially over large areas (Titus and Richman, 2001). We do not know exactly the potential accretion and subsidence rates of most wetlands and the thresholds at which relative sea level rise exceeds net elevation change, causing increased inundation and ultimately wetland loss. Based on the experiences of Louisiana, we can estimate that the maximum accretion rate may be less than 10 mm per year, but applicability to other systems is undetermined. Two things are clear: First, the limits depend on the source of material for accretion (*i.e.*, sediment or organic matter) and hence the rates of processes that introduce and remove the materials. Second, the rates of these processes will differ with location both locally within the coastal landscape and regionally due to climate, community, and hydrogeomorphic conditions.

Sea level rise and storm disturbance have not only severe consequences as described, but they are important drivers of the natural progression of coastal ecosystems. One can consider the coastal landscape as having a sequence of ecosystem states, each dependent upon a particular hydroperiod and tidal inundation regime (Brinson, Christian, and Blum, 1995; Hayden *et al.*, 1995; Christian *et al.*, 2000). For example in the mid-Atlantic states, coastal upland, which is rarely flooded, would be replaced by high salt marsh as sea level rises. High marsh is replaced by low marsh, and low marsh is replaced by intertidal flats. While sea level rise alone may effect these changes in state, they are promoted by disturbances that either kill vegetation (*e.g.*, salt intrusion from storms killing trees) or change elevation and hence hydroperiod (*e.g.*, erosion of sediment). It is unclear how accelerated sea level rise and frequency of severe storms will alter the balance of this sequence.

Normally one considers that disturbances would be local, such as salt water intrusion or wrack deposition. But these state changes can actually result from regional impacts of disturbance. For example, *Juncus roemerianus* is a rush species commonly found in high marshes along the coasts of mid-Atlantic, southern Atlantic, and Gulf of Mexico regions of the United States. It is less common where astronomical tidal signals are strong (Woerner and Hackney, 1997; Brinson and Christian, 1999), and it is replaced by *Spartina alterniflora* or perhaps other species. Any disturbance that increases the strength of astronomical tides promotes this shift. Such a disturbance could be the breaching of barrier islands in which increased flow through new inlets may foster more dominant astronomical tides and the ecosystem state change. The predicted disintegration of barrier islands as a consequence of intense storm damage acting from a higher base sea level has catastrophic implications (Riggs and Ames, 2003). Coastal barriers function to protect mainland shorelines from tidal energy, storm surge, and wave forces, such that loss of the protections implies catastrophic inundation, erosion, and loss of wetlands and other coastal habitats on mainland shores as well as back-barrier shores.

Sea level rise and increased frequency of intense storms will influence salt marsh and other wetland habitats by erosion and salt water intrusion, thereby influencing fish and wildlife production, available quantity of fresh water, and provision of human values, with consequences for management.

7.2.4.2.6 Joint Effects of Increasing Temperature and Carbon Dioxide

As a consequence of increasing global temperatures, the limits of climate-adapted habitats are expected to shift longitudinally. Temperate herbaceous species that dominate the tidal wetlands throughout many U.S. estuaries may be replaced by more tropical species such as mangroves (Harris and Cropper Jr., 1992). Salt marshes and mangroves are not interchangeable, despite the fact that both provide structure to support productive ecosystems and perform many of the same ecosystem functions. Mangroves store up to 80% of their biomass in woody tissue whereas salt marshes lose 100% of their aboveground biomass through litterfall each year (Mitsch and Gosselink, 2000). Production of litter facilitates detrital foodwebs and supports many ecological processes in wetlands, so this distinction has implications for materials cycling such as carbon sequestration (Chmura *et al.*, 2003). There are significant differences in structural complexity and biological diversity between these wetland systems. These differences will affect the capacity of the wetlands to assimilate upland runoff, maintain their vertical position and provide flood control. Temperature-driven species redistribution will be further complicated as sea level increases and vegetation is forced landward.

Since pre-industrial times, the atmospheric concentration of carbon dioxide (CO_2) has risen by 35% to 379 ppm in 2005 (IPCC, 2007). Ice cores have proven that this concentration is significantly greater than the natural range over the last 650,000 years (180–300 ppm). In addition, the annual average growth rate in CO_2 concentrations over the last 10 years is larger than the average growth rate since the beginning of continuous direct atmospheric measurements: 1.9 ppm per year average from 1995–2005 compared with 1.4 ppm per year average from 1960 to 2005 (IPCC, 2007). Because CO_2 is required for photosynthesis, these changes may have implications for estuarine vegetation. Plants can be divided based on the way in which they assimilate CO_2. C3 plants include the vast majority of plants on earth (~95%) and C4 plants, which include crop plants and some grasses, comprise most of the rest. Early in the process of CO_2 assimilation, C3 plants form a pair of three carbon molecules whereas C4 plants form four carbon molecules. The distinction between C3 and C4 species at elevated atmospheric CO_2 content is that C3 species increase photosynthesis with higher CO_2 levels and C4 species generally do not (Drake *et al.*, 1995). In wetland systems dominated by C3 plants (*e.g.*, mangroves, many tidal fresh marshes), elevated CO_2 will increase photosynthetic potential and may increase the related delivery of ecosystems services from these systems (Drake *et al.*, 2005). Ongoing research is examining the potential for shifts in wetland community composition driven by elevated CO_2. Data from one of these efforts indicate that despite the advantage afforded to C3 species at higher CO_2 levels, CO_2 increases alone are unlikely to cause black mangrove to replace cordgrass in Louisiana marshes (U.S. Geological Survey, 2006). However, many important estuarine ecosystem effects from elevated CO_2 levels have been documented, including increases in fluxes of CO_2 and methane (Marsh *et al.*, 2005), augmented nitrogen fixation by associated microbial communities (Dakora and Drake, 2000), increased methanogenesis (Dacey, Drake, and Klug, 1994) and changes in the quantity and composition of root material (Curtis *et al.*, 1990).

The joint effects of rising temperature and increased CO_2 concentrations will influence composition and production of shoreline plants that are critical habitat providers and contributors to detrital food chains, thereby also affecting fish and wildlife production and provision of human values and inducing need for management adaptations.

7.2.4.2.7 Effects of Increased CO_2 on Acidification of Estuaries

Ocean acidification is the process of lowering the pH of the oceans by the uptake of CO_2 from the atmosphere. As atmospheric CO_2 increases, more CO_2 is partitioned into the surface layer of the ocean (Feely *et al.*, 2004). Since the industrial revolution began to increase atmospheric CO_2 significantly, the pH of ocean surface waters has deceased by about 0.1 units and it is estimated that it will decrease by another 0.3–0.4 units by 2100 as the atmospheric concentration continues to increase (Caldeira and Wickett, 2003). The resulting decrease in pH will affect all calcifying organisms because as pH decreases, the concentration of carbonate decreases, and when carbonate becomes under-saturated, structures made of calcium carbonate begin to dissolve. However, dissolution of existing biological calcium carbonate structures is only one aspect of the threat of acidification; another threat is that as pH falls and carbonate becomes undersaturated it requires more and more metabolic energy for an organism to deposit calcium carbonate. The present lowered pH is estimated to have reduced the growth of reef-building by about 20% (Raven, 2005). While corals get the most attention regarding acidification, a wide spectrum of ocean and estuarine organisms are affected, including coraline algae; echinoderms such as sea urchins, sand dollars, and starfish; as well as coccolithophores, foraminifera, crustaceans, and molluscan taxa with shells, of which pteropods are particularly important (Orr *et al.*, 2005). The full ecological consequences of the reduction in calcification by marine calcifiers are uncertain, but it is likely that the biological integrity of ocean and estuarine ecosystems will be seriously affected (Kleypas *et al.*, 2006).

Effects of climate change on estuarine acidification will influence water quality, provision of some biogenic habitat like coral reefs, fish and wildlife production, and human values, thus implying need for management adaptation.

7.2.4.2.8 Effects of Climate Change on Hypoxia

Low dissolved oxygen (DO) is a problematic environmental condition observed in many U.S. estuaries (Bricker *et al.*, 1999). Although a natural summer feature in some systems, the frequency and extent of hypoxia have increased in Chesapeake Bay, Long Island Sound, the Neuse River Estuary, and the Gulf of Mexico over the past several decades (Cooper and Brush, 1993; Paerl *et al.*, 1998; Anderson and Taylor, 2001; Rabalais, Turner, and Scavia, 2002; Cooper *et al.*, 2004; Hagy *et al.*, 2004; Scavia, Kelly, and Hagy, 2006). Persistent bottom water hypoxia (*e.g.*, DO concentration < 2.0 mg per L) results from interactions among meteorology and climate, the amounts and temporal patterns of riverine inflows, estuarine circulation, and biogeochemical cycling of allochthonous and autochthonous organic matter (Kemp *et al.*, 1992; Boicourt, 1992; Buzzelli *et al.*, 2002; Conley *et al.*, 2002). Over time, the repeated bottom water hypoxia can alter biogeochemical cycling, trophic transfers, and estuarine production at higher trophic levels (Baird *et al.*, 2004). Ecological and economic consequences of fish kills, bottom habitat degradation, and reduced production at the highest trophic levels in response to low DO have provided significant motivation to understand and manage hypoxia (Tenore, 1970; Officer *et al.*, 1984; Turner, Schroeder, and Wiseman, 1987; Diaz and Rosenberg, 1995; Hagy *et al.*, 2004).

Various scenarios predict that climate change will influence the vulnerability of estuaries to hypoxia through changes in stratification caused by alterations in freshwater runoff, changes in water temperature, increases in sea level, and altered exchanges with the coastal ocean (Peterson *et al.*, 1995; Scavia *et al.*, 2002). Additionally, warmer temperatures should increase metabolism by the water-column and benthic microbial communities, whose activity drives the depletion of DO. Many of the factors that have been found to contribute to the formation of hypoxia (Borsuk *et al.*, 2001; Buzzelli *et al.*, 2002) will be affected by one or more predicted changes in climate (Table 7.3). Because hypoxia affects valued resources, such as fish and wildlife production, reductions in hypoxia are a management target for many estuaries, and adaptations will be required as a consequence of climate change.

7.2.4.2.9 Effects of Changing Freshwater Delivery

Climate change is predicted to affect the quality, rate, magnitude, and timing of the freshwater delivered to estuaries (Alber, 2002), potentially exacerbating existing human modifications of these flows, as described by Sklar and Browder (1998). However, the exact nature of these changes is difficult to predict for a particular estuary, in part because there is not clear agreement among GCMs on precipitation changes over drainage basins (National Assessment Synthesis Team, 2000). There does seem to be agreement among models that increases in frequencies of extreme rainfall will occur (Scavia *et al.*, 2002), suggesting that there will be changes in potential freshwater inflow amounts and patterns (hydrographs). These inflows will then be subjected to human modifications that differ across estuaries. For example, where dams are used in flood regulation, there is reduced variability within and among seasons, damping, for example, normally peak flows at snowmelt in temperate regions (Poff *et al.*, 1997; Alber, 2002). In some watersheds, increased reuse of wastewater in agriculture, municipalities, and industry may offset changes in supply by reducing demand for “clean” freshwater.

The potential physical and chemical consequences of altered freshwater flows to estuaries include changes in salinity and stratification regimes, loadings of nutrients and sediments, water residence times, and tidal importance (reviewed in Alber, 2002). Potential biological consequences include changes in species composition, distribution and abundance, as well as primary and secondary productivity, in response to the altered availability of light, nutrients, and organic matter (Cloern *et al.*, 1983; Howarth *et al.*, 2000; Alber, 2002).

Increases in the delivery of freshwater to estuaries may enhance estuarine circulation and salt wedge penetration up the estuary (Gedney *et al.*, 2006), resulting in stronger vertical stratification. For individual estuaries there is the potential for increased freshwater inflow to shift the degree of mixing along the gradient from the fully mixed toward the stratified state. Those estuaries that receive increased supplies of organic matter and nutrients and exhibit enhanced stratification may be particularly susceptible to enhanced hypoxia and the negative effects described in the previous section. However, at some level, increased freshwater delivery will reduce residence time and thus reduce the potential for hypoxia. This threshold will be specific to individual estuaries and difficult to predict in a generic sense.

In some estuaries, climate change may also lead to a reduction in freshwater inflow that will generally increase salinity. This could lead to more salt-water intrusion upstream, negatively affecting species intolerant of marine conditions (Copeland, 1966; Alber, 2002) and/or lengthening the estuary by extending the distance along the freshwater to full seawater gradient (Alber, 2002). Water residence times within the estuary will likely increase with reduced freshwater inflow, potentially allowing enhanced stratification (both in temperature and salinity) and therefore creating a more stable system in which phytoplankton can grow and reproduce (Cloern *et al.*, 1983; Howarth *et al.*, 2000). Thus, one might expect a greater response to nutrients—*i.e.,* greater primary productivity and/or larger phytoplankton populations (Mallin *et al.*, 1993)—than under baseline rates of freshwater discharge. This may be especially true for estuaries that are currently somewhat "protected" from eutrophication symptoms by high freshwater flow, such as the Hudson River (Howarth *et al.*, 2000). However, reduced flushing times will also keep water in the estuary longer, potentially increasing the risks posed by pollutants and pathogens (Alber and Sheldon, 1999; Sheldon and Alber, 2002).

Other biological consequences of changing freshwater delivery include alterations in secondary productivity (the directions of which are difficult to predict), the distributions of plants and sessile invertebrates (Alber, 2002), and cues for mobile organisms such as fish, especially migratory taxa with complex life histories (Whitfield, 1994; Whitfield, 2005). Not surprisingly, therefore, a whole branch of management is developing around the need to determine the optimal freshwater flows required to maintain desired ecosystem services (*e.g.*, Robins *et al.*, 2005; Rozas *et al.*, 2005).

Changes in freshwater delivery to the estuary will affect freshwater quantity, water quality, stratification, bottom habitats, fish and wildlife production, and human values, inducing needs for management adaptation.

7.2.4.2.10 Phenology Modifications and Match/Mismatch

Estuaries are characterized by high temporal variability on multiple time scales and spatial variability, which includes sharp environmental gradients with distance upstream and vertically in the water column (Remane and Schlieper, 1971). One mode of adaptation that many free-living estuarine species use to exploit the many resources of estuaries is to move in and out of the estuary, as well as upstream and downstream within the estuary, on a complex temporal schedule. A study in North Carolina found that the most abundant fish species in small tributaries of the upper estuary differed in 10 of the 12 months of the year (Kuenzler *et al.*, 1977). Ten different species were dominant during the 12 months of the year. To accomplish such movements many estuarine species have evolved behavior that uses various sensory cues to control the timing of their activities (Sims *et al.*, 2004). The timing of behavior cued by environment information is referred to as "phenology" (Mullins and Marks, 1987; Costello, Sullivan, and Gifford, 2006). The best understood type of phenology that occurs in estuaries involves matching critical feeding stages with the timing of primary productivity blooms (Scavia *et al.*, 2002). As many estuarine stressors are altered by climate change we can expect that phenology will be one of the first biological processes to be seriously disrupted.

Changing phenology has large implications for fish and wildlife production because trophic coupling of important species in the food chain can be disrupted, thereby presenting a need for management adaptation.

7.2.4.2.11 River Discharge and Sea Level Impacts on Anadromous Fishes

Anadromous fishes, such as Pacific salmon, are an important economic and cultural resource that may be particularly vulnerable to significant shifts in coastal climates in the Pacific Northwest and Alaska. The combined effect of shifts in seasonal precipitation, storm events, riverine discharge, and snowmelt (Salathé, 2006; Mote, 2006) are likely to change a broad suite of environmental conditions in coastal wetlands upon which salmon depend at several periods in their life histories. The University of Washington's Climate Impacts Group (UW-CIG) has summarized current climate change in the Pacific Northwest to include region-wide warming of ~0.8ºC in 100 years, increased precipitation, a decline in snowpack, especially at lower elevations, and an earlier spring (Climate Impacts Group, University of Washington, 2007). The UW-CIG predictions for future climate change in the region include an increase in average temperatures on the order of 0.1–0.6ºC (best estimate = 0.3ºC) per decade throughout the coming century, with the warming occurring during all seasons but with the largest increases in the summer. Precipitation is also likely to increase in winter and decrease in summer, but with no net change in annual mean precipitation. As a consequence, the mountain snowpack will diminish and rivers that derive some of their flow from snowmelt will likely demonstrate reduced summer flow, increased winter flow, and earlier peak flow. Lower-elevation rivers that are fed mostly by rain may also experience increased wintertime flow due to increases in winter precipitation. Summer river flows in the Pacific Northwest are estimated to decline by as much as 30% and droughts would become more common (Leung and Qian, 2003), implying significant changes in estuarine salinity distribution that has not yet been examined in any detail. Chapter 6, Wild and Scenic Rivers, provides an expanded discussion of these and other climate change effects on rivers in the United States.

Contemporary estimates of eustatic sea level rise associated with trends in climate change have ranged from 34–50 cm per century (Church, 2001). More recent estimates that have taken into account measurements of continental glacier movement, such as in Greenland, project increased rates from 75–100 cm per century (Meehl *et al.*, 2005) to 2.2–3.4 m by 2100 (Overpeck *et al.*, 2006; Otto-Bliesner *et al.*, 2006). However, relative sea level rise will differ considerably on regional and local scales due to variability in isostatic rebound, local extractions of subsurface fluids like ground water and hydrocarbons, and rapid tectonic events like earthquakes and vulcanism.

Because different anadromous species occupy estuarine wetlands according to their divergent life history strategies, impacts of these climate changes vary between and within species. In the case of Pacific salmon, the “ocean-type” species and life history types would be the most vulnerable because they occupy transitional estuarine waters significantly longer than “stream-type” salmon. For instance, juvenile Chinook and chum salmon representing this “ocean-type” life history strategy may occupy estuarine wetlands for over 90 days (Simenstad, Fresh, and Salo, 1982), seeking (1) refugia from predation at their small size, (2) time to achieve physiological adaptation from freshwater to marine salinities, and (3) high densities of appropriate prey organisms. Based on our knowledge of the habitat requirements and landscape transitions of migrating juvenile ocean-type salmon (Simenstad *et al.*, 2000; Parson *et al.*, 2001; Mote *et al.*, 2003), as sea water penetrates further up the estuary, the present spatial coincidence will change of necessary physical habitats like marsh platforms and tidal creeks with the appropriate salinity regime. This would have potentially large impacts on the ocean-type salmon performance.

In the Pacific Northwest, shifts from snowmelt runoff to more winter storm precipitation will potentially disrupt the migration timing and residence of juvenile salmon in estuarine wetlands. For example, juvenile Chinook salmon in many watersheds migrate to estuaries coincident with the spring freshet of snowmelt and occupy the extensive brackish marshes available to them during that period. This opportunity often diminishes as water temperatures increase and approach physiologically marginal limits (*e.g.*, 19–20ºC) with the decline of snowmelt and flows in early summer. Under the current climate change/variability scenarios, much of the precipitation events will now be focused in the winter, providing less brackish habitat opportunities during the expected juvenile salmon migration and even more limiting temperatures during even lower summer flows. Whether migration and other life history patterns of salmon could adapt to these climate shifts are unknown.

The sustainability of estuarine wetlands under recent sea level rise scenarios is also of concern if estuarine habitat utilization by anadromous fish is density-dependent. Estuaries that are positioned in a physiographic setting allowing transgressive inundation, such as much of the coastal plain of the southeastern and Gulf of Mexico coasts, have a buffer that will potentially allow more inland development of estuarine wetlands. Other coasts, such as those of New England and the Pacific Northwest, have more limited opportunities for transgressive development of estuarine wetlands, and many estuaries are already confined by upland agricultural or urban development that would prevent further inland flooding (Brinson, Christian, and Blum, 1995). For one example, Hood (Hood, 2007) found that a 45-cm sea level rise over the next century would result in a 12% loss, and an 80-cm rise would eliminate 22%, of the tidal marshes in the Skagit River delta (Puget Sound, Washington), which could be translated to an estimated reduction in estuarine rearing capacity for juvenile Chinook salmon of 211,000 to 530,000 fish, respectively. These estimates are based entirely on the direct inundation effects on vegetation and do not incorporate the potential response of existing marshes to compensate for the increased rate of sea level rise, which can include increased sediment accretion and maintenance of marsh plain elevation or increased marsh progradation due to higher sediment loads from the river (see section 7.2.4.2.15 below). Nor do these estimates take into account increased marsh erosion from greater winter storm activity or changes in salinity distribution due to declining summer river flows. Court cases have already overturned general permits for shoreline armoring where salmon (an endangered species under ESA) would be harmed and with projected rises in sea level, the needs of salmon may come even more often into conflict with management policies that generally permit bulkheads and other shoreline armoring to protect private property.

Salmon represent such an iconic fish of great importance to fisheries, wildlife, subsistence uses, and human culture that climate-related impacts on salmon populations would require management adaptation.

7.2.4.2.12 Effects of Climate Change on Estuarine State Changes

The many direct and indirect influences of climate change may combine to cause fundamental shifts in ecosystem structure and functioning. Some shifts, such as those associated with transgression of wetlands, can be considered part of the normal responses to sea-level rise (Brinson, Christian, and Blum, 1995; Christian *et al.*, 2000). Of particular concern is the potential for ecosystems to cross a threshold beyond which there is a rapid transition into a fundamentally different state that is not part of a natural progression. Ecosystems typically do not respond to gradual change in key forcing variables in a smooth, linear fashion. Instead, there are abrupt, discontinuous, non-linear shifts to a new state (or "regime") when a threshold is crossed (Scheffer *et al.*, 2001; Scheffer and Carpenter, 2003; Burkett *et al.*, 2005). Particularly relevant here is the hypothesis that gradual changes in "slow" variables that operate over long time scales can cause threshold-crossing when they alter interactions among "fast" variables whose dynamics happen on short temporal scales (Carpenter, Ludwig, and Brock, 1999; Rinaldi and Scheffer, 2000). We anticipate that some climate changes will fall into this category, such as gradual increases in temperature. The diversity of additional stressors arising from consequences of climate change greatly enhances the likelihood of important stressor interactions. Thus, in estuaries, where so many stressors operate simultaneously, there is great potential for interactions among stressors to drive the system into an alternative state.

Regime shifts can sometimes be catastrophic and surprising (Holling, 1972; Scheffer and Carpenter, 2003; Foley *et al.*, 2005), and reversals of these changes may be difficult, expensive, or even impossible (Carpenter, Ludwig, and Brock, 1999). Moreover, the social and economic effects of discontinuous changes in ecosystem state can be devastating when accompanied by the interruption or cessation of essential ecosystem services (Scheffer *et al.*, 2001; *e.g.*, Foley *et al.*, 2005). Recognizing and understanding the drivers of regime change and the inherent nonlinearities of biological responses to such change is a fundamental challenge to effective ecosystem management in the face of global climate change (Burkett *et al.*, 2005; Groffman *et al.*, 2006).

All the potential regime shifts described below have large implications for sustaining biogenic habitat, provision of fish and wildlife, and many human values, thereby implying need for management adaptation.

7.2.4.2.13 Climate Change Effects on Suspension-Feeding Grazers and Algal Blooms

The Eastern oyster (*Crassostrea virginica*) is a historically dominant species in estuaries along the Atlantic and Gulf of Mexico coasts of the United States. At high abundances, oysters play major roles in the filtration of particles from the water column, biodeposition of materials to the benthos, nutrient cycling, and the creation of hard substrate habitat in otherwise soft-bottom systems (Kennedy, 1996; Coen, Luckenbach, and Breitburg, 1999; Newell and Ott, 1999; Newell, Cornwell, and Owens, 2002). Dominant consumers (*e.g.*, the schyphomedusan sea nettle, *Chrysaora quinquecirrha*) are dependent on oysters for habitat for sessile stages, and large numbers of estuarine fish species benefit either directly or indirectly from habitat and secondary production of oyster reefs (Coen, Luckenbach, and Breitburg, 1999; Breitburg *et al.*, 2000). Oysters are structural as well as biological ecological engineers (Jones, Lawton, and Shachak, 1994), and have been shown to reduce shoreline erosion (Meyer, Townsend, and Thayer, 1997) and facilitate regrowth of submerged aquatic vegetation by reducing nearshore wave action.

Oyster abundances in Atlantic Coast estuaries have declined sharply during the past century, with a precipitous decline in some systems during the past two to three decades. The primary stressors causing the recent decline are likely overfishing and two pathogens: *Haplosporidium nelsoni*—the non-native protist that causes MSX—and *Perkinsus marinus*, a protistan that causes Dermo and is native to the United States but has undergone a recent range expansion and possible increase in virulence (Rothschild *et al.*, 1994; National Research Council, 2004). Both overfishing and disease cause responses in the relatively slow-responding (*i.e.*, years to decades) adult oysters and oyster reefs, making recovery to the oyster-dominant regime quite difficult. High sediment loading (Cooper and Brush, 1993), eutrophication (Boynton *et al.*, 1995), and blooms of ctenophores (Purcell *et al.*, 1991) may further contribute to oyster decline or prevent recovery to the high-oyster state. These factors—all of which are likely to increase with changes in climate—appear to act most strongly on the larval and newly settled juvenile stages, raising the possibility that this system will at best exhibit hysteretic recovery to the high-oyster state.

7.2.4.2.14 N-Driven Shift from Vascular Plants to Planktonic Micro- and Benthic Macroalgae

Seagrasses are believed to be in the midst of a global crisis in which human activities are leading to large scale losses (Orth *et al.*, 2006). Human and natural impacts have had demonstrable detrimental effects on SAV (Short and Wyllie-Echeverria, 1996). Enhanced loading of nutrients to coastal waters has been found to alter primary producer communities through shifts toward species with faster growth nutrient uptake rates (Duarte, 1991). The shift is often toward phytoplankton, which reduces light availability and can lead to losses of other benthic primary producers such as seagrasses. The disappearance of seagrass below critical light levels is dramatic (Duarte, 1991), and has been linked to nutrient loading in some systems (Short and Burdick, 1996). In Waquoit Bay, Massachusetts, replacement of SAV by macroalgae has also been observed and was primarily attributed to shading (Hauxwell *et al.*, 2001). Increases in macroalgal biomass, macroalgal canopy height and decreases in SAV biomass were linked to N loading rate using a space-for-time substitution (Hauxwell *et al.*, 2001). It is essential to understand the potential for thresholds in water quality parameters that may lead to loss of SAV through a state change. SAV is sensitive to environmental change and thus may serve as “coastal canaries,” providing an early warning of deteriorating conditions (Orth *et al.*, 2006). SAV also provides significant ecological services (Williams and Heck Jr., 2001) and its loss would have appreciable effects on overall estuarine function.

7.2.4.2.15 Non-linear Marsh Accretion with Sea Level Rise

Coastal inundation is projected to lead to land loss and expansion of the sub-tidal regions along estuarine shorelines (Riggs, 2002). Intertidal habitats that do not accrete or migrate landward proportionally to relative sea level rise are susceptible to inundation. Wetlands are often present in these areas and have shown the ability to keep up with increases in sea level in some systems (Morris *et al.*, 2002). However, the ability to maintain their vertical position is uncertain, and depends on a suite of factors (Moorhead and Brinson, 1995). Recent work in the Venice Lagoon found a bimodal distribution of marsh (higher elevation) and flat (lower elevation) intertidal habitats, with few habitats at intermediate intertidal elevations (Fagherazzi *et al.*, 2006). The findings indicate that there may be an abrupt transition from one habitat type to another. Should this model hold true for a broad range of coastal systems, there are clearly significant implications for coastal geomorphology and the ecological services provided by the different habitat types.

7.3 Adapting to Climate Change

Biologists have traditionally used the term “adaptation” to apply to intrinsic biological responses to physical or biological changes that may serve to perpetuate the species, community, or ecosystem. This definition includes behavioral, physiological, and evolutionary adaptation of species. This question therefore arises: Can biological adaptation be relied upon to sustain ecosystem services from national estuaries under conditions of present and future climate change? In the short term of one or two decades, the capability of estuarine organisms to migrate further toward the poles in response to warming temperatures and further up the shore in response to rising water levels has potential to maintain estuarine ecosystem processes and functioning that do not differ greatly from today’s conditions. However, over longer time frames of perhaps 20 or 30 years or more, depending on the magnitude of climate changes, estuarine ecosystems may not be able to adapt biologically and thereby retain high similarity to present systems. The scope and pace of current and anticipated future climate change are too great to assume that management goals will be sustained by intrinsic biological adjustments without also requiring management adaptation (Parmesan and Galbraith, 2004; Parmesan, 2006; Pielke *et al.*, 2007).

The extremely high natural variability of estuarine environments has already selected for organisms, communities, and ecosystems with high capacity for natural physiological, behavioral, and perhaps also evolutionary adaptation (Remane and Schlieper, 1971; Wolfe, 1986). Nevertheless, the present rates of change in many variables like temperature and the absolute levels of key environmental variables like CO_2 concentration that may ultimately be reached could fall outside the historical evolutionary experience of estuarine organisms. The historical experience with environmental variability may not help much to achieve biological adaptation. While behavioral (*e.g.*, migration) adaptation of individual species may take place to some degree, the dramatic suite of projected changes in estuarine environments and stressors that we summarized earlier poses complex challenges to individual species, even those of estuaries, on a timetable that is inconsistent with the capacity for evolutionary change to keep up (Pielke *et al.*, 2007). Even if evolutionary change could proceed at a rapid pace, the diversity of environmental changes implies that conflicting demands may be placed on selection such that adaptation to all change may be compromised. The success of individual species in adapting to climate change does not lead to intrinsic resilience at the community and ecosystems levels of organization. Because virtually all ecosystem processes involve some form of interaction between or among species, biological adaptation by individual species to the climate-driven changes is not a process that will protect functioning estuarine ecosystems because species adapt and migrate at differing rates (Sims *et al.*, 2004; Parmesan, 2006).

Among the most important species of the estuary that dictate overall community composition and ecosystem dynamics are the structural foundation species, namely intertidal marsh plant and subtidal seagrass (SAV) vegetation. Donnelly and Bertness (2001) have assembled ecological evidence that, starting in the late 1990s, the low marsh plant *Spartina alterniflora* has begun to move upslope and invade the higher marsh of New England that are typically occupied by a more diverse mix of *Juncus gerardi*, *Distichlis spicata*, and *Spartina patens*. Their paleontological assessment revealed that in times of rapid sea level rise in the late 19th and early 20th centuries *Spartina alterniflora* similarly grew upwards and dominated the high marsh. Such replacement of species and structural diversity of foundation species is likely to modify the functioning of the salt marsh ecosystem and affect its capacity to deliver traditional goods and services. Similarly, among SAV species, some like *Halodule wrightii* are known to be better colonizers with greater ability to colonize and spread into disturbed patches than other seagrasses like *Thalassia testudinum* (Stephan, Peuser, and Fonseca, 2001). In general, seagrasses that recolonize by seed set can move into newly opened areas more readily than those that largely employ vegetative spread. Analogous to the marsh changes, if storm disturbance and rising water levels favor more opportunistic seagrass species, then the new SAV community may differ from the present one and provide different ecosystem services. Vascular plants of both intertidal and shallow subtidal estuaries possess characteristically few species relative to terrestrial habitats (Day, Jr. *et al.*, 1989; Orth *et al.*, 2006), so these differences in behavior of important foundation species in the marsh and in SAV beds will have disproportionately large influences on function. Thus, the web of interactions among biotic and abiotic components of the estuarine ecosystem cannot be expected to be preserved through intrinsic biological adaptation alone, which cannot regulate the physical changes. Management adaptations must be considered to sustain ecosystem services of national estuaries. Examples of specific adaptation options are presented in Box 7.4 and elaborated further throughout the sections that follow.

7.3.1 Potential for Adjustment of Traditional Management Approaches to Achieve Adaptation to Climate Change

Three different time frames of management adaptation can be distinguished: (1) avoidance of any advance adaptation strategy (leading to *ad hoc* reactive responses); (2) only planning for management responses to climate change and its consequences (leading to coordinated, planned responses initiated either after indicators reveal the urgency or after emergence of impacts); and (3) taking proactive measures to preserve valuable services in anticipation of consequences of climate change. Rational grounds for choosing among these three options involve consideration of the risks and reversibility of predicted negative consequences and the costs of planning and acting now as opposed to employing retroactive measures. Political impediments and lack of effective governance structures may lead to inaction even if planning for intervention or initiating proactive intervention represents the optimal strategy. For example, the partitioning of authority for environmental and natural resource management in the United States among multiple federal and state agencies inhibits effective implementation of ecosystem-based management of our estuarine and ocean resources (Peterson and Estes, 2001; Pew Center on Global Climate Change, 2003; U.S. Commission on Ocean Policy, 2004; Titus, 2004).

Planning for adaptation to climate change without immediate implementation may represent the most prudent response to uncertainty over timing and/or intensity of negative consequences of global change on estuarine ecosystem services, provided that advance actions are not required to avoid irreversible damage. Issues of costs also deserve attention in deciding whether to delay management actions. An ounce of prevention may be worth a pound of cure. For example, by postponing repairs and vertical extensions of levees around New Orleans, the estimated costs for retroactive repair and all necessary restorations of about $54 billion following Hurricanes Katrina and Rita greatly exceed what proactive levee reconstruction would have cost (Kates *et al.*, 2006). On the other hand, the protections provided against natural disasters are typically designed to handle more frequent events, such as storms and floods occurring more frequently than once a century, but inadequate to defend against major disasters like the direct hit by a category 5 hurricane. Such management protections even enhance losses and restoration costs by promoting development under the sense of short-term security (Kates *et al.*, 2006). This example has direct relevance to adaptation management in the estuary because there is broad consensus that climate change is increasing sea levels and increasing frequency of intense hurricanes (IPCC, 2007). Engineered dikes for estuarine shorelines may represent one possible management adaptation, protective of some human values but injurious to natural resources. Thus, the need for understanding the effectiveness and consequences of alternative management policies relating to dikes, levees, and other such structural defenses makes the New Orleans experience relevant.

A decision to postpone implementation of adaptation actions may rely on continuing scientific monitoring of reliable indicators and modeling. Based on inputs from evolving ocean observing systems, model predictions could provide comfort that necessary actions, although delayed, may still be timely. Other important prospective management actions may be postponed because they are not politically feasible until an event alters public opinion sufficiently to allow their implementation. Such adaptations are best planned in advance to anticipate the moment when they could be successfully triggered. Other management actions may involve responding to events and therefore only have relevance in a retrospective context. Catastrophic events provide opportunities for changes that increase ecological and human community resilience, by addressing long-standing problems such as overbuilding in floodplains or degradation of coastal wetlands (Box 7.5) (H. John Heinz III Center for Science, Economics, and the Environment, 2002). However, pressures to expediently restore conditions to their familiar pre-disaster state often lead to the loss of these opportunities (Mileti, 1999). Therefore, decisions about whether and where to rebuild after damage from major floods and storms should be carefully examined and planned in advance in order to avoid making poorer judgments during chaotic conditions that follow these types of incidents. This strategy becomes more valuable as flood damages increase.

Proactive intervention in anticipation of consequences of climate change represents rational management under several conditions. These conditions include irreversibility of undesirable ecosystem changes, substantially higher costs to repair damages than to prevent them, risk of losing important and significant ecosystem services, and high levels of scientific certainty about the anticipated change and its ecological consequences (Titus, 1998; 2000). Avoiding dramatic structural ("phase") shifts in estuarine ecosystem state may represent a compelling motivation for proactive management because such shifts threaten continuing delivery of many traditional ecosystem services and are typically difficult or exceedingly expensive to reverse (Groffman *et al.*, 2006). Reversibility is especially at issue in cases of potential transitioning to an alternative stable state because positive feedbacks maintain the new state and resist reversal (Petraitis and Dudgeon, 2004). For example, the loss of SAV increases the near-bottom currents because of loss of a baffle to flow, such that seagrass seeds are less likely to be deposited and seedlings more likely to be eroded; this feedback makes reestablishment of lost beds much more difficult. With adequate knowledge of the critical tipping point and ongoing monitoring of telling indicators, proactive intervention could in some cases be postponed and still be completed in time to prevent climate change from pushing the system over the threshold into a new phase. Nevertheless, many processes involved in ecosystem change possess substantial inertia such that even after adjusting levels of drivers, a memory of past stress will continue to modify the system, making postponement of action inadvisable. Climate change itself falls into this class of processes in that if greenhouse gas emissions were capped today, the Earth would continue to warm for decades (IPCC, 2007).

Financial costs of climate change may be minimized by some types of proactive management. For example, enacting legislation that prohibits bulkheads and other engineered structures and requires rolling easements could preserve or at least delay loss of important shallow-water habitats, such as salt marsh, by allowing them to migrate inland as sea level rises (Box 7.6) (Titus, 1998). Such laws to require rolling easements is not likely to be ruled a taking, especially if enacted before the property is developed because "the law of erosion has long held that the public tidelands migrate inland as sea level rises, legislation saying that this law will apply in the future takes nothing" (Titus, 1998). However, absent such a law and this interpretation of it, the costs of loss of habitat and associated ecosystem services may exceed the value of property losses that would occur if property owners could not protect their investment. Some other proactive steps that enhance adaptation to climate change are likely to cost very little and deserve immediate inclusion in policy and management plans. For example, the simple incorporation of climate change consequences in management plans for natural and environmental resources will trigger inclusion of forward-looking modifications that might provide resistance to climate change, build resiliency of ecological and socio-economic systems and avoid interventions incompatible with anticipated change and sustained ecosystem services (Titus, 2000). Principles for environmental planning could be adopted that (1) prohibit actions that will exacerbate negative consequences of climate change, (2) allow actions that are climate-change neutral, and (3) promote actions that provide enhanced ecosystem resilience to climate change. Such principles may lead to many low-cost modifications of existing management plans that could be initiated today.

The scientific basis for predicting climate change and its ecosystem consequences must be especially compelling to justify any costly decisions to take proactive steps to enhance adaptation to climate change. Willingness to take costly actions should vary with the magnitude of predicted consequences, the uncertainty associated with the predictions, and the timing of the effects. The scientific basis for the predictions must also be transparent, honest, and effectively communicated not just to managers but also to the general public who ultimately must support adaptation interventions. Thus, there is an urgent need to continue to refine the scientific research on climate change and its ecosystem consequences to reduce uncertainty over all processes that contribute to climate change and sea level rise so that future projections and GCM (General Circulation Models) scenarios are more complete and more precise. Because of the tremendous publicity associated with the release of each IPCC report, this process of periodic re-evaluation of the science and publication of the consensus report plays an integral role in public education. Scientific uncertainty about the magnitudes and timetables of potentially important processes, such as melting of the Greenland ice sheet (Dowdeswell, 2006; Rignot and Kanagaratnam, 2006), leads to their exclusion from IPCC projections. Further scientific research will allow inclusion of such now uncertain contributions to change.

7.3.2 Management Adaptations to Sustain Estuarine Services

7.3.2.1 Protecting Water Quality

All national estuaries, and estuaries more generally, include water quality as a priority management target. The federal Clean Water Act serves to identify explicit targets for estuarine water quality nationwide, but state and local programs can also include other numeric standards for explicit parameters. Some CCMPs specify explicit, sometimes numeric, targets for specific member estuaries. Parameters that possess federally mandated standards include chlorophyll concentration; turbidity; dissolved oxygen; fecal coliform bacteria; nutrient loading where TMDLs apply; and conditions for NPDES discharge permits that maintain balanced and indigenous communities of fish, shellfish, and wildlife. In addition, coastal marsh and other riparian wetland buffers serve to treat non-point-source storm waters before they enter the open waters of estuaries, so preserving marsh extent and functionality is an important management target relating to water quality (Mitsch and Day Jr, 2006).

Perhaps the greatest threat to estuarine water quality from climate change derives from the loss of water treatment of diffuse nutrient pollution by constricted tidal marsh and wetland buffers (Box 7.7). These vegetated buffers are threatened by the joint effects of sea level rise and increasingly intense storms interacting with hardening of estuarine shorelines through installation of bulkheads, dikes, and other engineered structures (Titus, 1998). Such structures are now readily permitted along estuarine shorelines to protect private property and public infrastructure from shoreline erosion; however, by preventing orderly retreat of intertidal and shallow subtidal habitats shoreward as sea level rises (Schwimmer and Pizzuto, 2000), marsh will be lost and its functions eliminated over extensive portions of estuarine shorelines (Titus, 2000; Reed, 2002; Committee on Mitigating Shore Erosion along Sheltered Coasts, National Research Council, 2006). The loss of salt marsh on coastal barriers is further facilitated by beach nourishment, which prevents natural processes of coastal barrier recession through overwash. Overwash of sediments to the estuarine shoreline is a process that extends and revitalizes salt marsh on the protected side of coastal barriers.

Estuarine shorelines differ in their susceptibility to erosion and recession under rising sea levels (U.S. Environmental Protection Agency, 1989) . Relative sea level is rising at very different rates around the country and the globe. The subsiding shores of the Louisiana Gulf Coast represent the region of the country where the salt marsh loss induced by sea level rise is greatest (U.S. Environmental Protection Agency, 1989). These marsh losses on the Mississippi River Delta are enhanced by modification of river flows in ways that inhibit sediment delivery to the marshes and by extraction of subsurface fluids (oil and gas). Extraction of groundwater from shallow aquifers also induces subsidence and enhances relative sea level rise along the shores of some estuaries like San Francisco Bay. For many estuaries, salt marsh does not currently face increased flooding and erosion from rising sea levels, either because relative sea level is not rising rapidly in these regions or because the accumulation of organic peat and trapping and deposition of largely inorganic sediments by emergent marsh plants is elevating the land surface at a rate sufficient to keep up with sea level rise (Reed, 2002). Despite the capability of salt marsh to rise with sea level, this gradual process produces a marsh on an elevated platform where the estuarine shore is increasingly more steeply sloped. The consequently deeper water does not dissipate wave energy as readily as the previously shallow slope, leading to increased risk of shoreline and marsh erosion at the margin (Committee on Mitigating Shore Erosion along Sheltered Coasts, National Research Council, 2006). Therefore, even marsh shores that today are maintaining elevation and position as sea level rises are at risk of greater erosion at their seaward margin in the future. Nevertheless, substantial geographic variation exists in erosion risk and susceptibility to marsh loss (U.S. Environmental Protection Agency, 1989).

Maintaining present management policy allowing bulkheads will likely lead to a scenario of ultimate loss of marsh and a walled estuary wherever development exists on the shoreline. Only on undeveloped estuarine shorelines can marshes recede landward, but with such dramatic expansion of coastal human communities, little undeveloped estuarine shoreline is likely to remain except in public parks, reserves, and sanctuaries. Along estuarine salinity gradients, much more development takes place toward the ocean end and less up-estuary. Therefore, as sea level rises, an increasing fraction of remaining marsh habitat will be found along these undefended, up-estuary shores (see maps in SAP 4.1; U.S. Climate Change Science Program, 2007). All specific water quality parameters for which standards exist will suffer under this scenario of current management without adaptation, and reactive management holds little promise of reversal of impacts. Reactive marsh restoration would require removals of at least some portion of the engineered walls protecting estuarine shoreline property so as to allow flooding of the proper elevations supporting salt marsh restoration. Implementing any public policy that would lead directly to widespread private property loss represents a large challenge under the prevailing property rights laws, but one that should be decided in favor or retaining the estuarine habitats if done in a way that involves rolling easements to preserve the public tidelands (Titus, 1998).

The process of retreat achieved by rolling easements or by some other administrative construct has been discussed in the U.S. for at least two decades. Retreat has an advantage over establishment of fixed buffer zones because the abandonment need not be anticipated and shoreline use modified until sea level has risen enough to require action (Titus, 1998). An analogous proactive response to global climate change and sea level rise is being actively considered in the United Kingdom and European Union and is known as “managed alignment” (Department for Environment, Food and Rural Affairs (DEFRA) and Department of the Environment, 2002). Managed alignment refers to deliberately realigning engineering structures affecting rivers, estuaries, and the coastline. The process could involve retreating to higher ground, constructing set-back levees, shortening the length of levees and seawalls, reducing levee heights, and widening a river flood plain. The goals of managed realignment may be to:

(1) reduce engineering costs by shortening the overall length of levees and seawalls that require maintenance;
(2) increase the efficiency and long-term sustainability of flood and coastal levees by recreating river, estuary, or coastal wetlands and using their flood and storm buffering capacity;
(3) provide other environmental benefits through re-creation of natural wetlands; or
(4) construct replacement coastal wetlands in or adjacent to a designated European site to compensate for wetland losses resulting from reclamation or coastal squeeze.

Under this UK/EU perspective, the goods and services provided by wetland coastal defenses against sea level rise appear to outweigh anticipated costs under some scenarios.

Locally in the U.S., proactive management to protect tidal marshes, on which water quality of estuaries so strongly depends, may have some notable success in the short term, although prospects of longer-term success are less promising. Only Maine, Rhode Island, and Massachusetts have regulations in place that recognize the need to allow wetlands the capacity to migrate inland as sea level rises and thereby provide long-term protection (Titus, 2000). An alternative to bulkheading is using natural breakwaters of oysters in quiescent waters of Atlantic and Gulf Coast estuaries to dissipate wave action and thus help inhibit shoreline and marsh erosion inshore of the reef. Rock sills can be installed in front of tidal marshes along more energetic estuarine shores where oysters would not survive (Committee on Mitigating Shore Erosion along Sheltered Coasts, National Research Council, 2006). Such natural and artificial breakwaters can induce sediment deposition behind them and thereby can help sediments rise and marshes persist with growing sea levels. As sea level rises, oyster reefs can also grow taller and rock sills can be elevated, thereby keeping up protection by the breakwaters. Oysters are active suspension feeders and help reduce turbidity of estuarine waters. Rock breakwaters in the estuary are also often colonized by oysters and other suspension feeding invertebrates. Restoration of oyster reefs as breakwaters and even installation of rock breakwaters contribute to water quality through the oysters’ feeding and through protection of salt marshes by these alternatives to bulkheads and dikes. This proactive adaptation to sea level rise and risk of damaging storms will probably fail to be sustainable over longer time frames because such breakwaters are not likely to provide reliable protection against shoreline erosion in major storms as sea level continues to rise. Ultimately, the owners of valuable estuarine shoreline may not be satisfied with breakwaters as their only defense against the rising waters and may demand permission to install levees, bulkheads, or alternative forms of shoreline armoring. This could lead to erosion of all intertidal habitats along the shoreline and consequent loss of the tidal marsh in developed areas. Some of these losses of marsh acreage would be replaced by progressive drowning of river mouths and inundation of flood plains up-estuary as sea level rises. The most promising suite of management adaptations on those highly developed shorelines down-estuary is likely a combination of rolling easements, setbacks, density restrictions, and building codes (Titus, 1998). Political resistance may preclude local implementation of this adaptation, but financial costs of implementation are reasonable, if done before the shoreline is developed (Titus, 2000).

Given the political barriers to implementing these management adaptations to protect coastal wetlands, globally instituted mitigation of climate change may be the only means in the longer term of several decades to centuries of avoiding large losses of tidal marsh and its water treatment functions. Losses will be nearly total along estuarine shorelines where development is most intense, especially in the zone of high hurricane risk from Texas to New York (see SAP 4.1; U.S. Climate Change Science Program, 2007). Although rapid global capping of greenhouse gas emissions would still result in decades of rising global temperatures and consequent physical climatic changes (IPCC, 2007), it may be possible in the short term (years to a few decades) to partially alleviate damage to tidal marshes and diminution of their water treatment role on developed shores by local management adaptations, such as installation of natural and artificial breakwaters. On undeveloped estuarine shorelines, implementation of rolling easements is a critical need before development renders this approach too politically and financially costly. However, much public education will be necessary for this management adaptation to be accepted.

Estuarine water quality is also threatened by a combination of rising temperature, increased pulsing and, in many regions like the east coast, growing quantities of freshwater riverine discharge, and more energetic upstream wedging of sea waters with rising sea level (Scavia *et al.*, 2002). Temperature increases drive faster biochemical rates, including greater rates of microbial decomposition and animal metabolism, which inflate oxygen demand. When increased fresh water discharges into the estuary, this less-dense fresh water at the surface, when combined with stronger salt water wedging on the bottom, will enhance water column stability because of greater density stratification. Such conditions are the physical precursor to development of estuarine bottom water hypoxia and anoxia in warm seasons because oxygen-rich surface waters are too light to be readily mixed to depth (Paerl *et al.*, 1998). This water quality problem leads to persistent hypoxia and anoxia, creating dead zones on the bottoms of estuaries, one of the most serious symptoms of eutrophication (Paerl *et al.*, 1998; Bricker *et al.*, 1999). Under higher water temperatures and extended warm seasons, high oxygen demand is likely to extend for longer periods of the year while greater stratification further decreases dissolved oxygen in bottom waters. Erosion of riparian marshes from rising water levels also adds previously sequestered organic carbon to the estuary, further increasing oxygen demand for its microbial decomposition. In regions such as the Pacific Northwest, where summertime droughts are predicted rather than summer increases in storm-driven pulses of rain, this scenario of greater water-column stability and higher oxygen demand at elevated temperature will not apply. Nevertheless, negative consequences of summertime drought are likely also.

Failing to act in advance of increases in incidence, scope, and duration of bottom water hypoxia implies widespread climate-related modifications of many estuaries, inconsistent with maintaining a balanced indigenous population of fish, shellfish, and wildlife. Nutrient reduction in the watershed and airshed could limit algal blooms and thereby reduce organic loading and oxygen demand (Conley *et al.*, 2002). However, discharge limits for point sources are already close to what is technically feasible in many rivers. From an economic standpoint, further limiting atmospheric nitrogen deposition would affect many activities, such as electric power generation, industrial operations, and automobile use. It is possible that wetland restoration over the drainage basin could be greatly enhanced to reduce the fraction of diffuse nutrient loading that reaches the estuary and help counteract the increased estuarine stratification and warming temperatures that drive higher microbial decomposition and oxygen demand (Mitsch and Day Jr, 2006). Thus, integrated management of nutrient sources and wetland treatment of nutrients can play a role in management to limit eutrophication and hypoxia.

At state levels of management, recognition of the likelihood of climate change and anticipation of its consequences could lead to important proactive steps, some with potentially minimal costs. Regulatory change represents one major example of an institutional approach at this level. Maine, Rhode Island, and Massachusetts deserve praise for appropriately responding to risk of wetland loss under sea level rise by instituting regulations to allow landward migration of these habitats (Titus, 2000). Examination of state laws, agency rules, and various management documents in North Carolina, on the other hand, suggests that climate change is rarely mentioned and almost never considered. One example of how rule changes could provide proactive protection of water quality would be to anticipate changes in sea level rise and storm intensity by modifying riparian buffer zones accordingly to maintain water quality. Permitting rules that constrain locations for construction of landfills, hazardous waste dumps, mine tailings, and facilities that store toxic chemicals could be modified to insure that even under anticipated future conditions of sea level rise, shoreline recession, and intense storms, these facilities would remain not only outside today’s floodplains but also outside the likely floodplains of the future. Riverine floodplain maps and publicly run flood insurance coverage could be redrafted to reflect expectations of flooding frequency and extent under changing rainfall amounts and increasing flashiness of rainfall as it is delivered in more intense discrete storms. Such changes in floodplain maps would have numerous cascading impacts on development activities along the river edges in the entire watershed, many of which would help protect water quality during floods. Water quality degradation associated with consequences of floods from major storms like hurricanes can persist for many months in estuaries (Paerl and Bales, 2001). Thus, if climate change leads to increases in storm intensity, proactive protection of riparian floodplains could help reduce the levels of pollutants that are delivered during those floods. Acting now to address this stressor helps enhance ecosystem resiliency to impacts of climate change on eutrophication and pollution by toxicants. Floodplains may offer some of the last remaining undeveloped components of our coastal landscape over which transgressive expansion of sea level might occur with minimal human impact, so expanding protected areas of floodplains also helps build resilience of the socioeconomic system. Even during the past two decades, many estuarine watersheds have experienced multiple storms that exceeded standards for “100-year floods,” implying that recomputation and remapping of those hazardous riverine floodplains is already necessary.

7.3.2.2 Sustaining Fisheries and Wildlife Populations

Sustaining fish production and wildlife populations represent important management goals of most national estuaries and essentially all estuaries nationwide. Fisheries are likely to suffer large declines from both of the major processes that affect water quality: (1) loss of tidal marshes associated with rising sea levels and enhanced incidence of intense storms as these drivers interact with hardened shorelines; and (2) increased frequency, scope, and duration of bottom-water hypoxia arising from stronger stratification of the estuarine water column and greater microbial oxygen demand at higher temperatures. Marshes and other wetlands perform many valuable ecosystem services (Millennium Ecosystem Assessment, 2005), several of which lead to enhanced fish production. Numerous studies have demonstrated the high use of salt marshes by killifish, grass shrimps, and crabs, which are important prey for larger commercially important fishes and for wading birds at higher trophic levels. Salt marsh habitat supports several endemic species of birds, such as some rails, and small mammals, some of which are on federal or state threatened and endangered lists (Greenberg *et al.*, 2006). The combination of high primary production and structural protection makes the marsh significant as a contributor to important detrital-based food webs based on export of vascular plant detritus from the marsh, and also means that the marsh plays a valuable role as nursery habitat for small fishes and crustaceans (Peterson *et al.*, In Press). Zimmerman, Minello, and Rozas (2000) demonstrated that penaeid shrimp production in bays along the Gulf of Mexico varies directly with the surface area of the salt marsh within the bay. Maintaining complexity of salt marsh landscapes can also be an important determinant of fish, shellfish, and wildlife production, especially preserving marsh edge environments (*e.g.*, Peterson and Turner, 1994). Thus, marsh loss and modification in estuaries are expected to translate directly into lost production of fish and wildlife.

The climate-driven enhancement of bottom water hypoxia and anoxia will result in further killing of oysters and other sessile bottom invertebrates (Lenihan and Peterson, 1998), thereby affecting the oyster fishery directly and other fisheries for crabs, shrimp, and demersal fishes indirectly (Lenihan *et al.*, 2001). These demersal consumers prey upon the benthic invertebrates of the estuary during their nursery use of the system in the warm season of the year. When the benthic invertebrates are killed by lack of oxygen and resulting deadly hydrogen sulfide, fish production declines as energy produced by phytoplankton enters microbial loops and is thereby diverted from passing up the food chain to higher tropic levels (Baird *et al.*, 2004). This enhanced diversion of energy away from pathways leading to higher trophic levels will not only affect demersal fish production but also diminish populations of sea birds and marine mammals, such as bottle-nosed dolphins and killer whales. Because estuaries contribute so greatly to production of coastal fisheries generally, such reductions in fish and wildlife transcend the boundaries of the estuary itself.

Fish and wildlife suffer additional risks from climate change beyond those associated with loss of marsh and other shoreline habitats and those associated with enhanced hypoxia. Higher temperatures are already having and will likely have additional direct effects on estuarine species. Increased temperature is associated with lower bioenergetic efficiency and greater risk of disease and parasitism. As temperatures increase, species will not move at equal rates pole-ward (Parmesan, 2006), so new combinations will emerge with likely community reorganization, elevating abundances of some fishes and crustaceans while suppressing others. Locally novel native species will appear through natural range expansion as water warms, adding to the potential for community reorganization. In addition, introductions of non-native species may occur at faster rates because disturbed communities appear more susceptible to invasion. Finally, the changes in riverine flows—both amounts and temporal patterns—may change estuarine physical circulation in ways that affect transport of larval and juvenile life stages, altering recruitment of fish and valuable invertebrates.

The challenges of adapting management to address impacts of climate change on fish and wildlife thus include all those already presented for water quality, because the goals of preventing loss of tidal marsh and other shallow shoreline habitats and of avoiding expansion of hypoxic bottom areas are held in common. However, additional approaches may be available or necessary to respond to risks of declines in fish and wildlife. For example, fisheries management at federal and state levels is committed to the principle of sustainability, which is usually defined as maintaining harvest levels at some fixed amount or within some fixed range. With climate-driven changes in estuarine ecosystems, sustainable fisheries management will itself need to become an adaptive process as changes in estuarine carrying capacity for target stocks occur through direct responses to warming and other physical factors and indirect responses to changes in biotic interactions. Independent of any fishing impacts, there will be a moving target for many fish, shellfish, and wildlife populations, needing adaptive definitions of what is sustainable. This goal calls for advance planning for management responses to climate change but not implementation until the ecosystem changes have begun. Absent any advance planning, stasis of management could conceivably induce stock collapses by inadvertent overfishing of a stock in decline from climate modifications.

Extermination of injurious non-native species after their introduction into estuarine systems has not proved feasible. However, one proactive type of management adaptation in contemplation of possible enhancement of success of introduced species into climate-disrupted estuarine ecosystems may be to strengthen rules that prevent the introductions themselves. This action would be especially timely as applied to the aquarium fish trade, which is now a likely vector of non-native fish introductions (*e.g.*, National Ocean Service, 2005). Local removals of invasive non-natives combined with restoration of the native species may be a locally viable reactive management response to improve marsh characteristics that promote propagation and production of fish and wildlife. This type of action may best be applied to vascular plants of the salt marsh. Such actions taken now to reduce impacts of current stressors represent means of enhancing ecosystem resilience to impacts of climate change on fish and wildlife.

7.3.2.3 Preserving Habitat Extent and Functionality

All national estuaries and managers of estuarine assets nationwide identify preservation of habitat as a fundamental management goal. The greatest threat to estuarine habitat extent and function from climate change arises as sea level rise and enhanced incidence of intense storms interacts with the presence of structural defenses against shoreline erosion. As explained in our description of threats to water quality and fisheries, barriers that prevent horizontal migration of tidal marshes inland will result in loss of tidal marsh and other intertidal and then shallow subtidal habitats. This process will include losses to seagrass beds and other submerged aquatic vegetation down-shore of bulkheads, because if the grass cannot migrate up-slope, the lower margin will die back from light limitation (Dennison *et al.*, 1993; Short and Wyllie-Echeverria, 1996) as water levels rise. The presence of bulkheads enhances the rate of erosion below them because wave energy is directed downwards after striking a hard wall, excavating and lowering the sediment elevation faster than if no bulkhead were present (Tait and Griggs, 1990). As shoreline erosion below bulkheads continues along with rising water levels, all presently intertidal habitat will become covered by water even at low tide, removing those habitats that are most productive, critical for sustaining fish and wildlife, and important to maintaining water quality. Galbraith *et al*. (2002) modeled this process for installation of dikes on Galveston Island and concluded that intertidal habitat for shorebirds would decline by 20%. The enhancement of bottom water hypoxia through induction of more intense water column stratification and greater microbial degradation rates at higher temperatures will not eliminate the deeper subtidal habitat of estuaries but will degrade its functions over wider areas of “dead zones” of the nation’s estuaries as climate change proceeds.

The challenges of adapting management to address impacts of climate change on estuarine habitat extent and function thus include all those already presented for water quality, because the most important goals of preventing loss of marsh and ultimately other shallow shoreline habitats and of avoiding expansion of hypoxic bottom areas are held in common. However, additional approaches may be available or necessary to respond to risks of areal and functional declines in estuarine habitats. At local levels, expanding the planning horizons of land use planning to incorporate the predictions of consequences of global change over at least a few decades would represent a rational proactive process. Such a longer view could inhibit risky development and simultaneously provide protections for important estuarine habitats, especially salt marshes and mangroves at risk from barriers that inhibit recession. Land use plans themselves rarely incorporate hard prohibitions against development close to sensitive habitats and have limited durability over time, as local political pressure for development and desires for protection of environmental assets wax and wane. Nevertheless, requiring planners to take a longer-term view could have only positive consequences in educating local decision makers about what lies ahead under alternative development scenarios. States run ecosystem restoration programs, largely targeted toward riparian wetlands and tidal marshes. The choice of sites for such restoration activities can be improved by strategically selecting only those where the restored wetland can move a sufficient distance up-slope as sea level rises. Thus, planning and decision making for ecosystem restoration may require purchase of upland development rights or property to insure transgression potential, unless that upland is already publicly owned and managed to prevent construction of any impediment to orderly movement. This consideration of building in resilience to future climate change is necessary for compensatory habitat restorations that must mitigate for past losses for any restoration project that is projected to last long enough that recession would occur. In areas that are presently largely undeveloped, legislation requiring establishment of rolling easements represents a more far-reaching solution to preventing erection of permanent barriers to inland migration of tidelands. Rolling easements do not require predictions about the degree and rate of sea level rise and shoreline erosion. Purchasing development rights has the disadvantage that the uncertainty about rate of sea level rise injects uncertainty over whether enough property has been protected. In addition, rolling easements allow use of waterfront property until the water levels rise enough to require retreat and thus represent a lower cost (Titus, 2000). Implementation of either solution should not be delayed because delay will risk development of the very zone that requires protection.

At state and federal levels, environmental impact statements and assessments of consequences of beach nourishment do not sufficiently incorporate consideration of climate change and its impacts. Similarly, management policies at state and local levels for responding to the joint risks posed by sea level rise and increased frequencies or intensities of storms, including hurricanes, have not recognized the magnitude of growth in costs of present shoreline protection responses as global change continues. Most state coastal management programs discourage hardening of shorelines such as installation of sea walls, groins, and jetties, because they result in adverse effects on the extent of the public beach (Pilkey and Wright III, 1988). Beach nourishment, a practice involving repeated use of fill to temporarily elevate and extend the width of the intertidal beach, is the prevailing (Titus, 2000), rapidly escalating, and increasingly expensive alternative. On average, the fill sands last three to five years (Leonard, Clayton, and Pilkey, 1990) before eroding away, requiring ongoing nourishment activities indefinitely. As sea level rises, more sand is needed to restore the desired shoreline position at escalating cost. The public debate over environmental impacts of and funding for beach nourishment will change as longer-term consequences are considered. Because beach nourishment on coastal barriers inhibits overwash of sediments during storms and the consequent landward retreat of the coastal barrier, erosion of the estuarine shoreline is intensified without this source of additional sediments. Continually elevating the shore of barrier land masses above their natural level relative to depth on the continental shelf implies that wave energy will not be as readily dissipated by bottom friction as the waves progress towards shore. This process brings more and more wave energy to the beach and increases risk of storm erosion and substantial damage to the land mass in major storms. Within less than a century, the rising sea may induce geomorphological changes historically typical of geological time scales (Riggs and Ames, 2003). These changes include predicted fragmentation of coastal barriers by new inlets and even disintegration and loss of many coastal barriers (Riggs and Ames, 2003). Such changes would cause dramatic modifications of the estuaries lying now in protected waters behind the coastal barriers and would shift inland the mixing zone of fresh and salt waters. As climate change progresses and sea level continues to rise, accompanied by higher frequencies of hurricanes and other storms, the beach nourishment widely practiced today on ocean beaches (Titus, 2000) may become too expensive to sustain nationwide (Titus *et al.*, 1991; Yohe *et al.*, 1996), especially if the federal government succeeds in withdrawing from current funding commitments. Miami Beach and other densely developed ocean beaches are likely to generate tax dollars sufficient to continue beach nourishment with state and local funding. Demand for groins, geotubes, sand bags, and other structural interventions will likely continue to grow as oceanfront property owners seek protection of their investment. These come at a price of loss of beach, which is the public trust resource that attracts most people to such areas. Retreat from and abandonment of coastal barriers affected by high relative rates of sea level rise and incidence of intense storms does not seem to represent a politically viable management adaptation.

7.3.2.4 Preserving Human Values

All national estuaries recognize that estuaries provide diverse ecosystem services to people living in close proximity and to others who benefit from the estuaries' resources and functions, even passively. This category of human values relies on so many functions that the CCMPs vary widely in terms of the services they highlight and target for special management protection or restoration. Various consequences of climate change will modify these human values, and a complete assessment of how and by how much for each of the diverse values would be extensive. Nevertheless, it is clear that implications of many predictable climate-induced changes in the estuarine ecosystems are serious. Humans have a public trust stake in all other major management targets of the national estuaries, including water quality, fish and wildlife, and habitat, so to that extent we already address issues of perhaps the most importance to human interests in the estuary. However, other human values not expressly included deserve comment. Conflicts between private values of people living on estuarine shores and the public trust values are already evident but will become increasingly prominent as sea level rises.

Probably, the most serious effects of climate change on private human values associated with estuaries are those arising from climate-driven increases in shoreline erosion, flooding, and storm damage. Rising sea level and increased incidence of intense storms brings higher risk of extensive loss of real estate, houses, infrastructure, and even lives on estuarine shores. The houses and properties at greatest risk are those on coastal barriers lying between the ocean and outer estuary because development on such coastal barriers is exposed during major storms to large waves in addition to storm surge and high winds. Economic and social costs of major storm events under conditions of elevated sea level may be staggeringly high, as illustrated by hurricane damage during the past decade. The management of such risks can already be considered proactive: on ocean beaches, nourishment is practiced to widen and elevate the beach and bulkheads are widely installed on estuarine shorelines. However, each of these defenses is largely ineffective against major storms, and climate change predictions project more such storms developing on a continually warming Earth. Additional proactive management in the future may involve construction of dikes and levees designed to withstand major storms and capable of vertical extension as sea level increases. Such intervention into natural processes on ocean and estuarine shores is technically feasible but probably affordable only where development is intense enough to have created very high aggregate real estate values. It sacrifices public trust values for private values. Long-term sustainability of such barriers is questionable. In places experiencing rapid erosion but lacking dense and expensive development, shoreline erosion is likely to be accepted and retreat and abandonment occur. Even before extensive further storm-related losses of houses, businesses, and infrastructure on ocean and estuarine shores, property values may deflate as sea level and risks of storm and flood damage increase. Many property insurers are already cancelling coverage and discontinuing underwriting activities along wide swaths of the coast in the areas most at risk to hurricanes, from Texas through New York. State governments are stepping into that void, but policy coverage is far more costly. Availability of mortgage loans may be the next economic blow to coastal development. As losses from storms mount further, the financial risks of home ownership on estuarine shorelines may create decreased demand for property and thus cause losses in real estate values.

Comprehensive planning could be initiated now at federal, tribal, state, and local levels to act proactively, or opportunistically after major storm events, to modify rules or change policies to restructure development along coastal barrier and estuarine shorelines to avoid future loss of life and property, and at the same time protect many environmental assets and ecosystem services in the interest of the public trust. For example, doing up-front planning to prevent rebuilding in hazardous areas of high flood risk and storm damage may be feasible. Establishing setbacks from the water and buffer widths, based on the new realities of shoreline erosion and on reliable predictions of shoreline position into the future, may be possible if advance planning is complete so that rules or policies can be rapidly implemented after natural disasters. Many programs such as federal flood insurance and infrastructure development grants subsidize development. For undeveloped coastal barriers, such subsidies were prohibited by the Coastal Barriers Resources Act and these prohibitions could be extended to other estuarine and coastal shorelines now at high and escalating risk. Local land use plans could be modified to influence redevelopment after storms and direct it into less risky areas. Nevertheless, such plans would result in financial losses to property owners who cannot make full use of their land. Land trusts and programs to protect water quality, habitat, and fisheries may provide funding to purchase the most risky shorelines of high resource value.

7.3.2.5 Water Quantity

Many national estuaries, especially those on the Pacific coast where snowmelt is a large determinant of the hydroperiod, identify water quantity issues among their management priorities. Such water quantity issues will become growing concerns directly and indirectly for all estuaries as climate continues to change. Projected global climate change includes modifications in rainfall amount and temporal patterns of delivery, in processes that influence how much of that rain falling over the watershed reaches the estuary, and in how much salt intrusion occurs from altered river flows and rising sea levels penetrating into the estuary. These climate changes interact strongly with human modifications of the land and waterways as well as with patterns of water use and consumption. The models predicting effects of climate change on rainfall amount are not all in agreement, complicating adoption of proactive management measures. Thus, complex questions of adaptive management arise that would help smooth the transition into the predictably different rainfall future, whose direction of change is uncertain. Many of these questions will have site (basin)-specific conditions and solutions; however a generic overview is possible.

As freshwater delivery patterns change and salt water penetration increases in the estuaries, many processes that affect important biological and human values will be affected. Where annual freshwater delivery to the estuary is reduced, and in cases where only seasonal reductions occur, salt water intrusion into groundwater will influence the potable yield of aquifers. In the Pacific Northwest, predicted patterns of precipitation change imply that increased salt water penetration up-estuary will be a summertime phenomenon when droughts are likely. Fresh water is already a limiting resource globally (Postel, 1992) and is a growing issue in the United States even in the absence of climate change. Failure to develop proactive management responses will have serious consequences on human welfare and economic activity. Proaction includes establishing or broadening “use containment areas” (where withdrawal is allocated and capped) in the managed allocation of aquifer yields so that uses are sustainable even under predicted climate-related changes in recharge rates and salt water infiltration. This may result in the need to develop reverse osmosis plants to produce potable water and replace ground water sources currently tapped to supply communities around estuaries. Further actions may be needed to modify permitting procedures for affected development, plan for growing salt water intrusion as sea level rises, and maintain aquifer productivities. Proactive planning measures for water shortage can include much greater water re-use and conservation.

The enhanced flashiness of run-off from seasonal rainfall events, as they come in discrete, more intense storms and fall upon more impervious surface area in the drainage basin, will have several consequences on human values and on natural resources of management priority. Greater pulsing of rain runoff reaching the rivers will lead to much higher frequency and extent of floods after intense storms. The resulting faster downstream flows will erode sediment from estuarine shorelines and thus reduce the area of shallow habitats along the shores. In the Pacific Northwest, rain-on-snow events are major sources of flood waters (Marks *et al.*, 1998; Mote *et al.*, 2003) and are likely to become more frequent and intense under current climate change scenarios. These events have economic, health and safety, and social consequences for humans living or working in the newly enlarged flood plain. Bank stability and riparian habitats are threatened by increased water velocities in flood flows, which would affect water quality and ultimately fish and wildlife. When these pulses of water reach the estuary, they bring pollutants from land as well as nutrient and organic loading that have negative effects on estuarine functions for relatively long periods of time, on the order of a year or more. In estuaries where freshwater runoff is increased by global climate change, and in all estuaries where salt water has penetrated further upstream as sea level rises, the specific locations of important zones of biogeochemical processes and biotic use will shift in location. These shifts may have the effects of moving those zones, such as the turbidity maximum zone, which could influence the performance of anadromous fishes that make use of different portions of the rivers and estuaries for completing different life history stages and processes. Accurate modeling of such position changes in estuaries could allow proactive management to protect fish and wildlife habitats along the rivers and estuaries that will become critical habitats for propagation of important fish stocks as positional shifts occur.

7.3.3 New Approaches to Management in the Context of Climate Change

Little attention has historically been paid to preserving and enhancing ecosystem resilience in the management of estuaries and estuarine resources. Resilience refers to the amount of disturbance that can be tolerated by a socioecological system (*e.g.*, an estuary plus the social system interacting with it) before it undergoes a fundamental shift in its structure and functioning (Holling, 1972; Carpenter *et al.*, 2001; Gunderson *et al.*, 2002; Carpenter and Kinne, 2003). The ability of a system to maintain itself despite gradual changes in its controlling variables or its disturbance regimes is of particular concern for those interested in predicting responses to climate change. Importantly, resilience of a socioecological system results in part from appropriate management strategies. Human behaviors can reduce resilience in a variety of ways, including increasing flows of nutrients and pollutants; removing individual species, whole functional groups (*e.g.*, seagrasses, bivalves), or whole trophic levels (*e.g.*, top predators); and altering the magnitude, frequency, and duration of disturbance regimes (Carpenter *et al.*, 2001; Folke *et al.*, 2004). Importantly, climate change has the potential to exacerbate poor management and exploitation choices and cause undesirable regime shifts in ecosystems, as seen in the North Sea cod fishery and recent declines in coral reefs (Walther *et al.*, 2002). It is critical that we pursue wise and active adaptive management in order to prevent undesirable regime changes in response to climate change.

In recent years, basic research has dramatically improved our understanding of the ecosystem characteristics that help promote resilience. For example, the study of the roles of biodiversity in ecosystem dynamics has demonstrated several examples where productivity (Tilman and Downing, 1994; Naeem, 2002), biogeochemical functioning (Solan *et al.*, 2004), and community composition (Duffy, 2002; Bruno *et al.*, 2005) are stabilized under external stresses if biodiversity is high. Worm *et al.* (2006) likewise demonstrated that many services of marine ecosystems, including fisheries production, and ecosystem properties, such as resilience, are greater in more diverse systems. Some evidence exists to suggest that proliferation of non-native species can be suppressed by ecosystem biodiversity (*e.g.*, Stachowicz, Whitlatch, and Osman, 1999; but see Bruno *et al.*, 2004). These research results have not yet been directly translated into management of estuarine systems. This represents a promising approach to the goal of enhancing adaptation in contemplation of climate change. However, acting on the knowledge that higher biodiversity implies higher resilience represents a challenge.

Absent system-specific knowledge, some management actions are likely to preserve or enhance biodiversity (genetic, species, and landscape) and thus support resilience, based upon current theory and some empirical evidence. Maintaining high genetic diversity provides high potential for evolutionary adaptation of species and provides short-term resilience against fluctuating environmental conditions (Hughes and Stachowicz, 2004).

This goal may be achieved by establishing diversity refuges, which in aggregate protect each of a suite of genotypes. Implementing this proactive management concept depends on knowledge of genetic diversity and spatial patterns of its genotypic distribution, a task most readily achieved for structural habitat providers such as marsh and sea grasses and mangroves. Maintaining or restoring habitat and ecosystem diversity and spatial heterogeneity is another viable management goal, again most applicable to the important plants that provide habitat structure. Preserving or creating landscapes of the full mix of different systems and including structural corridors among landscape elements otherwise fragmented or isolated can be predicted to enhance resilience by enabling migrations to sustain biodiversity across the landscape (Micheli and Peterson, 1999). Structural complexity of vegetation has been related to its suitability for use of some (endangered) species (Zedler, 1993), so preserving or restoring the vegetational layering and structure of tidal marshes, seagrass meadows, and mangroves has potential to stabilize estuary function in the face of climate perturbations.

Analogous need exists for enhanced understanding of factors that contribute to resilience of human communities and of human institutions in the context of better preparation for consequences of changing climate. Both social science and natural science monitoring may require expansion to track possible fragility and look for signs of cracks in the system, as a prelude to instigating adaptive management to prevent institutional and ecological disintegration. For example, more attention should be paid to tracking coastal property values, human population movements, demography, insurance costs, employment, unemployment, attitudes, and other critical social and economic variables in order to indicate need for proactive interventions as climate change stresses increase. An analogous enhancement of in-depth monitoring of the natural ecosystem also has merit; this likely would require changes in indicators now monitored to be able to enhance resilience through active intervention of management when the need becomes evident. Thus, monitoring in a context of greater understanding of organizational process in socio-economic and natural systems is one means of enhancing resilience.

Both managers and the general public need better education to raise awareness of how important management adaptation will be if negative impacts of climate change are to be averted or minimized. Surely, managers undergo continuing education almost daily as they conduct their jobs, but targeted training on expected changes within the ecosystem they are responsible for managing is an emerging necessity. Re-education is necessary to counteract the disinformation that has recently been circulated to support agendas of various interest groups. Careful articulation of uncertainties about the magnitudes, timelines, and consequences of climate change will also be important. Such education is vital to induce the broad conversations necessary for public stakeholders and managers to rethink in fundamental ways how we have previously treated and managed estuaries to provide goods and services of value.

Whereas we have used the term “management adaptation” to mean taking management actions that expressly respond to or anticipate climate change and are intended to counteract or minimize any of its negative implications, natural resource managers and academics have developed a different process termed “adaptive management” (Walters, 1986). Adaptive management in this context (see Chapter 9, Synthesis) refers to designing and implementing regulations or other management actions as an experiment, and employing rigorous methods of assessing the impacts of the management action. Monitoring the status of the response variables provides the data against which the management action's effectiveness can be judged. This blending of experimental design into management provides perhaps the most rigorous means of testing implications of management actions. Adaptive management has the valuable characteristic that it continuously re-evaluates the basis on which predictions are made, so that as more information becomes available to reduce the uncertainties over physical and biological changes associated with climate change, the framework of adaptive management is in place to incorporate that new knowledge. Use of this approach where feasible in testing management adaptations to global climate change can provide much needed insight in reducing uncertainty about how to modify management to preserve delivery of ecosystem services.

Because its holistic nature includes the full complexity of interactions among components, the most promising new approach to adapt estuarine management to global climate change is the further development and implementation of ecosystem-based management (EBM) of estuarine ecosystem services in a way that incorporates climate change expectations (Peterson and Estes, 2001). The concept of EBM has its origins among land managers, where it is most completely developed (Grumbine, 1994; Christensen *et al.*, 1996). EBM is an approach to management that strives for a holistic understanding of the complex of interactions among species, abiotic components, and humans in the system and evaluates this complexity in pursuit of specific management goals (Lee, 1993; Christensen *et al.*, 1996). Ecosystem-based management explicitly considers different scales and thus may serve to meet the challenges of estuarine management, which ranges across scales from national and state planning and regulation to local implementation actions. Practical applications of the EBM approach are now evolving for ocean ecosystems (Pikitch *et al.*, 2004) and hold great promise for achieving sustainability of ecosystem services. Both the Pew Oceans Commission (2003) and the U.S. Commission on Ocean Policy (2004) have identified EBM as our greatest hope and most urgent need for preserving ecosystem services from the oceans. The dramatic potential impacts of climate change on estuarine ecosystems imply many transformations that simply developing and applying EBM cannot reverse, but development of synthetic models for management hold great promise for optimizing estuarine ecosystem services in a changing world. Ecosystems are sufficiently complex that no model will include all components and processes, so the more simplified representations of the estuarine system might best be used to generate hypotheses about the effectiveness of alternative management actions that are then tested through rigorous protocols of adaptive management. One widely advocated approach to implementing EBM does not require an elaborated understanding of ecosystem structure and dynamics, and may be applicable to solve important management challenges in estuaries; it is the implementation of marine protected areas (Halpern, 2003; Roberts *et al.*, 2003; Micheli *et al.*, 2004). This tool is most applicable where fishery exploitation and collateral habitat injury exist; clearly, these issues apply to many estuarine systems.

7.3.4 Prioritization of Management Responses

Setting priorities is important to the development of management adaptations to respond to global climate change. Because responsibilities for managing estuaries are scattered among so many different levels of government and among so many different organizations within levels of government, building the requisite integrated plan of management responses will be difficult. EBM is designed to bring these disparate groups together to achieve the integration and coordination of efforts (Peterson and Estes, 2001), but implementing EBM for national estuaries and other estuaries may require changes in governance structures. The State of North Carolina has made progress in bringing together diverse state agencies with management authority for aspects of estuarine fisheries habitats in its Coastal Habitat Protection Plan, which approaches an EBM. However, this governance method is targeted toward producing fish rather than the complete scope of critical estuarine functions and broad suite of estuarine goods and services. This model approach also lacks a mechanism to engage the relevant federal authorities. The national estuaries actually bring to the table a wider range of managers and stakeholders, including those from federal, tribal, state, and local levels, as are contemplated in the genesis of an EBM plan. However, the CCMPs that arise from the national estuaries do not carry any force of regulation and often lack explicit numerical targets, instead expressing wish lists and goals for improvements that are probably unattainable without substantially more resources and powers. Perhaps the national estuaries could provide the basis for a new integrative governance structure for estuaries that could be charged with setting priorities among the many management challenges triggered by global climate change.

Factors that probably would dictate priorities are numerous, including socio-economic consequences of inaction, feasibility of effective management adaptations, the level of certainty about the projected consequence of climate change, the time frame in which action is best taken, the popular and political support for action, and the reversibility of changes that may occur in the absence of effective management response. Clearly, the processes that threaten to produce the greatest loss of both natural ecosystem services and human values is the rise of sea level and ascendancy of intense storms with implications for land inundation, property loss, habitat loss, water quality degradation, declines in fisheries and in wildlife populations associated with shallow shoreline habitats, and salt water intrusion into aquifers. This issue attracts the most attention in the media and from the public, but the global capping of greenhouse gases may not represent a feasible management response. Thus, various means of removing and preventing engineered shoreline armoring such as bulkheads, levees, and dikes, combined with shoreline property acquisition may be the focus of discussion if their costs are not an overwhelming impediment. Because the complexity of intermingled responsibilities for managing interacting components inhibits establishment of ecosystem-based management, attention to modifying governance structures to meet this crisis would also rank high among priorities.

7.4 Case Study: The Albemarle-Pamlico Estuarine System

7.4.1 Introduction

We chose the Albemarle-Pamlico Estuarine System (APES) for our case study. APES provides a range of ecosystem services, extending over a diversity of ecosystem types, which provide the basis for the management goals of the Albemarle-Pamlico National Estuary Program (APNEP). Like other estuaries, the ecosystem services of APES are climate sensitive, and this sensitivity affects the ability to meet management goals. A range of adaptation options exist for climate-sensitive management goals. Many of these adaptation options are applicable across estuarine ecosystems generally. Furthermore, because APNEP represents one of the first national estuaries, documentation of management successes and failures (Korfmacher, 1998; Korfmacher, 2002) exists for its 20-year history. Extensive data and decision support information are available for the system and are likely to continue to be gathered into the future. We highlight a few key climate-related issues in this case study, including warming and altered precipitation patterns, but especially accelerated sea level rise and increased frequency of intense storms.

7.4.2 Historical Context

Like many important estuaries, the Albemarle-Pamlico ecosystem has experienced a long history of human-induced changes including species depletion, habitat loss, water quality degradation, and species invasion (Lotze *et al.*, 2006). About 800 years ago, indigenous Native Americans initiated agriculture in the basin, and approximately 400 years ago Europeans began to colonize and transform the land. Since then, the human population around the estuary has increased by two orders of magnitude from that in 1700 (Lotze *et al.*, 2006). Before European colonization, North Carolina had about 11 million acres of wetlands, of which only 5.7 million remain today. About one-third of the wetland conversion, mostly to managed forests and agriculture, has occurred since the 1950s (U.S. Geological Survey, 1999). Since 1850, the amount of cropland has increased 3.5-fold. More recent land use patterns show that 20% of the basin area consists of agricultural lands, 60% is forested, and relatively little is urbanized (Stanley, 1992). Over the last three decades, the production of swine has tripled and the area of fertilized cropland has almost doubled (Cooper *et al.*, 2004). These changes in land-use patterns and increases in point and non-point nutrient loading have induced multiple changes in water quality, with the greatest changes appearing during the last 50–60 years (Cooper *et al.*, 2004).

Over the last two to three centuries in the Albemarle and Pamlico Sounds, overexploitation, habitat loss, and pollution have resulted in the depletion and loss of many marine species that historically have been of economic or ecological importance (Lotze *et al.*, 2006). Of the 44 marine mammals, birds, reptiles, fish, invertebrates, and plants for which sufficient time series information exists, 24 became depleted (<50% of former abundance), 19 became rare (<90%), and 1 became regionally extinct by 2000 (Lotze *et al.*, 2006). Great losses also occurred among the subtidal bottom habitats. Historical accounts from the late 1800s indicate that bays and waterways near the mainland once had extensive beds of seagrass, while today seagrass is limited to the landward side of the barrier islands (Mallin *et al.*, 2000). Oyster reef acreage has been diminished over the last 100 years as a consequence of overharvesting, habitat disturbance, pollution, and most recently Dermo (*Perkinsus marinus*) infections (North Carolina Department of Environmental and Natural Resources, 2006).

7.4.3 Geomorphological and Land Use Contexts and Climate Change

Climate change impacts on APES may take numerous forms. Warming in and of itself can alter community and trophic structure through differential species-dependent metabolic, phenological, and behavioral responses. Changes in precipitation patterns also may have species-specific consequences. In combination, warming and precipitation patterns affect evapotranspiration, soil moisture, groundwater use and recharge, and river flow patterns. The current rate of relative rise in mean sea level in this geographic region is among the highest for the Atlantic coast, with estimates commonly over 3 mm per year and in at least one study as high as 4.27 mm per year (Zervas, 2001). The anticipated scenario of increasing frequency of intense storms in combination with rising sea levels creates a likelihood of dramatic physical and biological changes in ecosystem state for APES because the very integrity of the Outer Banks that create the protected estuaries behind them is at risk (Riggs and Ames, 2003; Paerl *et al.*, 2006).

APES is a large and important complex of rivers, tributary estuaries, extensive wetlands, coastal lagoons and barrier islands. Its 73,445 km^2 watershed (Stanley, 1992) is mostly in North Carolina but extends into southern Virginia (Figure 7.3). The largest water body is Pamlico Sound to the southeast, with two major tributaries, the Neuse and the Tar-Pamlico Rivers. Both rivers empty into drowned river estuaries, the Neuse River Estuary (NRE) and the Pamlico River Estuary (PRE), which connect to Pamlico Sound. Albemarle Sound is farther north with two major tributaries, the Chowan and the Roanoke Rivers, and a number of local tributary estuaries. Other smaller sounds connect the Albemarle and the Pamlico (Roanoke and Croatan Sounds), and the Currituck Sound extends along the northeastern portion of the complex.

Figure 7.3. The Albermarle-Pamlico National Estuary Program region (Albemarle-Pamlico National Estuary Program, 2007).

The geological framework for coastal North Carolina, including APES has recently been summarized by Riggs and Ames (2003). The system represents several drowned river valley estuaries that coalesce into its large coastal lagoon (Figure 7.3). The coastal plane, estuaries and sounds have a very gentle slope in which Quarternary sediments are underlain largely by Pliocene sediments. Much of this sediment is organic rich mud arising from eroding peat of swamps and marshes (Riggs, 1996). The gentle slope has allowed major shifts in position of the shoreline and barrier islands as sea level has risen and fallen. Furthermore, the position and number of inlets has changed along the barrier islands, promoting or limiting the exchange of fresh and seawater.

Much of the watershed is within the coastal plain with low elevations that affect land use. Moorhead and Brinson (1995) estimate that 56% of the peninsula between the Albemarle Sound and PRE is less than 1.5 m in elevation. Fifty-three percent of the peninsula's area is composed of wetlands, and 90% contains hydric soils. Thus, this region of the watershed is sparsely populated and largely rural. In contrast, other regions are more highly developed. The barrier islands, the famous "Outer Banks" of North Carolina, are a mosaic of highly developed lands for tourism and protected natural areas. The southeastern portion of Virginia in the APES basin is highly urbanized, and the piedmont origins of the Neuse and Tar Rivers in North Carolina are highly populated. Agriculture and silvaculture are important land uses and economic drivers in the region. Urban economies dominate much of southeastern Virginia. And a relatively new trend is the development of high-end and retirement subdivisions along the "Inner Banks," the mainland shore zone of the complex. The watershed's population exceeds 3,000,000 people including Virginia. However, only about 25% are found in coastal counties of North Carolina, based on estimates for 2000 (Federal Emergency Management Agency, 2007). A significant portion of this population is considered "vulnerable" to strong storms and thus faces risks from climate change (*i.e.,* people who live in evacuation zones for storm surge or who are subject to risks from high winds by living in mobile homes). The low-lying lands and basic nature of services and infrastructure of the rural environment pose growing risks of flood damage as sea level and storm intensities rise to land uses, infrastructure (*e.g.,* water delivery from aquifers, waste water treatment facilities, roads, and buildings) and even human lives.

Another characteristic of the system's geomorphology makes it uniquely susceptible to climate change drivers. The exchange of water between the ocean and the sounds is restricted by the few and small inlets that separate the long, thin barrier islands (Giese, Wilder, and Parker, 1985; Riggs and Ames, 2003). This restricted connectivity greatly dampens amplitude of astronomical tides and limits the degree to which seawater is mixed with freshwater. Temperature increases may have significant impacts on the APES because its shallow bays have limited exchange with ocean waters, which serve as a cooling influence in summer.

Water quality has been a recurring management concern for APES and APNEP. The tributary rivers generally have high concentrations of dissolved nutrients. This fosters high primary productivity in tributary estuaries, but under most circumstances nutrient concentrations in the sounds remain relatively low (Peierls, Christian, and Paerl, 2003; Piehler *et al.*, 2004). Most nutrient loading derives from non-point sources, although nitrogen loading from point sources may account for up to 60–70% in summer months (Steel and Carolina, 1991). Nitrogen deposition from the atmosphere may account for an additional 15–32% (Paerl, H.W., Dennis, and Whitall, 2002). Phosphorus loading to the Pamlico River Estuary was greatly enhanced by phosphate mining, which accounts for about half of the total point source phosphorus loadings to this estuary and officially began in 1964 (Copeland and Hobbie, 1972; Stanley, 1992). Loading has decreased dramatically in recent years as treatment of mine wastes has improved. High surface sediment concentrations of the toxic heavy metals arsenic, chromium, copper, nickel, and lead are found in the Neuse River Estuary, possibly associated with industrial and military operations, while high cadmium and silver levels in Pamlico River Estuary most likely result from phosphate mining discharges (Cooper *et al.*, 2004). In 1960, hypoxia was first reported in the Pamlico River Estuary (Hobbie, Copeland, and Harrison, 1975). Since then, hypoxic and anoxic waters in the Pamlico River Estuary and Neuse River Estuary were mostly of short duration (days to weeks) but have resulted in death of benthic invertebrates on the bottom and fish kills (Stanley and Nixon, 1992; Buzzelli *et al.*, 2002; Cooper *et al.*, 2004). Nuisance and toxic algal blooms are reported periodically (Burkholder *et al.*, 1992; Bricker *et al.*, 1999), and about 22 aquatic plants and 116 aquatic animals, of which 22 occur in marine or marine-freshwater habitats, have been identified as non-indigenous species in North Carolina (U.S. Geological Survey, 2005). Increases in temperatures are expected to enhance hypoxia and its negative consequences, through the combined effects of increased metabolism and, to a lesser degree, decreased oxygen solubility.

The interactions between relative sea level rise, shoreline morphology, and bay ravinement could have significant impacts on estuarine water quality and ecosystem function in the APES. Losses of wetlands to inundation could lead to a large shift in function from being a nitrogen sink to being a nitrogen source. Both planktonic and benthic primary producers may be affected by, and mediate, changes in water quality, nutrient and material fluxes across the sediment-water interface that may result from sea level rise (Figure 7.4). Changes in the water column productivity affect particle composition and concentration, which in turn increases turbidity and feedback to modify further the balance between water column and benthic productivity. Inundated sediments will then be subject to typical estuarine stressors (*e.g.,* salinity, changes in water table, isolation from atmosphere) that can lead to dissolution of particulates, desorption of nutrients or organic matter, and altered redox states. These changes result in fluxes of nutrients and DOC that could radically transform the proportion of productivity and heterotrophic activity in the water above the sediment and in the rest of the estuary. Nutrient management plans generally assume that the frequency and magnitude of bottom water hypoxia will decrease by reducing watershed inputs of dissolved inorganic nitrogen and organic matter that either indirectly or directly fuel water column and benthic respiration (Kemp *et al.*, 1992; Conley *et al.*, 2002). However, factors such as the nutrient and sediment filtration capacity of wetlands under flooded conditions of higher sea levels, and the potential for a large organic matter input from erosion and disintegration of now inundated wetlands, create uncertainty about progress in containing eutrophication across different scales and render the determination of management targets and forecasting of hypoxia extremely difficult.

Figure 7.4. Feedbacks between nutrient and sediment exchange and primary production in the benthos and water column. A plus symbol indicates enhancement and a minus symbol suppression.

Because of the large fetch of the major sounds and tributary estuaries, wind tides control water levels and wave energy can be quite high. Wind tides can lead to extended flooding and high erosion rates, especially within the eastern and southern parts of the complex (Brinson, 1991; Riggs and Ames, 2003). Furthermore, the barrier islands are prone to breaching during storms, and geological history demonstrates the fragility of this thin strip of sand and reveals the locations of highest risk of breaching. Formation of persistent inlets within the barrier islands would increase oceanic exchange and thereby the amplitude of astronomical tides. This, in turn, could profoundly alter the ecology of both aquatic and wetland ecosystems in the APES.

The size, geomorphology, and location of the APES complex make it an important source of ecosystem services for the region and the nation. The largest economic contribution of APES today derives from tourism and recreation. The Outer Banks attract people from around the world. Populations during the prime summer season considerably exceed winter populations. The Outer Banks include the most economically important acreage of the complex along with ecologically important natural areas. These coastal barriers are also the most sensitive to the combination of sea level rise and increased frequency of intense storms. Barrier island geomorphology is constantly changing on short and long time scales, increasing and decreasing in width with sand movement and both forming and closing inlets during storms. Inlets have broken through the Outer Banks repeatedly over the past century and paleo records from the past few thousand years demonstrate dramatic movements in location and character of the barriers as sea level has changed (Riggs and Ames, 2003). But human structures on the islands and human uses of the barrier islands' natural resources have now changed the degree to which natural geological processes occur. Construction and maintenance of Route 12 along the Outer Banks has restricted washover and the movement of sand from the seaward side of the islands to the sound side. Furthermore, the presence of houses, condominiums, hotels, etc. produces conflicts between maintaining the natural geomorphic processes that allow island migration landwards as sea level rises and protecting human infrastructure. Rising sea level and increased frequency of intense storms enhances the potential beach erosion, thereby increasing costs of beach nourishment, and increases risk of island disintegration, leading to increased political pressure to legalize hard structures on the ocean shoreline.

Beaches are a major natural resource and drive many coastal economies. Because the presence of houses, condominiums, and roads and other infrastructure leads to defense of the shoreline position and prevents natural recession, beach erosion now reduces beach widths as sea level is rising. North Carolina prohibits hard structures (*e.g.*, bulkheads, jetties, and permanent sand bags) on the ocean shoreline. Instead, erosion is countered by beach nourishment, in which sand is dredged from offshore. This is a temporary and expensive solution. It also has potentially significant impacts on the living resources of the beach, such as shorebirds and resident invertebrates (Peterson and Bishop, 2005; Peterson *et al.*, 2006). Erosion of beaches tends to occur with the major axis parallel to the islands (*i.e.*, meters or tens of meters of erosion of beach along hundreds to thousands of meters along the beach face). Breaching of new inlets and overwash events penetrate more into the islands. A recent breach occurred on Hatteras Island during Hurricane Isabel, but it was quickly closed by the U.S. Army Corps of Engineers to permit road reconstruction and automobile travel along the Outer Banks. Riggs and Ames (2003) have projected that under higher stands of sea level, future hurricanes may create numerous large, new inlets and break the chain of coastal barriers that forms the eastern edge of the entire APES system. They mapped locations of the paleochannels along the islands and identified these as the most likely locations for such breaches. Such events represent the most dramatic consequences of climate change to APES. Extensive new inlets would lead to an entirely new tidal, salinity, wave, and hydrodynamic regime within APES, and in turn drastically change the ecology of the complex. Wise management for the future must include preparation for the possibility of events such as these and their consequences.

Natural areas in APES have been recognized for their significance as wildlife habitat, nurseries for aquatic species, stop-over sites (flyways) for migratory birds, and important spawning areas for anadromous fish. Recreational fishing and boating add to the attraction of the beaches, barrier islands, and natural areas within the watershed. The nursery services of the complex are also important to fisheries, both locally and along the entire eastern coast of the United States. Cape Hatteras sits at the biogeographic convergence of populations of northern and southern species, and many of these species use the sounds during their life cycles. Thus, the location of APES makes it particularly sensitive to any climate-related changes that alter migratory patterns of both birds and marine organisms.

The wetlands of the Albemarle Pamlico Sound complex are largely non-tidal and subject to irregular wind tides, as described above. In freshwater regions along the rivers and flood plains, swamp forests dominate. Pocosins—peat-forming ombrotrophic wetlands—are found in interstream divides. As sea level rises in oligohaline regions, swamp forests may continue to dominate or be replaced by brackish marshes. Irregularly flooded marshes, dominated by *Juncus roemerianus*, extend over much of the higher-salinity areas. Back barrier island marshes are dominated by *Spartina alterniflora*. The ability of these wetlands to respond to sea level rise is becoming compromised by increased human infrastructure. Roads, residential and urban developments, hard structures for shoreline stabilization, and agricultural ditching are preventing horizontal transgression of wetlands and promoting erosion of edges throughout the complex. Furthermore, development of the barrier islands has prevented natural overwash and inlet-forming processes that promote salt marsh development (Christian *et al.*, 2000; Riggs and Ames, 2003).

7.4.4 Current Management Issues and Climate Change

The Albemarle-Pamlico Estuarine System became part of the NEP (APNEP) in 1987. Initial programmatic efforts focused on assessments of the condition of the system through the Albemarle-Pamlico Estuarine Study. The results of these efforts were used in the stakeholder-based development of a Comprehensive Conservation and Management Plan (CCMP) in 1994. The CCMP presented objectives for plans in five areas: water quality, vital habitats, fisheries, stewardship, and implementation (Box 7.8) (Albemarle-Pamlico National Estuary Program, 1994). For each objective, issues of concern were identified and management actions proposed. None of the issues or proposed actions explicitly included climate change. In 2005, NEP Headquarters conducted its most recent triennial implementation review of APNEP. APNEP passed the implementation review and was found eligible for funding through FY 2008.

Although no management objective explicitly identifies climate change or its consequences, water quality, vital habitats, and fisheries are likely to be substantially affected by changes in climate. Recent efforts by APNEP and the State of North Carolina led to more direct consideration of the impacts of climate change. APNEP has identified indicators of condition of the system and begun the process for implementing their use. Multiple indicators assess condition of atmosphere, land, wetland, aquatic, and human components of the system. While some indicators focus on short-term changes in these components, many have meaning only in their long-term trends. Given a changing climate and associated impacts, these indicators place APNEP in position to assess these impacts for wise management. On a broader front, the legislature of North Carolina in 2006 established a commission on climate change to assess how climate change will affect the state and to propose actions to either minimize impacts or take advantage of them.

In 1987 North Carolina passed the Fisheries Reform Act, requiring both development of formal species management plans for each commercially and/or recreationally harvested fishery stock and the development of a Coastal Habitat Protection Plan (CHPP). The CHPP development and implementation process resembles an EBM at the state level because it requires consideration and integrated management of all factors that affect the quality of fish habitats in a synthetic, integrative fashion. To achieve this goal, staff from all appropriate state resource and environmental commissions came together to map coordinated approaches to achieve sustainability of habitat quantity and quality for fishery resources. This partnership among agencies, while only at the state level, addresses one of the biggest goals of EBM (Peterson and Estes, 2001). Commissions and agencies responsible for fisheries management (Marine Fisheries Commission), water quality and wetlands (Environmental Management Commission), and coastal development (Coastal Resources Commission) are the major entities, but the Sedimentation Control Commission and Wildlife Resources Commission also contribute. The CHPP does contemplate several aspects of climate change and human responses to threats such as beach and shoreline erosion, although long-term solutions are elusive. Now that a plan exists, the implementation of its short-term goals has yet to begin and may become contentious.

Other innovative programs and initiatives within North Carolina are the Ecosystem Enhancement Program (EEP), Clean Water Management Trust Fund (CWMTF), and the designation of estuaries as nutrient sensitive. EEP is an agency that coordinates wetland mitigation efforts to maximize their effectiveness. The North Carolina Department of Transportation’s mitigation needs are largely met through EEP. The program uses a watershed approach in planning mitigation projects. This allows a broad and comprehensive perspective that should be reconciled with climate change expectations. The CWMTF provides financial support for activities that improve or protect water quality. It offers an opportunity to link consideration of climate change to such activities, although no such link has been an explicit consideration. The designation of nutrient sensitivity allows enhanced controls on nutrient additions and total maximum daily loadings to the Neuse and Tar-Pamlico systems. In fact, regulations have been designed to not only curb expansion of nutrient enrichment but to roll it back with restrictions to both point- and non-point sources.

7.4.5 Recommendations for Environmental Management in the Face of Climate Change

We make three overarching recommendations for management of estuaries in the face of climate change: (1) maintain an appropriate environmental observing system; (2) educate a variety of audiences on long-term consequences; and (3) pursue adaptation and adaptive management. Each of these is described specifically for APES but has application to other estuaries in whole or part. Furthermore, each involves coordination of multiple initiatives and programs. It is this coordination that should be a major focus of APNEP in particular and NEP in general.

An appropriate observing system involves a network of programs that detects, attributes and predicts change at multiple scales. It includes sustained monitoring, data and information management, predictive model production, and communication of these products to users. The users include environmental managers, policy makers, and members of the public over a range of economic positions and status. Regulatory and policy needs require a variety of measurements to be made in a sustained way. These measurements extend to variables of physical, chemical, biological, and socioeconomic attributes of APES. Many have been identified by APNEP with its indicator program. These measurements must be made to respond to drivers at different time scales; while these time scales include short-term variation, the most important to this report are long-term trends and infrequent but intense disturbances.

There are other observing system initiatives within coastal North Carolina. These include the North Carolina Coastal Ocean Observing System and Coastal Ocean Research and Monitoring Program. Both have their emphases on the coastal ocean and near real-time products of physical conditions. However, their efforts need to be more directed toward the APES and other estuarine ecosystems to be more valuable to the people of North Carolina. More effort is needed to assess and understand the physical dynamics of the estuarine systems. Observations and analyses should be extended to characterize the physical and geochemical processes of catchment and riverine inflows, which are likely to change dramatically under changing climatic conditions. The systems also need to broaden their observations to include ecological and socioeconomic measurements. These measurements are less likely to be near real-time, but user needs do not require such quick reporting. We recommend that the coastal observing systems be linked explicitly to APNEP indicator activities.

Education is needed across the spectrum of society to produce informed stakeholders and thus facilitate enlightened management adaptations. The need for K–12 education on climate change is obvious, but there is also a lack of general understanding among adults. Education efforts are needed for the general public, policy makers, and even environmental managers. North Carolina has several significant programs that can promote this general understanding. APNEP and the Commission on Climate Change have been mentioned above. Public television and radio have a general mission to educate and have contributed time to the topic. Two other programs are (1) the Partnership for the Sounds, including the Estuarium in Washington, North Carolina, and (2) the North Carolina Aquariums. The latter includes three aquaria along the coast. These programs are in a unique position to teach the general public about climate change. We recommend that coordination among these different programs be fostered to promote education within the state.

Finally, adaptive management and adaptation strategies are essential to respond to the complex implications of climate change. Adaptive management recognizes the need for both sustained monitoring associated with observing systems and adaptive justification of intervention plans that reflect advances in our understanding of impacts of climate change and new insights on what experimental interventions are needed. Adaptive management also recognizes the important role of education that promotes better appreciation of a changing and uncertain world. Adaptive management is explicit within APNEP, CHPP, and EEP. It also is incorporated into controls on nutrient additions to alleviate the impacts of cultural eutrophication. It acknowledges the importance of the ecosystem perspective and breaks the regulatory mold of being specific to an issue, species, single source of pollution, etc. This enhances the ability to meet the challenges of climate change. One aspect of this change is the expectation that landscape units that are controlled by sea level will migrate. Beaches and wetlands will move shoreward. Regulations and policies that foster the ability to retreat from these landscape migrations are part of this adaptive approach. Adaptive management is an established approach in North Carolina, which can serve as a successful example nationally.

7.4.6 Barriers and Opportunities

APNEP possesses environmental and social barriers to effective implementation of management adaptation to climate change, yet at the same time various social and environmental characteristics represent favorable opportunities for adaptation. Indeed, APNEP was chosen for a case study because it could illustrate both significant barriers and opportunities. Perhaps its greatest single barrier to successful adaptation to climate change is the intractable nature of the challenge of preserving the integrity of the coastal barrier complex of the Outer Banks over the long time scales of a century and longer. These coastal barriers are responsible for creating the APNEP estuarine system, and a major breach in the integrity would ultimately convert the estuary into a coastal ocean embayment (Riggs and Ames, 2003). Current management employs beach nourishment to fortify the barrier, but this method will become increasingly expensive as sea level rises substantially, and thus would be politically infeasible. Construction of a seawall along the entire extent of the barrier complex also does not appear to be a viable option because of financial costs and loss of the beach that defines and enriches the Outer Banks.

Special opportunities for implementation of adaptive management in APNEP include the existence of the CHPP process, a legislatively mandated ecosystem-based management plan for preserving and enhancing coastal fisheries. This plan involves collaborative attentions by all necessary state agencies and thereby can overcome the historic constraints of compartmentalization of management authorities. This plan sets an admirable example for other states. Similarly, the novel state commission on effects of climate change that was legislated in 2005 also provides opportunity for education and participation of legislators in a process of looking forward, well beyond the usual time frames of politics, to serve as an example of proactivity for other states to emulate. Sparse human populations and low levels of development along much of the interior mainland shoreline of the APNEP complex provide opportunities for implementation of policies that protect the ability of the salt marsh and other shallow-water estuarine habitats to be allowed to retreat as sea level rises. Implementing the policies required to achieve this management adaptation would not be possible in places where development and infrastructure are so dense that the economic and social costs of shoreline retreat are high. Special funding to support purchase of rolling easements or other implementation methods can come from the Clean Water Management Trust Fund and the Ecosystem Enhancement Program of North Carolina, two facilitators of large coordinated projects. The State of North Carolina was among the first to establish basin-scale water quality management and has established novel methods of basin-wide capping of nutrient delivery to estuaries, such the Neuse River Estuary, involving ecosystem-based management through participation of all stakeholders. This too facilitates actions required to manage consequences of climate change to preserve management goals of a national estuary.

7.5 Conclusions

7.5.1 Management Response

(1) Maintaining the status quo in management of estuarine ecosystems would result in substantial losses of ecosystem services as climate change progresses.

(2) In the absence of effective management adaptation, climate-related failures will appear in all of the most important management goals identified in the CCMPs of national estuaries: maintaining water quality, sustaining fish and wildlife populations, preserving habitat, protecting human values and services, and fulfilling water quantity needs.

(3) Avoiding negative impacts in estuaries to either public trust or private property values on shore could only be achieved by management at the global scale by capping greenhouse gas emissions, a solution that, if accomplished today, would not prevent decades of change because of past emissions. Consequently, impacts of climate change and sea level rise, in particular, are inevitable. As an example, climate change impacts on sea level are already evident in the growing demand for and costs of beach nourishment.

(4) Many of the anticipated consequences of climate change occur via mechanisms involving interactions among stressors and therefore may not be widely appreciated by policy makers, managers, stakeholders, and the public.

(5) Among the consequences of climate change that threaten estuarine ecosystem services, the most serious involve interactions between climate-dependent processes and human responses to climate change. In particular, conflicts arise between sustaining public trust values and private property in that current policies protecting private shoreline property become increasingly injurious to public trust values as climate changes and sea level rises further.

(6) Many management adaptations to climate change to preserve estuarine services can be achieved at all levels of government at modest cost. One major form of adaptation involves recognition of the projected consequences of sea level rise and then application of policies that create buffers to anticipate associated consequences. An important example would be redefining riverine flood hazard zones to match the projected expansion of flooding frequency and extent.

(7) Other management adaptations can be designed to build resilience of ecological and social systems. These adaptations include choosing only those sites for habitat restoration that allow natural recession landward and thus provide resilience to sea level rise.

(8) Management adaptations to climate change can occur on three different time scales: (a) reactive measures taken in response to observed negative impacts; (b) immediate development of plans for management adaptation to be implemented later, either when an indicator signals that delay can occur no longer, or in the wake of a disastrous consequence that provides a window of socially feasible opportunity; or (c) immediate implementation of proactive policies. The factors determining which of these time frames is appropriate for any given management adaptation include balancing costs of implementation with the magnitude of risks of injurious consequences under the status quo of management; the degree of reversibility of negative consequences of climate change; recognition and understanding of the problem by managers and the public; the uncertainty associated with the projected consequences of climate change; the time table on which change is anticipated; and the extent of political, institutional, and financial impediments.

(9) A critical goal of monitoring is to establish and follow indicators that signal approach towards an ecosystem threshold that—once passed—implies passage of the system into an alternative state from which conversion back is difficult. Avoiding conversion into such alternative states, often maintained by positive feedbacks, is one major motivation for implementing proactive management adaptation. That is especially critical if the transition is irreversible or very difficult and costly to reverse, and if the altered state delivers dramatically fewer ecosystem services. One example of such ecosystem conversions involves nitrogen-induced conversion from an estuary dominated by submersed benthic grasses to an alternative dominated by seaweeds and planktonic

microalgae. Such work to establish important environmental indicators is already being done in national estuaries and can be used to monitor climate change impacts.

(10) One critically important management challenge is to implement actions to achieve orderly retreat of development from shorelines at high risk of erosion and flooding and to preclude development of undeveloped shorelines at high risk. Such proactive management actions have been inhibited in the past by: (a) uncertainty over or denial of climate change and its implications; (b) failures to include true economic, social, and environmental costs of present policies allowing and subsidizing such risky development; and (c) legal tenets of private property rights. One possible proactive management option would be to establish and enforce “rolling easements” along estuarine shorelines as sea level continues to rise, thereby sustaining the public ownership of tide lands.

(11) Management adaptation to climate change may include ending public subsidies that now support risky development on coastal barrier and estuarine shores at high risk of flooding and storm damage as sea level rises further and intense storms are more common. Although the flood insurance system as a whole may be actuarially sound, current statutes provide people along the water’s edge in eroding areas of highest risk with artificially low rates, subsidized by the flood insurance policies of people in relatively safe areas. Ending such subsidization of high risk developments would represent a form of management adaptation to sea level rise. The federal Coastal Barriers Resources Act provides some guidance for eliminating such subsidies for public infrastructure and private development, although this act applies only to a list of undeveloped coastal barriers and would require extension to all barriers and to estuarine shorelines to enhance its effectiveness as an adaptation to climate change.

(12) Building upon ongoing efforts to operationalize EBM for oceans, analogous research is required for estuarine ecosystems. This research needs to address a major intrinsic impediment to EBM of estuarine services, which is the absence of a synthetic governance structure that unites now disparate management authorities, stakeholders, and the public. The U.S. Commission on Ocean Policy appealed for just this type of modification of governance structure to serve to implement EBM. EBM is necessary to facilitate management of interacting stressors, an almost ubiquitous condition for estuaries, because under present governance schemes management authority is partitioned among separate agencies or entities. Although national estuaries lack regulatory authority, they do unite most, if not all, stakeholders and could conceivably be reconstructed as quite different entities to develop and implement ecosystem-based management. Such coordination among diverse management authorities must involve land managers in order to incorporate a major source of inputs to estuaries.

(13) Using the Albemarle-Pamlico National Estuarine Program as a case study illustrates several management challenges posed by changing climate. Risks of rising sea level together with increases in intense storms pose a serious threat to the integrity of the Outer Banks and thus to the character of the Albemarle and Pamlico Sounds, which are now sheltered and brackish, possessing little astronomical tide. A state analog to ecosystem-based management, the Coastal Habitat Protection Plan, unifies state agencies to provide

synthetic protection for fish habitats. This provides a model on which to base further development and application of estuarine ecosystem-based management. The Legislature of the State of North Carolina established a study commission to report on the consequences of climate change and to make recommendations for management responses. This procedure too can form a model for other states and the federal government through the NEP. Although the Albemarle-Pamlico National Estuary is among those most sensitive to climate change and has an active management planning process in place, the absence of explicit adaptive management consideration in its CCMP reflects a need for attention to this issue by NEPs.

(14) Contemplate pursuit of a Federal Executive Order on climate change analogous to the Environmental Justice Executive Order to increase awareness of the potential for catastrophe on our coasts. This could include requirements for substantive rather than superficial evaluations of climate change impact in NEPA.

(15) Include climate change sensitivity, resilience, and adaptation responses as priorities on all relevant funding programs at state and federal levels. In the absence of such actions, for example, climate impacts on estuarine wetlands will likely violate the national “no-net-loss of wetlands” policy, which underwrites the current application of the Clean Water Act, in two ways: (a) wetland loss due to climate will increasingly compound the continuing loss of wetlands due to development and inadequate mitigation; and; (b) measures used to protect human infrastructure from climate impacts will prevent wetland adaptation to climate change.

(16) Review all federal and state environmental programs to assess whether projected consequences of climate change have been adequately considered and whether adaptive management needs to be inserted to achieve programmatic goals. For example, Jimerfield *et al.* (2007) conclude that “There clearly needs to be [a] comprehensive approach by federal agencies and cooperating scientists to address climate change in the endangered species recovery context. The current weak and piece-meal approach will waste precious resources and not solve the problem we are facing.”

7.5.2 Research Priorities

7.5.2.1 Conceptual Gaps in Understanding

(1) There is urgent need for further study of factors affecting sea level rise that may be significant, but now remain so uncertain that they cannot yet be included in IPCC projections. This especially includes enhancing our understanding of processes and rates of melting of Antarctic and Greenland ice sheets as a function of changing temperature and other coupled climatic conditions. Furthermore, it is important to resolve uncertainties about the fate of water in liquid phase released from the Greenland ice sheet, which involves the ability to project how land surface levels will respond to release from the weight of ice cover.

(2) Our understanding of processes affecting elevation change in land masses needs to be enhanced generally so that risk of flooding, shoreline erosion, and storm damage can be

better based upon geography-specific predictions of change in relative sea level, which combines rate of eustatic sea level change with land subsidence or emergence rate.

(3) Establish quantitative monitoring and research in some model estuarine systems to develop mechanistic understanding of changes projected as consequences of climate change. Many climate change drivers (*e.g.*, CO_2 concentration, ocean temperature at the surface and with depth, sea level) are currently monitored. However, projected consequences (*e.g.*, shoreline erosion rates; estuarine physical circulation patterns; water column stratification and extent of hypoxia; species range extensions and subsequent consequences of interactions within these new combinations of predators, prey, and competitors; the incidence and impacts of disease and parasitism) require new targeted monitoring and research efforts to fill the many conceptual gaps in our understanding of these processes.

(4) Integrated, landscape-scale numerical modeling will have to become a fundamental tool to predict potential estuarine responses to the complex and often interacting stressors induced by climate change. For instance, in most cases significantly modified hydrology and sediment transport predictions will need to be linked at the estuarine interface to sea level and storm (wind/wave regime) predictions in order to evaluate the interactive effects on sediment accretion and erosion effects in estuarine marshes. Models will have to take into account complex aspects such as changes in contribution of snowmelt and rain-on-snow to timing, magnitude and hydroperiod of river discharges (*e.g.*, Mote, 2006), changes in storm tracks (*e.g.*, Salathé, 2006), and changes in sediment loading to and circulation within estuaries, and how river management and regulation will be a factor (Sanchez-Arcilla and Jimenez, 1997) Ultimately, these models will need to be tied to coastal management models and other tools that allow assessment of both climate change and human response and infrastructure response.

(5) Research is needed on alternative implementation mechanisms, costs, and feasibility of achieving some form of coastal realignment, probably involving rolling easements. This would include legal, social, and cultural considerations in alternative methods of resolving or minimizing conflicts between public trust and private property values in context of building resilience to climate change by requiring rolling easements for development in now largely undeveloped waterfront and riparian areas at risk of flooding, erosion, and storm damage.

7.5.2.2 Data Gaps

There is great need for socioeconomic research and monitoring of how social and economic variables and systems are changing and likely to change further in coastal regions as sea level rises. This includes developing better information on economic, social, and environmental costs of estuarine-relevant management policies under global climate change. Economic and social impacts of the growing abandonment of risky coastal areas by property insurers and the possible future challenges in finding mortgage loans in such regions may be important inputs into decisions on regulating development and redevelopment of such areas.

7.5.2.3 Governance Issues

(1) As stated in Management Response recommendation 12 above, a synthetic governance structure that unites now disparate management authorities, stakeholders and the public may be needed to address major impediments to EBM of estuarine services. NEPs could be restructured to develop and implement ecosystem-based management.

(2) EBM of estuaries involves at minimum an approach that considers the entire drainage basin. Management plans to control estuarine water quality parameters sensitive to eutrophication, for example, must take a basin-wide approach to develop understanding of how nutrient loading at all positions along the watershed is transferred downstream to the estuary. Basin-scale management by its very nature thus prospers from uniting local governments across the entire watershed to develop partnerships to coordinate rule development and implementation strategies. Often trading programs are available that allow economies to be realized in achieving management goals. To this end of facilitating management adaptation to climate change, new ecologically based partnerships of local governments could be promoted and supported.

7.5.2.4 Tool Needs

(1) New and enhanced research funds need to be invested in development and implementation of estuarine observing systems that are currently in a planning stage, such as NEON, ORION, US IOOS, and others. Fully integrate these observing systems with global coastal observing programs and the Global Earth Observation System. Whereas physical and chemical parameters lend themselves to automated monitoring by remote sensing and observing system platforms, more basic technological research is also necessary to allow monitoring of key biological variables as part of these observing systems. Furthermore, it is critical that current efforts to develop monitoring systems in coastal ocean waters be brought into estuaries and up into their watersheds, where the largest human populations concentrate and where ecosystem values are most imperiled.

(2) New, more complete, interdisciplinary models are needed projecting social, economic, and cultural consequences of alternative management scenarios under projected consequences of climate change. These models include decision tools that are accessible by and applicable to managers and policy makers at all levels of government.

(3) New tools are required to enhance local capacity for developing and implementing management adaptations in response to climate change.

(4) New tools are not enough: older well-accepted tools must be used more effectively. Government agencies responsible for monitoring the environment have been reducing their commitment to this mission because of funding cuts. Extending historical records of environmental conditions is now even more urgent as a means of detecting climate change.

7.5.2.5 Education

(1) Urgent need exists to inform policy makers, managers, stakeholders, and the public about the specific evidence of climate change and its predicted consequences on estuaries. Re-education of some audiences may require additional effort and media tools to combat past and future disinformation campaigns that create confusion. Education on the scale necessary will require new funding and educational initiatives. Effective efforts must involve diverse suites of educational media including information delivery on evolving platforms such as the internet and cell phones. The information cannot reach far enough or rapidly enough if restricted to traditional delivery in school curricula and classes, but must propagate through churches, civic organizations, and entertainment media. Such education is particularly challenging and requires creative approaches.

(2) One goal of education about implications of climate change for estuaries is to build capacity for local citizen involvement in decision making. This is particularly important because of the dramatic changes required to move from management-as-usual to adaptive management. Especially challenging is the process of reconsideration of developing and redeveloping shorelines at risk of flooding, erosion, and storm damage.

(3) Some countries and states provide periodic assessments of the state of their environment. Monitoring data from many National Estuary Programs often now serve this goal when placed in a sufficiently long time frame that extends back before establishment of the NEP program. Similar scoreboards relating the status of stressors associated with climate change and of the consequences of climate change might be valuable additions to websites for all national estuaries and for our country’s estuaries more broadly. To illustrate these aspects of climate change, longer-term records are required than those typically found in state of environment reports. One simple example would be provision of empirical data on sea level from local recording stations. Similarly, maps of historical shoreline movement would provide the public with a visual indication of site-specific risks. Historical hurricane tracks are similarly informative and compelling.

7.6 Appendix

7.6.1 Federal Legislation for Protection and Restoration of Estuaries

LEGISLATION	AS IT PERTAINS TO ESTUARIES	Link
Clean Water Act (1972, 1977, 1981, 1987)	Authorizes EPA to implement pollution control programs; established the basic structure for regulating discharges of pollutants and requirements to set water quality standards for all contaminants in surface waters.	http://www.epa.gov/region5/water/cwa.htm
• Sec. 320 National Estuary Program (1987)	Authorizes EPA to develop plans for improving or maintaining water quality in estuaries of national significance including both point and nonpoint sources of pollution.	http://www.epa.gov/owow/estuaries/
• Sec. 404. Permits for Dredged or Fill Materials (1987)	Authorizes the Corps of Engineers (U.S. Army) to issue permits for the discharge of dredged or fill material into the navigable waters at specified disposal sites.	http://www.epa.gov/owow/wetlands/
• SEC. 601 State Water Pollution Control Revolving Funds (1987)	Authorizes EPA to capitalize state grants for water pollution control revolving funds for (1) for construction of public treatment facilities (2) for management program under section 319 (nonpoint source), and (3) for conservation and management plans under section 320 (NEP).	http://www.epa.gov/owm/cwfinance/
Coastal Zone Management Act (1972)	Provides grants to states that develop and implement Federally approved coastal zone management plans; allows states with approved plans the right to review Federal actions; authorizes the National Estuarine Research Reserve System.	http://www.legislative.noaa.gov/Legislation/czma.html

LEGISLATION	AS IT PERTAINS TO ESTUARIES	Link
National Environmental Policy Act (NEPA) (1969)	Establishes national environmental policy for the protection, maintenance, and enhancement of the environment; integrates environmental values into decision making processes; requires federal agencies to integrate environmental values into their decision making processes by considering the environmental impacts of their proposed actions and reasonable alternatives to those actions.	http://www.epa.gov/compliance/nepa/
Magnuson-Stevens Fishery Conservation and Management Act (1996, amended)	Provides for the conservation and management of the fishery resources; ensures conservation; facilitates long-term protection of essential fish habitats; recognizes that one of the greatest long-term threats to the viability of fisheries is the continuing loss of marine, estuarine, and other aquatic habitats; promotes increased attention to habitat considerations.	http://www.nmfs.noaa.gov/sfa/
Endangered Species Act (1973)	Provides a means for ecosystems, upon which endangered species and threatened species depend, to be conserved; applicants for permits for activities that might harm endangered species must develop a Habitat Conservation Plan (HCP), designed to offset any harmful effects of the proposed activity.	http://www.fws.gov/Endangered/
National Flood Insurance Program (1968)	Component of FEMA that makes federally backed flood insurance available to homeowners, renters, and business owners in ~20,000 communities who voluntarily adopt floodplain management ordinances to restrict development in areas subject to flooding, storm surge or coastal erosion; identifies and maps the Nation’s floodplains.	http://www.fema.gov/business/nfip/

LEGISLATION	AS IT PERTAINS TO ESTUARIES	Link
Nonindigenous Aquatic Nuisance Prevention and Control Act (1990)	Provides means to prevent and control infestations of the coastal inland waters of the United States by nonindigenous aquatic nuisance species, control of ballast water and allows for development of voluntary State Aquatic Nuisance Species Management Plans.	http://nas.er.usgs.gov/links/control.asp
Coastal Barrier Resources Act (CBRA) (1982)	Designates various undeveloped coastal barrier islands for inclusion in the Coastal Barrier Resources System (System). Areas so designated are made ineligible for direct or indirect Federal financial assistance that might support development, including flood insurance, except for emergency life-saving activities.	http://www.fws.gov/habitatconservation/coastal_barrier.htm

7.7 References

Albemarle-Pamlico National Estuary Program, 1994: *Albemarle-Pamlico NEP Comprehensive Conservation and Management Plan.*

Albemarle-Pamlico National Estuary Program, 2007: Albemarle-Pamlico Sounds region. Albemarle-Pamlico National Estuary Program Website, http://www.apnep.org/pages/regions.html, accessed on 7-25-2007.

Alber, M. and J.E. Sheldon, 1999: Use of a date-specific method to examine variability in the flushing times of Georgia estuaries. *Estuarine Coastal and Shelf Science*, **49(4)**, 469-482.

Alber, M., 2002: A conceptual model of estuarine freshwater inflow management. *Estuaries*, **25(6)**, 1246-1261.

Anderson, T.H. and G.T. Taylor, 2001: Nutrient pulses, plankton blooms, and seasonal hypoxia in western Long Island Sound. *Estuaries*, **24(2)**, 228-243.

Arora, V.K., F.H.S. Chiew, and R.B. Grayson, 2000: The use of river runoff to test CSIRO 9 land surface scheme in the Amazon and Mississippi River Basins. *International Journal of Climatology*, **20(10)**, 1077-1096.

Baird, D., R.R. Christian, C.H. Peterson, and G.A. Johnson, 2004: Consequences of hypoxia on estuarine ecosystem function: energy diversion from consumers to microbes. *Ecological Applications*, **14(3)**, 805-822.

Barry, J.P., C.H. Baxter, R.D. Sagarin, and S.E. Gilman, 1995: Climate-related, long-term faunal changes in a California Rocky intertidal community. *Science*, **267(5198)**, 672-675.

Bearden, D.M., 2001: *National Estuary Program: a Collaborative Approach to Protecting Coastal Water Quality*. CRS Report for Congress #97-644, Congressional Research Service.

Beaugrand, G., P.C. Reid, F. Ibanez, J.A. Lindley, and M. Edwards, 2002: Reorganization of North Atlantic Marine Copepod biodiversity and climate. *Science*, **296**, 1692-1694.

Boesch, D.F., 1994: *Scientific Assessment of Coastal Wetland Loss, Restoration and Management in Louisiana*. Coastal Education and Research Foundation, pp. 1-103.

Boicourt, W.C., 1992: *Influences of Circulation Processes on Dissolved Oxygen in the Chesapeake Bay*. UM-SG-TS-92-01, University of Maryland Sea Grant College Publications, College Park, Maryland, pp.1-234.

Borsuk, M.E., D. Higdon, C.A. Stow, and K.H. Reckhow, 2001: A Bayesian hierarchical model to predict benthic oxygen demand from organic matter loading in estuaries and coastal zones. *Ecological Modelling*, **143(3)**, 165-181.

Boynton, W.R., J.H. Garber, R. Summers, and W.M. Kemp, 1995: Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries*, **18(1)**, 285-314.

Bozek, C.M. and D.M. Burdick, 2005: Impacts of seawalls on saltmarsh plant communities in the Great Bay Estuary, New Hampshire USA. *Wetlands Ecology and Management*, **13(5)**, 553-568.

Breitburg, D., S. Seitzinger, and J. Sanders, 1999: *The Effects of Multiple Stressors on Freshwater and Marine Ecosystems*. American Society of Limnology and Oceanography.

Breitburg, D.L., C. A. Baxter, R. W. Hatfield, R. W. Howarth, C. G. Jones, G. M. Lovett, and C. Wigand, 1998: Understanding effects of multiple stressors: ideas and challenges, In: *Successes, Limitations, and Frontiers in Ecosystem Science*, [Pace, M.L. and P.M. Groffman (eds.)]. Springer, New York, pp. 416-431.

Breitburg, D.L., L.D. Coen, M.W. Luckenbach, R. Mann, M. Posey, and J.A. Wesson, 2000: Oyster reef restoration: convergence of harvest and conservation strategies. *Journal of Shellfish Research*, **19(1)**, 371-377.

Breitburg, D.L. and G. F. Riedel, 2005: Multiple stressors in marine systems, In: *Marine Conservation Biology: the Science of Maintaining the Sea's Biodiversity*, [Norse, E. and L.B. Crowder (eds.)]. Marine Conservation Biology Institute.

Breitburg, D.L., J.G. Sanders, C.C. Gilmour, C.A. Hatfield, R.W. Osman, G.F. Riedel, S.P. Seitzinger, and K.G. Sellner, 1999: Variability in responses to nutrients and trace elements, and transmission of stressor effects through an estuarine food web. *Limnology and Oceanography*, **44(3)**, 837-863.

Bricker, S.B., C.G. Clement, D.E. Pirhalla, S.P. Orlando, and D.R.G. Farrow, 1999: *National Estuarine Eutrophication Assessment: Effects of Nutrient Enrichment in the Nation's Estuaries*. National Centers for Coastal Ocean Science, National Oceanic and Atmospheric Administration, Silver Spring, MD, pp. 1-71.

Brinson, M.M., 1991: *Ecology of a Nontidal Brackish Marsh in Coastal North Carolina*. U. S. Fish and Wildlife Service, National Wetlands Research Center, Slidell, Louisiana.

Brinson, M.M., 1993: *A Hydrogeomorphic Classification for Wetlands*. Technical Report WRP-DE-4, US Army Engineer Waterways Experiment Station; Available from National Technical Information Service, Vicksburg, Mississippi.

Brinson, M.M. and R.R. Christian, 1999: Stability and response of *Juncus roemerianus* patches in a salt marsh. *Wetlands*, **19(1)**, 65-70.

Brinson, M.M., R.R. Christian, and L.K. Blum, 1995: Multiple states in the sea-level induced transition from terrestrial forest to estuary. *Estuaries*, **18(4)**, 648-659.

Brown, J.J., 1993: The State and Indian nations' water resource planning. *Occasional Paper*, **19**.

Brown, J.J., 1994: Treaty rights: twenty years after the Boldt decision. *Wicazo Sa Review*, **10(2)**, 1-16.

Bruno, J.F., K.E. Boyer, J.E. Duffy, S.C. Lee, and J.S. Kertesz, 2005: Effects of macroalgal species identity and richness on primary production in benthic marine communities. *Ecology Letters*, **8(11)**, 1165-1174.

Bruno, J.F., C.W. Kennedy, T.A. Rand, and M.B. Grant, 2004: Landscape-scale patterns of biological invasions in shoreline plant communities. *Oikos*, **107(3)**, 531-540.

Burkett, V., D. Wilcox, R. Stottlemyer, W. Barrow, D. Fagre, J. Baron, J. Price, J.L. Nielsen, C.D. Allen, D.L. Peterson, G. Ruggerone, and T. Doyle, 2005: Nonlinear dynamics in ecosystem response to climatic change: case studies and policy implications. *Ecological Complexity*, **2(4)**, 357-394.

Burkholder, J.M., E.J. Noga, C.H. Hobbs, and H.B. Glasgow Jr, 1992: New 'phantom' dinoflagellate is the causative agent of major estuarine fish kills. *Nature*, **358(6385)**, 407-410.

Buzzelli, C.P., R.A. Luettich Jr, S.P. Powers, C.H. Peterson, J.E. McNinch, J.L. Pinckney, and H.W. Paerl, 2002: Estimating the spatial extent of bottom-water hypoxia and habitat degradation in a shallow estuary. *Marine Ecology Progress Series*, **230**, 103-112.

Caldeira, K. and M.E. Wickett, 2003: Anthropogenic carbon and ocean pH. *Nature*, **425(6956)**, 365-365.

Callaway, J.C., J.A. Nyman, and R.D. DeLaune, 1996: Sediment accretion in coastal wetlands: a review and a simulation model of processes. *Current Topics in Wetland Biogeochemistry*, **2**, 2-23.

Carpenter, S., B. Walker, J.M. Anderies, and N. Abel, 2001: From metaphor to neasurement: resilience of what to what? *Ecosystems*, **4(8)**, 765-781.

Carpenter, S.R. and O. Kinne, 2003: *Regime Shifts in Lake Ecosystems: Pattern and Variation*. International Ecology Institute, Luhe, Germany.

Carpenter, S.R., D. Ludwig, and W.A. Brock, 1999: Management of eutrophication for lakes subject to potentially irreversible change. *Ecological Applications*, **9(3)**, 751-771.

Carson, R., 1962: *Silent Spring*. Houghton Mifflin.

Chmura, G.L., S.C. Anisfeld, D.R. Cahoon, and J.C. Lynch, 2003: Global carbon sequestration in tidal, saline wetland soils. *Global Biogeochemical Cycles*, **17(4)**.

Christensen, N.L., A.M. Bartuska, J.H. Brown, S. Carpenter, C. D'Antonio, R. Francis, J.F. Franklin, J.A. MacMahon, R.F. Noss, D.J. Parsons, C.H. Peterson, M.G. Turner, and R.G. Woodmansee, 1996: The report of the Ecological Society of America Committee on the scientific basis for ecosystem management. *Ecological Applications*, **6(3)**, 665-691.

Christian, R.R., L. Stasavich, C. Thomas, and M. M. Brinson, 2000: Reference is a moving target in sea-level controlled wetlands, In: *Concepts and Controversies in Tidal Marsh Ecology*, [Weinstein, M.P. and D.A. Kreeger (eds.)]. Kluwer Press, The Netherlands, pp. 805-825.

Church, J.A., 2001: How fast are sea levels rising? *Science*, **294**, 802-803.

Climate Impacts Group, University of Washington, 2007: Climate change. University of Washington, http://www.cses.washington.edu/cig/pnwc/cc.shtml, accessed on 3-23-2007.

Cloern, J.E., A.E. Alpine, B.E. Cole, R.L.J. Wong, J.F. Arthur, and M.D. Ball, 1983: River discharge controls phytoplankton dynamics in the Northern San Francisco Bay Estuary. *Estuarine Coastal and Shelf Science*, **16(4)**.

Coastal Protection and Restoration Authority of Louisiana, 2007: *Louisiana's Comprehensive Master Plan for a Sustainable Coast*. Coastal Protection and Restoration Authority of Louisiana.

Coen, L.D., M. W. Luckenbach, and D. L. Breitburg, 1999: The role of oyster reefs as essential fish habitat: a review of current knowledge and some new perspectives, [Benaka, L.R. (ed.)]. Bethesda, Maryland, pp. 438-454.

Committee on Mitigating Shore Erosion along Sheltered Coasts, National Research Council, 2006: *Mitigating Shore Erosion Along Sheltered Coasts*. pp.1-188.

Committee on Mitigating Wetland Losses, National Research Council, 2001: *Compensating for Impacts Under the Clean Water Act*. National Academies Press, Washington, DC.

Conley, D.J., S. Markager, J. Andersen, T. Ellermann, and L.M. Svendsen, 2002: Coastal eutrophication and the Danish national aquatic monitoring and assessment program. *Estuaries*, **25(4)**, 848-861.

Cooper, S.R. and G.S. Brush, 1993: A 2,500-year history of anoxia and eutrophication in Chesapeake Bay. *Estuaries*, **16**, 617-626.

Cooper, S.R., S.K. McGlothlin, M. Madritch, and D.L. Jones, 2004: Paleoecological evidence of human impacts on the Neuse and Pamlico Estuaries of North Carolina, USA. *Estuaries*, **27(4)**, 617-633.

Copeland, B.J., 1966: Effects of decreased river flow on estuarine ecology. *Journal Water Pollution Control Federation*, **38**, 1831-1839.

Copeland, B.J. and J.E. Hobbie, 1972: *Phosphorus and Eutrophication in the Pamlico River Estuary, N. C., 1966-1969- A SUMMARY*. 1972-65, University of North Carolina Water Resources Research Institute, Raleigh, North Carolina.

Costello, J.H., B.K. Sullivan, and D.J. Gifford, 2006: A physical-biological interaction underlying variable phenological responses to climate change by coastal zooplankton. *Journal of Plankton Research*, **28(11)**, 1099-1105.

Cropper, C.R., 2005: The study of endocrine-disrupting compounds: past approaches and new directions. *Integrative and Comparative Biology*, **45**, 194-2000.

Curtis, P.S., L.M. Balduman, B.G. Drake, and D.F. Whigham, 1990: Elevated Atmospheric CO_2 Effects on Belowground Processes in C3 and C4 Estuarine Marsh Communities. *Ecology*, **71(5)**, 2001-2006.

Dacey, J.W.H., B.G. Drake, and M.J. Klug, 1994: Stimulation of methane emission by carbon dioxide enrichment of marsh vegetation. *Nature*, **370(6484)**, 47-49.

Dakora, F. and B.G. Drake, 2000: Elevated CO_2 stimulates associative N2 fixation in a C3 plant of the Chesapeake Bay wetland. *Plant, Cell and Environment*, **23(943)**, 953.

Davis, M.B., 1983: Holocene vegetational history of the eastern United States, [Wright, H.E., Jr. (ed.)]. University of Minnesota Press, Minneapolis, MN, pp. 166-181.

Day, J.W., Jr., D.F. Boesch, E.J. Clairain, G.P. Kemp, S.B. Laska, W.J. Mitsch, K. Orth, H. Mashriqui, D.J. Reed, L. Shabman, C.A. Simenstad, B.J. Streever, R.R. Twilley, C.C. Watson, J.T. Wells, and D.F. Whigham, 2007: Restoration of the Mississippi Delta: lessons from Hurricanes Katrina and Rita. *Science*, **315(5819)**, 1679-1684.

Day, J.W., Jr., C.A.S. Hall, W.M. Kemp, and A. Yanez-Arancibia, 1989: *Estuarine Ecology*. Wiley and Sons, New York, NY.

Dayton, P.K., S. Thrush, and F.C. Coleman, 2002: *Ecological Effects of Fishing in Marine Ecosystems of the United States*. Pew Oceans Commission.

Dennison, W.C., R.J. Orth, K.A. Moore, J.C. Stevenson, V. Carter, S. Kollar, P.W. Bergstrom, and R.A. Batiuk, 1993: Assessing water quality with submersed aquatic vegetation. *BioScience*, **43(2)**, 86-94.

Department for Environment, Food and Rural Affairs (DEFRA) and Department of the Environment, 2002: *Managed Realignment Review - Project Report*. Policy Research Project FD 2008, DEFRA, Cambridge, UK.

Diaz, R.J. and R. Rosenberg, 1995: Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology Annual Review*, **33**, 245-303.

Dobson, A. and J. Foufopoulos, 2001: Emerging infectious pathogens of wildlife. *Philosophical Transactions: Biological Sciences*, **356(1411)**, 1001-1012.

Donnelly, J.P. and M.D. Bertness, 2001: Rapid shoreward encroachment of salt marsh cordgrass in response to accelerated sea-level rise. *Proceedings of the National Academy of Sciences of the United States of America*, **98(25)**, 14218-14223.

Dowdeswell, J.A., 2006: Atmospheric science: the Greenland ice sheet and global sea-level rise. *Science*, **311(5763)**, 963-964.

Drake, B.G., L. Hughes, E. A. Johnson, B. A. Seibel, M. A. Cochrane, V. J. Fabry, D. Rasse, and L. Hannah, 2005: Synergistic Effects, In: *Climate Change and Biodiversity*, [Lovejoy, T.E. and L. Hannah (eds.)]. Yale University Press, New Haven, pp. 296-316.

Drake, B.G., M.S. Muehe, G. Peresta, M.A. Gonzalez-Meler, and R. Matamala, 1995: Acclimation of photosynthesis, respiration and ecosystem carbon flux of a wetland on Chesapeake Bay, Maryland to elevated atmospheric CO_2 concentration. *Plant and Soil*, **187(2)**, 111-118.

Duarte, C.M., 1991: Seagrass depth limits. *Aquatic Botany*, **40(4)**, 363-377.

Duffy, J.E., 2002: Biodiversity and ecosystem function: the consumer connection. *Oikos*, **99(2)**, 201-219.

Emanuel, K., 2005: Increasing destructiveness of tropical cyclones over the past 30 years. *Nature*, **436(7051)**, 686-688.

Fagherazzi, S., L. Carniello, L. D'Alpaos, and A. Defina, 2006: Critical bifurcation of shallow microtidal landforms in tidal flats and salt marshes. *Proceedings of the National Academy of Sciences of the United States of America*, **103(22)**, 8337-8341.

Federal Emergency Management Agency, 1991: *Projected Impact of Relative Sea Level Rise on the National Flood Insurance Program*. Washington, DC.

Federal Emergency Management Agency, 2007: Chapter 01 - description of study area. Comprehensive Hurricane Data Preparedness, FEMA Study Web Site, http://chps.sam.usace.army.mil/USHESDATA/NC/Data/chapter1/chapter01_description.html, accessed on 3-23-2007.

Feely, R.A., C.L. Sabine, K. Lee, W. Berelson, J. Kleypas, V.J. Fabry, and F.J. Millero, 2004: Impact of anthropogenic CO_2 on the $CaCO_3$ system in the oceans. *Science*, **305(5682)**, 362-366.

Foley, J.A., R. DeFries, G.P. Asner, C.C. Barford, G.B. Bonan, S.R. Carpenter, F.S. Chapin III, M.T. Coe, G.C. Daily, H.K. Gibbs, J.H. Helkowski, T. Holloway, E.A. Howard, C.J. Kucharik, C. Monfreda, J. Patz, I.C. Prentice, N. Ramankutty, and P.K. Snyder, 2005: Global consequences of land use. *Science*, **309(5734)**, 570-574.

Folke, C., S. Carpenter, B. Walker, M. Scheffer, T. Elmqvist, L.H. Gunderson, and C.S. Holling, 2004: Regime shifts, resilience, and biodiversity in ecosystem management. *Annual Review of Ecology and Systematics*, **35**, 557-581.

Fonseca, M.S., B.E. Julius, and W.J. Kenworthy, 2000: Integrating biology and economics in seagrass restoration: How much is enough and why? *Ecological Engineering*, **15(3)**, 227-237.

Ford, S.E., 1996: Range extension by the oyster parasite Perkinsus marinus into the northeastern United States: response to climate change? *Journal of Shellfish Research*, **15**, 45-56.

Galbraith, H., R. Jones, R. Park, J. Clough, S. Herrod-Julius, B. Harrington, and G. Page, 2002: Global climate change and sea level rise: potential losses of intertidal habitat for shorebirds. *Waterbirds*, **25(2)**, 173-183.

Gedney, N., P.M. Cox, R.A. Betts, O. Boucher, C. Huntingford, and P.A. Stott, 2006: Detection of a direct carbon dioxide effect in continental river runoff records. *Nature*, **439(7078)**, 835-838.

Giese, G.L., H.B. Wilder, and G.G. Parker, 1985: *Hydrology of Major Estuaries and Sounds of North Carolina*. USGS Water-Supply Paper 2221, USGS, pp.1-108.

González, J.L. and T.E. Törnqvist, 2006: Coastal Louisiana in crisis: subsidence or sea level rise? *Eos, Transactions American Geophysical Union*, **87(45)**, 493-498.

Greenberg, R., J.E. Maldonado, S. Droege, and M.V. McDonald, 2006: A global perspective on the evolution and conservation of their terrestrial vertebrates. *BioScience*, **56(8)**, 675-685.

Griffin, D.A. and P.H. LeBlond, 1990: Estuary/ocean exchange controlled by spring-neap tidal mixing. *Estuarine Coastal and Shelf Science*, **30(3)**, 275-297.

Groffman, P.M., J.S. Baron, T. Blett, A.J. Gold, I. Goodman, L.H. Gunderson, B.M. Levinson, M.A. Palmer, H.W. Paerl, G.D. Peterson, N.L. Poff, D.W. Rejesk, J. Reynolds, M.G. Turner, K.C. Weathers, and J. Wiens, 2006: Ecological thresholds: the key to successful environmental management or an important concept with no practical application? *Ecosystems*, **9(1)**, 1-13.

Grumbine, R.E., 1994: What is ecosystem management? *Conservation Biology*, **8(1)**, 27-38.

Guenette, S., T. Lauck, and C. Clark, 1998: Marine reserves: from Beverton and Holt to the present. *Reviews in Fish Biology and Fisheries*, **8(3)**, 251-272.

Gunderson, L.H., C. S. Holling, L. Pritchard, and G. D. Peterson, 2002: A summary and a synthesis of resilience in large scale systems, In: *Resilience and Behavior of Large-Scale Systems*, Island Press, Washington, DC, pp. 3-20.

H. John Heinz III Center for Science, Economics, and the Environment, 2002: *Human Links to Coastal Disasters*. Washington, DC.

Hagy, J.D., W.R. Boynton, C.W. Keefe, and K.V. Wood, 2004: Hypoxia in Chesapeake Bay, 1950-2001: long-term change in relation to nutrient loading and river flow. *Estuaries*, **27(4)**, 634-658.

Hakalahti, T., A. Karvonen, and E.T. Valtonen, 2006: Climate warming and disease risks in temperate regions Argulus coregoni and Diplostomum spathaceum as case studies. *Journal of Helminthology*, **80(2)**, 93-98.

Hall, S.R., A.J. Tessier, M.A. Duffy, M. Huebner, and C.E. Cβceres, 2006: Warmer does not have to mean sicker: temperature and predators can jointly drive timing of epidemics. *Ecology*, **87(7)**, 1684-1695.

Halpern, B.S., 2003: The impact of marine reserves: Do reserves work and does reserve size matter? *Ecological Applications*, **13(1)**, S117-S137.

Harley, C.D.G. and R. Hughes, 2006: Reviews and synthesis: the impacts of climate change in coastal marine systems. *Ecology Letters*, **9(2)**, 228-241.

Harris, L.D. and W. P. Cropper Jr., 1992: Between the devil and the deep blue sea: implications of climate change for Florida's fauna, In: *Global Warming and Biological Diversity*, [Peters, R.L. and T.E. Lovejoy (eds.)]. Yale University Press, New Haven, CT, pp. 309-324.

Harvell, C.D., C.E. Mitchell, J.R. Ward, S. Altizer, A.P. Dobson, R.S. Ostfeld, and M.D. Samuel, 2002: Climate warming and disease risks for terrestrial and marine biota. *Science*, **296(5576)**, 2158-2162.

Harvell, D., R. Aronson, N. Baron, J. Connell, A. Dobson, S. Ellner, L. Gerber, K. Kim, A. Kuris, and H. McCallum, 2004: The rising tide of ocean diseases: unsolved problems and research priorities. *Frontiers in Ecology and the Environment*, **2(7)**, 375-382.

Hauxwell, J., J. Cebrian, C. Furlong, and I. Valiela, 2001: Macroalgal canopies contribute to eelgrass (*Zostera marina*) decline in temperate estuarine ecosystems. *Ecology*, **82(4)**, 1007-1022.

Hayden, B.P., M.C.F.V. Santos, G. Shao, and R.C. Kochel, 1995: Geomorphological controls on coastal vegetation at the Virginia Coast Reserve. *Geomorphology*, **13**, 283-300.

Health Ecological and Economic Dimensions of Global Change Program, 1998: *Marine Ecosystems: Emerging Diseases As Indicators of Change. Health of the Oceans From Labrador to Venezuela*. Year of the ocean special report The Center for Conservation Medicine and CHGE Harvard Medical School, Boston, MA, pp.1-85.

Hobbie, J.E., B. J. Copeland, and W. G. Harrison, 1975: Sources and fates of nutrients in the Pamlico River estuary, North Carolina, In: *Chemistry, Biology and the Estuarine System*, [Cronin, L.E. (ed.)]. Academic Press, New York, NY, pp. 287-302.

Holling, C.S., 1972: Resilience and stability of ecological systems. *Research Report*, **4**, 1-23.

Hood, W.G., 2007: Personal communication. Skagit System Cooperative, La Conner, WA.

Howarth, R.W., J.R. Fruci, and D. Sherman, 1991: Inputs of sediment and carbon to an estuarine ecosystem: influence of land use. *Ecological Applications*, **1(1)**, 27-39.

Howarth, R.W., D.P. Swaney, T.J. Butler, and R. Marino, 2000: Climatic control on eutrophication of the Hudson River estuary. *Ecosystems*, **3(2)**, 210-215.

Hughes, A.R. and J.J. Stachowicz, 2004: Genetic diversity enhances the resistance of a seagrass ecosystem to disturbance. In: *Proceedings of the National Academy of Sciences of the United States of America* 2004.

IPCC, 2001: *Climate Change 2001: Impacts, Adaptation, and Vulnerability*. Cambridge Univeristy Press, Cambridge, UK.

IPCC, 2007: *Climate Change 2007: the Physical Science Basis. Summary for Policymakers*.

Jackson, J.B.C., M.X. Kirby, W.H. Berger, K.A. Bjorndal, L.W. Botsford, B.J. Bourque, R.H. Bradbury, R. Cooke, J. Erlandson, and J.A. Estes, 2001: Historical overfishing and the recent collapse of coastal ecosystems. *Science*, **293(5530)**, 629-638.

Jimerfield, S., M. Waage, and W. Snape, 2007: *Global Warming Threats and Conservation Actions in Endangered Species Recovery Plans: a Preliminary Analysis*. Center for Biological Diversity.

Jones, C.G., J.H. Lawton, and M. Shachak, 1994: Organisms as ecosystem engineers. *Oikos*, **69(3)**, 373-386.

Kates, R.W., C.E. Colten, S. Laska, and S.P. Leatherman, 2006: Reconstruction of New Orleans after Hurricane Katrina: a research perspective. *Proceedings of the National Academy of Sciences of the United States of America*, **103(40)**, 14653-14660.

Kemp, W.M., P.A. Sampou, J. Garber, J. Tuttle, and W.R. Boynton, 1992: Seasonal depletion of oxygen from bottom waters of Chesapeake Bay: roles of benthic and planktonic respiration and physical exchange processes. *Marine Ecology Progress Series MESEDT,* **85(1)**.

Kennedy, V.S., 1996: The ecological role of the Eastern oyster, *Crassostrea virginica*, with remarks on disease. *Journal of Shellfish Research*, **15**, 177-183.

Kennedy, V.S., R.R. Twilley, J.A. Kleypas, J.H. Cowan, Jr., and S.R. Hare, 2002: *Coastal and Marine Ecosystems & Global Climate Change: Potential Effects on U.S. Resources*. Pew Center on Global Climate Change, pp.1-64.

Kennish, M.J., 1999: *Estuary Restoration and Maintenance: the National Estuary Program*. CRC Press Inc.

Kleypas, J.A., R.A. Feely, V.J. Fabry, C. Langdon, C.L. Sabine, and L.L. Robbins, 2006: *Impacts of Ocean Acidification on Coral Reefs and Other Marine Calcifiers: a Guide for Future Research*. Workshop Report, National Science Foundation, National Oceanic and Atmospheric Administration, and the U.S. Geological Survey.

Korfmacher, K.S., 1998: Invisible successes, visible failures: paradoxes of ecosystem management in the Albemarle-Pamlico estuarine study. *Coastal Management*, **26(3)**, 191-212.

Korfmacher, K.S., 2002: Science and ecosystem management in the Albemarle-Pamlico Estuarine study. *Ocean & Coastal Management*, **45**, 277-300.

Kuenzler, E.J., P.J. Mulholland, L.A. Ruley, and R.P. Sniffen, 1977: *Water Quality in North Carolina Coastal Plain Streams and Effects of Channelization*. 127, Water Resources Research Institute of the University of North Carolina, Raleigh, NC.

Lafferty, K.D. and L.R. Gerber, 2002: Good medicine for conservation biology: the intersection of epidemiology and conservation theory. *Conservation Biology*, **16(3)**, 593-604.

Lafferty, K.D., J.W. Porter, and S.E. Ford, 2004: Are diseases increasing in the ocean? *Annual Review of Ecology, Evolution and Systematics*, **35**, 31-54.

Lee, K.N., 1993: *Compass and Gyroscope: Integrating Science and Politics for the Environment*. Island Press, Washington, DC.

Lenihan, H.S. and C.H. Peterson, 1998: How habitat degradation through fishery disturbance enhances impacts of hypoxia on oyster reefs. *Ecological Applications*, **8(1)**, 128-140.

Lenihan, H.S., C.H. Peterson, J.E. Byers, J.H. Grabowski, G.W. Thayer, and D.R. Colby, 2001: Cascading of habitat degradation: oyster reefs invaded by refugee fishes escaping stress. *Ecological Applications*, **11(3)**, 764-782.

Leonard, L., T. Clayton, and O. Pilkey, 1990: An analysis of replenished beach design parameters on U. S. East Coast barrier islands. *Journal of Coastal Research*, **6(1)**, 15-36.

Leung, L.Y.R. and Y. Qian, 2003: Changes in seasonal and extreme hydrologic conditions of the Georgia Basin/Puget Sound in an ensemble regional climate simulation for the mid-century. *Canadian Water Resources Journal*, **28(4)**, 605-631.

Levin, L.A., D.F. Boesch, A. Covich, C. Dahm, C. Erseus, K.C. Ewel, R.T. Kneib, A. Moldenke, M.A. Palmer, and P. Snelgrove, 2001: The function of marine critical transition zones and the importance of sediment biodiversity. *Ecosystems*, **4(5)**, 430-451.

Li, M., A. Gargett, and K. Denman, 2000: What determines seasonal and interannual variability of phytoplankton and zooplankton in strongly estuarine systems? *Estuarine Coastal and Shelf Science*, **50(4)**, 467-488.

Lotze, H.K., H.S. Lenihan, B.J. Bourque, R.H. Bradbury, R.G. Cooke, M.C. Kay, S.M. Kidwell, M.X. Kirby, C.H. Peterson, and J.B.C. Jackson, 2006: Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science*, **312(5781)**, 1806-1809.

Lyman, J.M., J.K. Willis, and G.C. Johnson, 2006: Recent cooling of the upper ocean. *Geophysical Research Letters*, **33**, L18604.

Mallin, M.A., J.M. Burkholder, L.B. Cahoon, and M.H. Posey, 2000: North and South Carolina coasts. *Marine Pollution Bulletin*, **41(1)**, 56-75.

Mallin, M.A., H.W. Paerl, J. Rudek, and P.W. Bates, 1993: Regulation of estuarine primary production by watershed rainfall and river flow. *Marine Ecology Progress Series*, **93(1/2)**, 1999-203.

Marks, D., J. Kimball, D. Tingey, and T. Link, 1998: The sensitivity of snowmelt processes to climate conditions and forest cover during rain-on-snow: a case study of the 1996 Pacific Northwest flood. *Hydrological Processes*, **12(10)**, 1569-1587.

Marsh, A.S., D.P. Rasse, B.G. Drake, and J.P. Megonigal, 2005: Effect of elevated CO_2 on carbon pools and fluxes in a brackish marsh. *Estuaries*, **28**, 695-704.

Meehl, G.A., W.M. Washington, W.D. Collins, J.M. Arblaster, A. Hu, L.E. Buja, W.G. Strand, and H. Teng, 2005: How much more global warming and sea level rise? *Science*, **307(5716)**, 1769-1772.

Meyer, D.L., E.C. Townsend, and G.W. Thayer, 1997: Stabilization and erosion control value of oyster cultch for intertidal marsh. *Restoration Ecology*, **5(1)**, 93-99.

Meyer, J.N. and R.T. Di Giulio, 2003: Heritable adaptation and fitness costs in killifish (*Fundulus heteroclitus*) inhabiting a polluted estuary. *Ecological Applications*, **13(2)**, 490-503.

Micheli, F., B.S. Halpern, L.W. Botsford, and R.R. Warner, 2004: Trajectories and correlates of community change in no-take marine reserves. *Ecological Applications*, **14(6)**, 1709-1723.

Micheli, F. and C.H. Peterson, 1999: Estuarine vegetated habitats as corridors for predator movements. *Conservation Biology*, **13(4)**, 869-881.

Mileti, D.S., 1999: *Disasters by Design: a Reassessment of Natural Hazards in the United States*. Joseph Henry Press.

Millennium Ecosystem Assessment, 2005: *Ecosystems and Human Well-Being: Wetlands and Water*. World Resources Institute, Washington, DC.

Mitsch, W.J. and J.W. Day Jr, 2006: Restoration of wetlands in the Mississippi-Ohio-Missouri (MOM) River Basin: experience and needed research. *Ecological Engineering*, **26**, 55-69.

Mitsch, W.J. and J.G. Gosselink, 2000: *Wetlands*. John Wiley, New York.

Mooney, H.A. and R.J. Hobbs, 2000: *Invasive Species in a Changing World*. Island Press, Washington, DC.

Moorhead, K.K. and M.M. Brinson, 1995: Response of wetlands to rising sea level in the lower coastal plain of North Carolina. *Ecological Applications*, **5(1)**, 261-271.

Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon, 2002: Responses of coastal wetlands to rising sea level. *Ecology*, **83(10)**, 2869-2877.

Mote, P.W., 2006: Climate-driven variability and trends in mountain snowpack in western North America. *Journal of Climate*, **19(23)**, 6209-6220.

Mote, P.W., E.A. Parson, A.F. Hamlet, W.S. Keeton, D. Lettenmaier, N. Mantua, E.L. Miles, D.W. Peterson, D.L. Peterson, R. Slaughter, and A.K. Snover, 2003: Preparing for climatic change: the water, salmon, and forests of the Pacific Northwest. *Climatic Change*, **61(1)**, 45-88.

Mullins, P.H. and T.C. Marks, 1987: Flowering phenology and seed production of Spartina anglica. *Journal of Ecology*, **25(4)**, 1037-1048.

Mydlarz, L.D., L.E. Jones, and C.D. Harvell, 2006: Innate immunity, environmental drivers, and disease ecology of marine and freshwater invertebrates. *Annual Review of Ecology, Evolution and Systematics*, **37**, 251-288.

Myers, R.A., J.K. Baum, T.D. Shapherd, S.P. Powers, and C.H. Peterson, 2007: Cascading effects of the loss of apex predatory sharks from a coastal ocean. *Science*, **315(5820)**, 1846-1850.

Naeem, S., 2002: Ecosystem consequences of biodiversity loss: The evolution of a paradigm. *Ecology*, **83(6)**, 1537-1552.

National Assessment Synthesis Team, 2000: *Climate Change Impacts on the United States: the Potential Consequences of Climate Variability and Change*. U.S. Global Change Research Program, Washington, DC.

National Coastal Assessment Group, 2000: *Coastal: The Potential Consequences of Climate Variability and Change*. pp.1-181.

National Marine Fisheries Service, 2006: *Recovery Plan for the Hawaiian Monk Seal*. National Marine Fisheries Service, Silver Spring, MD, pp.1-148.

National Ocean Service, 2005: Lionfish discovery story. NOAA Website, www.oceanservice.noaa.gov/education/stories/lionfish/lion03_blame.html, accessed on 7-25-2007.

National Research Council, 2004: *Non-Native Oysters in the Chesapeake Bay*. Committee on Non-native Oysters in the Chesapeake Bay, National Research Council, National Academies Press, Washington, DC.

New York State, 1992: Tidal wetlands - land use regulations. **6 NYCRR Part 661**.

New York State Department of Environmental Conservation, 2006: Hudson Valley climate change conference, December 4, 2006. New York State Department of Environmental Conservation, http://www.dec.state.ny.us/website/hudson/hvcc.html, accessed on 3-23-2007.

Newell, R.I.E., J.C. Cornwell, and M.S. Owens, 2002: Influence of simulated bivalve biodeposition and microphytobenthos on sediment nitrogen dynamics: a laboratory study. *Limnology and Oceanography*, **47(5)**, 1367-1379.

Newell, R.I.E. and J. A. Ott, 1999: Macrobenthic communities and eutrophication, In: *Ecosystems at the Land-Sea Margin: Drainage Basin to Coastal Sea*, [Malone, T.C., A. Malej, L. Harding, N. Smodlaka, and R. Turner (eds.)]. American Geophysical Union, Washington, DC, pp. 265-293.

Nixon, S.W., 1995: Coastal marine eutrophication: a definition, social causes, and future concerns. *Ophelia*, **41**, 199-219.

North Carolina Department of Environmental and Natural Resources, 2006: Stock status of important coastal fisheries in North Carolina. North Carolina Department of Environmental and Natural Resources, North Carolina Department of Environmental and Natural Resources, Division of Marine fisheries, http://www.ncfisheries.net/stocks/index.html, accessed on 3-23-2007.

Officer, C.B., R.B. Biggs, J.L. Taft, L.E. Cronin, M.A. Tyler, and W.R. Boynton, 1984: Chesapeake Bay anoxia: origin, development, and significance. *Science*, **223(4631)**, 22-26.

Orr, J.C., V.J. Fabry, O. Aumont, L. Bopp, S.C. Doney, R.A. Feely, A. Gnanadesikan, N. Gruber, A. Ishida, and F. Joos, 2005: Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature*, **437(7059)**, 681-686.

Orth, R.J., T.J.B. Carruthers, W.C. Dennison, C.M. Duarte, J.W. Fourqurean, K.L. Keck, Jr., A.R. Hughes, G.A. Kendrick, W.H. Kenworthy, S. Olyarnik, F.T. Short, M. Waycott, and S.L. Williams, 2006: A global crisis for seagrass ecosystems. *BioScience*, **56(12)**, 987-996.

Otto-Bliesner, B.L., S.J. Marshall, J.T. Overpeck, G.H. Miller, and A. Hu, 2006: Simulating arctic climate warmth and icefield retreat in the last interglaciation. *Science*, **311(5768)**, 1751-1753.

Overpeck, J.T., B.L. Otto-Bliesner, G.H. Miller, D.R. Muhs, R.B. Alley, and J.T. Kiehl, 2006: Paleoclimatic evidence for future ice-sheet instability and rapid sea-level rise. *Science*, **311(5768)**, 1747-1750.

Paerl, H.W., R.L. Dennis, and D.R. Whitall, 2002: Atmospheric deposition of nitrogen: Implications for nutrient overenrichment of coastal waters. *Estuaries*, **25**, 677-693.

Paerl, H.W. and J.D. Bales, 2001: Ecosystem impacts of three sequential hurricanes (Dennis, Floyd, and Irene) on the United States' largest lagoonal estuary, Pamlico Sound, NC. *Proceedings of the National Academy of Sciences of the United States of America*, **98(10)**, 5655-5660.

Paerl, H.W., J.L. Pinckney, J.M. Fear, and B.L. Peierls, 1998: Ecosystem responses to internal and watershed organic matter loading: consequences for hypoxia in the eutrophying Neuse River Estuary, North Carolina, USA. *Marine Ecology Progress Series*, **166**, 17-25.

Paerl, H.W., L.M. Valdes, A.R. Joyner, B.L. Peierls, M.F. Piehler, S.R. Riggs, R.R. Christian, L.A. Eby, L.B. Crowder, J.S. Ramus, E.J. Clesceri, C.P. Buzzelli, and R.A. Luettich, Jr., 2006: Ecological response to hurricane events in the Pamlico Sound System, North Carolina, and implications for assessment and management in a regime of increased frequency. *Estuaries and coasts*, **29(6A)**, 1033-1045.

Park, R.A., M.S. Treehan, P.W. Mausel, and R.C. Howe, 1989: *The Effects of Sea Level Rise on US Coastal Wetlands*. EPA-230-05-89-052, Office of Policy, Planning, and Evaluation, US Environmental Protection Agency, Washington, DC.

Parker Jr., R.O. and R.L. Dixon, 1998: Changes in a North Carolina reef fish community after 15 years of intense fishing- global warming implications. *Transactions of the American Fisheries Society*, **127(6)**, 908-920.

Parmesan, C. and H. Galbraith, 2004: *Observed Impacts of Global Climate Change in the US*. Pew Center on Global Climate Change, Arlington, VA.

Parmesan, C. and G. Yohe, 2003: A globally coherent fingerprint of climate change impacts across natural systems. *Nature*, **421**, 37-42.

Parmesan, C., 2006: Ecological and evolutionary responses to recent climate change. *Annual Review of Ecology, Evolution and Systematics*, **37**, 637-669.

Parson, E.A., P. W. Mote, A. Hamlet, N. Mantua, A. Snover, W. Keeton, E. Miles, D. Canning, and K. G. Ideker, 2001: Potential consequences of climate variability and change for the Pacific Northwest, In: *The Potential Consequences of Climate Variability and Change: Foundation Report*, Report by the National Assessment Synthesis Team for the US Global Change Research Program, Cambridge University Press, Cambridge, UK, pp. 247-281.

Pauly, D., V. Christensen, J. Dalsgaard, R. Froese, and F. Torres Jr, 1998: Fishing down marine food webs. *Science*, **279(5352)**, 860-863.

Peierls, B.L., R.R. Christian, and H.W. Paerl, 2003: Water quality and phytoplankton as indicators of hurricane impacts on a large estuarine ecosystem. *Estuaries*, **26(5)**, 1329-1343.

Peterson, C.H., K.W. Able, C. Frieswyk DeJong, M.F. Piehler, C.A. Simenstad, and J.B. Zedler, In Press: Practical proxies for tidal marsh ecosystem services. *Marine Ecology Progress Series*.

Peterson, C.H. and M.J. Bishop, 2005: Assessing the environmental impacts of beach nourishment. *BioScience*, **55(10)**, 887-896.

Peterson, C.H., M.J. Bishop, G.A. Johnson, L.M. D'Anna, and L.M. Manning, 2006: Exploiting beach filling as an unaffordable experiment: benthic intertidal impacts propagating upwards to shorebirds. *Journal of Experimental Marine Biology and Ecology*, **338(2)**, 205-221.

Peterson, C.H. and R. Black, 1988: Density-dependent mortality caused by physical stress interacting with biotic history. *American Naturalist*, **131(2)**, 257-270.

Peterson, C.H. and J. A. Estes, 2001: Conservation and management of marine communities, [Bertness, M.D., S.D. Gaines, and M.E. Hay (eds.)]. pp. 469-508.

Peterson, C.H., S.D. Rice, J.W. Short, D. Esler, J.L. Bodkin, B.E. Ballachey, and D.B. Irons, 2003: Long-term ecosystem response to the Exxon Valdez oil spill. *Science*, **302(5653)**, 2082-2086.

Peterson, D., D. Cayan, J. DiLeo, M. Noble, and M. Dettinger, 1995: The role of climate in estuarine variability. *American Scientist*, **83(1)**, 58-67.

Peterson, G.W. and R.E. Turner, 1994: The value of salt marsh edge vs interior as a habitat for fish and decapod crustaceans in a Louisiana tidal marsh. *Estuaries,* **17(1)**, 235-262.

Petraitis, P.S. and S.R. Dudgeon, 2004: Detection of alternative stable states in marine communities. *Journal of Experimental Marine Biology and Ecology*, **300(1)**, 343-371.

Pew Center on Global Climate Change, 2003: *Innovative Policy Solutions to Global Climate Change: the U.S. Domestic Response to Climate Change: Key Elements of a Prospective Program*. pp.1-8.

Piehler, M.F., L.J. Twomey, N.S. Hall, and H.W. Paerl, 2004: Impacts of inorganic nutrient enrichment on the phytoplankton community structure and function in Pamlico Sound, NC USA. *Estuarine Coastal and Shelf Science*, **61(197)**, 207.

Pielke, R., G. Prins, S. Rayner, and D. Sarewitz, 2007: Climate change 2007: lifting the taboo on adaptation. *Nature*, **445(597)**, 598.

Pikitch, E.K., C. Santora, E.A. Babcock, A. Bakun, R. Bonfil, D.O. Conover, P. Dayton, P. Doukakis, D. Fluharty, and B. Heneman, 2004: Ecosystem-based fishery management. *Science*, **305(5682)**, 346-347.

Pilkey, O.H. and H.L. Wright III, 1988: Seawalls versus beaches. *Journal of Coastal Research*, **4**, 41-64.

Poff, N.L., J.D. Allan, M.B. Bain, J.R. Karr, K.L. Prestegaard, B.D. Richter, R. Sparks, and J. Stromberg, 1997: The natural flow regime: a new paradigm for riverine conservation and restoration. *BioScience*, **47**, 769-784.

Postel, S., 1992: *The Last Oasis-Facing Water Scarcity*. Norton & Co, New York.

Poulin, R., 2005: Global warming and temperature-mediated increases in cercarial emergence in trematode parasites. *Parasitology*, **132(01)**, 143-151.

Poulin, R. and K.N. Mouritsen, 2006: Climate change, parasitism and the structure of intertidal ecosystems. *Journal of Helminthology*, **80(2)**, 183-191.

Pritchard, D.W., 1967: *What Is an Estuary: Physical Viewpoint*. Publication Number 83, American Association for the Advancement of Science, Washington, DC, pp.3-5.

Purcell, J.E., F.P. Cresswell, D.G. Cargo, and V.S. Kennedy, 1991: Differential ingestion and digestion of bivalve larvae by the scyphozoan Chrysaora quinquecirrha and the ctenophore Mnemiopsis leidyi. *Biological Bulletin, Marine Biological Laboratory, Woods Hole*, **180(1)**, 103-111.

Rabalais, N.N., R.E. Turner, and D. Scavia, 2002: Beyond science into policy: Gulf of Mexico hypoxia and the Mississippi River. *BioScience*, **52(2)**, 129-142.

Rahmstorf, S., 2007: A semi-empirical approach to projecting future sea-level rise. *Science*, **315(5810)**, 368-370.

Ramus, J., L.A. Eby, C.M. McClellan, and L.B. Crowder, 2003: Phytoplankton forcing by a record freshwater discharge event into a large lagoonal estuary. *Estuaries*, **26(5)**, 1344-1352.

Raven, J., 2005: *Ocean Acidification Due to Increasing Atmospheric Carbon Dioxide*. The Royal Society, London.

Reed, D.J., 1995: The response of coastal marshes to sea-level rise: survival or submergence? *Earth Surface Processes and Landforms*, **20(1)**, 39-48.

Reed, D.J., 2002: Sea-level rise and coastal marsh sustainability: geological and ecological factors in the Mississippi Delta plain. *Geomorphology*, **48(1)**, 233-243.

Remane, A. and C. Schlieper, 1971: *Biology of Brackish Water*. Wiley Interscience Division, John Wiley & Sons, New York, NY.

Riggs, S.R., 1996: Sediment evolution and habitat function of organic-rich muds within the Albemarle estuarine system, North Carolina. *Estuaries*, **19(2A)**, 169-185.

Riggs, S.R., 2002: *The Soundfront Series: Shoreline Erosion in North Carolina Estuaries*. UNC-SG-01-11.

Riggs, S.R. and D.V. Ames, 2003: *Drowning the North Carolina Coast: Sea-Level Rise and Estuarine Dynamics*. UNC-SG-03-04, NC Sea Grant College Program, Raleigh, NC, pp.1-152.

Rignot, E. and P. Kanagaratnam, 2006: Changes in the velocity structure of the Greenland ice sheet. *Science*, **311(5763)**, 986-990.

Rinaldi, S. and M. Scheffer, 2000: Geometric analysis of ecological models with slow and fast processes. *Ecosystems*, **3(6)**, 507-521.

Ritchie, K., 2006: Regulation of microbial populations by coral surface mucus and mucus-associated bacteria. *Marine Ecology Progress Series,* **322**, 1-14.

Roberts, C.M., S. Andelman, G. Branch, R.H. Bustamante, J.C. Castilla, J. Dugan, B.S. Halpern, K.D. Lafferty, H. Leslie, and J. Lubchenco, 2003: Ecological criteria for evaluating candidate sites for marine reserves. *Ecological Applications*, **13(1)**, S199-S214.

Robins, J.B., I.A. Halliday, J. Staunton-Smith, D.G. Mayer, and M.J. Sellin, 2005: Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine & Freshwater Research*, **56(3)**, 343-360.

Root, T.L., J. Price, K.R. Hall, S.H. Schnelder, C. Rosenzweig, and A.J. Pounds, 2003: Fingerprints of global warming on wild animals and plants. *Nature*, **421**, 57-60.

Rothschild, B.J., J.S. Ault, P. Goulletquer, and M. Heral, 1994: Decline of the Chesapeake Bay oyster population: a century of habitat destruction and overfishing. *Marine ecology progress series. Oldendorf*, **111(1)**, 29-39.

Roy, B.A., S. Guesewell, and J. Harte, 2004: Response of plant pathogens and herbivores to a warming experiment. *Ecology*, **85(9)**, 2570-2581.

Rozas, L.P., T.J. Minello, I. Munuera-Femandez, B. Fry, and B. Wissel, 2005: Macrofaunal distributions and habitat change following winter-spring releases of freshwater into the Breton Sound estuary, Louisiana(USA). *Estuarine Coastal and Shelf Science*, **65(1-2)**, 319-336.

Ruiz, G.M., J.T. Carlton, E.D. Grosholz, and A.H. Hines, 1997: Global invasions of marine and estuarine habitats by non-indigenous species: Mechanisms, extent, and consequences. *American Zoologist*, **37(6)**, 621-632.

Salathé, E.P., 2006: Influences of a shift in North Pacific storm tracks on western North American precipitation under global warming. *Geophysical Research Letters*, **33(19)**.

Sanchez-Arcilla, A. and J.A. Jimenez, 1997: Physical impacts of climatic change on deltaic coastal systems (I): an approach. *Climatic Change*, **35(1)**, 71-93.

Sarmiento, J.L., R. Slater, R. Barber, L. Bopp, S.C. Doney, A.C. Hirst, J. Kleypas, R. Matear, U. Mikolajewicz, and P. Monfray, 2004: Response of ocean ecosystems to climate warming. *Global Biogeochemical Cycles*, **18(3)**.

Scavia, D., J.C. Field, D.F. Boesch, R.W. Buddemeier, V. Burkett, D.R. Cayan, M. Fogarty, M.A. Harwell, R.W. Howarth, C. Mason, D.J. Reed, T.C. Royer, A.H. Sallenger, and J.G. Titus, 2002: Climate change impacts on U.S. coastal and marine ecosystems. *Estuaries*, **25(2)**, 149-164.

Scavia, D., E.L.A. Kelly, and J.D. Hagy, 2006: A simple model for forecasting the effects of nitrogen loads on chesapeake bay hypoxia. *Estuaries and coasts*, **29(4)**, 674-684.

Scheffer, M. and S.R. Carpenter, 2003: Catastrophic regime shifts in ecosystems: linking theory to observation. *Trends in Ecology & Evolution*, **18(12)**, 648-656.

Scheffer, M., S. Carpenter, J.A. Foley, C. Folke, and B.H. Walker, 2001: Catastrophic shifts in ecosystems. *Nature*, **413**, 591-596.

Schwimmer, R.A. and J.E. Pizzuto, 2000: A model for the evolution of marsh shorelines. *Journal of Sedimentary Research Section A: Sedimentary Petrology and Processes*, **70(5)**, 1026-1035.

Seitz, R.D., R.N. Lipcius, N.H. Olmstead, M.S. Seebo, and D.M. Lambert, 2006: Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. *Marine Ecology Progress Series*, **326**, 11-27.

Sheldon, J.E. and M. Alber, 2002: A comparison of residence time calculations using simple compartment models of the Altamaha River Estuary, Georgia. *Estuaries*, **25(6)**, 1304-1317.

Short, F.T. and D.M. Burdick, 1996: Quantifying eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. *Estuaries*, **19(3)**, 730-739.

Short, F.T. and S. Wyllie-Echeverria, 1996: Natural and human induced disturbance of seagrass. *Environmental Conservation*, **23**, 17-27.

Simenstad, C.A., K. L. Fresh, and E. O. Salo, 1982: The role of Puget Sound and Washington coastal estuaries in the life history of Pacific salmon: an unappreciated function, In: *Estuarine Comparisons*, [Kennedy, V.S. (ed.)]. Academic Press, New York, NY, pp. 343-364.

Simenstad, C.A., R. M. Thom, D. A. Levy, and D. L. Bottom, 2000: Landscape structure and scale constraints on restoring estuarine wetlands for Pacific coast juvenile fishes, In: *Concepts and Controversies in Tidal Marsh Ecology*, [Weinstein, M.P. and D.A. Kreeger (eds.)]. Kluwer Academic Publishing, Dordrecht, pp. 597-630.

Sims, D.W., V.J. Wearmouth, M.J. Genner, A.J. Southward, and S.J. Hawkins, 2004: Low-temperature-driven early spawning migration of a temperate marine fish. *Journal of Animal Ecology*, **73(2)**, 333-341.

Sklar, F.H. and J.A. Browder, 1998: Coastal environmental impacts brought about by alterations to freshwater flow in the Gulf of Mexico. *Environmental Management*, **22(4)**, 547-562.

Snover, A.K., P.W. Mote, L. Whitley Binder, A.F. Hamlet, and N.J. Mantua, 2005: *Uncertain Future: Change and Its Effects on Puget Sound. A Report for the Puget Sound Action Team by the Climate Impacts Group*. Center for Science in the Earth System, Joint Institute for the Study of the Atmosphere and Oceans, University of Washington, Seattle.

Solan, M., B.J. Cardinale, A.L. Downing, K.A.M. Engelhardt, J.L. Ruesink, and D.S. Srivastava, 2004: Extinction and ecosystem function in the marine benthos. *Science*, **306(5699)**, 1177-1180.

Southward, A.J., S.J. Hawkins, and M.T. Burrows, 1995: Seventy years' observations of changes in distribution and abundance of zooplankton and intertidal organisms in the western English Channel in relation to rising sea temperature. *Journal of Thermal Biology*, **20(1)**, 127-155.

Southward, A.J., O. Langmead, N.J. Hardman-Mountford, J. Aiken, G.T. Boalch, P.R. Dando, M.J. Genner, I. Joint, M.A. Kendall, N.C. Halliday, R.P. Harris, R. Leaper, N. Mieszkowska, R.D. Pingree, A.J. Richardson, D.W. Sims, T. Smith, A.W. Walne, and S.J. Hawkins, 2004: Long-term oceanographic and ecological research in the western English Channel. *Advances in Marine Biology*, **47**, 1-105.

Spyres, J., 1999: Rising tide: global warming accelerates coastal erosion. Erosion Control, http://www.forester.net/ec_9909_rising_tide.html, accessed on 3-22-2007.

Stachowicz, J.J., R.B. Whitlatch, and R.W. Osman, 1999: Species diversity and invasion resistance in a marine ecosystem. *Science*, **286(5444)**, 1577-1579.

Stanley, D.W., 1992: *Historical Trends: Water Quality and Fisheries, Albemarle-Pamlico Sounds, With Emphasis on the Pamlico River Estuary*. UNC-SG-92-04, University of North Carolina Sea Grant College Program Publication, Institute for Coastal and Marine Resources, East Carolina University, Greenville, NC.

Stanley, D.W. and S.W. Nixon, 1992: Stratification and bottom-water hypoxia in the Pamlico River Estuary. *Estuaries*, **15(3)**, 270-281.

Steel, J. and N. Carolina, 1991: *Albemarle-Pamlico Estuarine System: Technical Analysis of Status and Trends*. Albemarle-Pamlico Estuarine Study Report 91-01, Environmental Protection Agency National Estuary Program, Raleigh, NC.

Stephan, C.D., R.L. Peuser, and M.S. Fonseca, 2001: *Evaluating Fishing Gear Impacts to Submreged Aquatic Vegetation and Determining Mitigation Strategies*. ASMFC Habitat Management Series No. 5, Atlantic States Marine Fisheries Commission, Washington, DC.

Syvitski, J.P.M., C.J. Voeroesmarty, A.J. Kettner, and P. Green, 2005: Impact of humans on the flux of terrestrial sediment to the global coastal ocean. *Science*, **308(5720)**, 376-380.

Tait, J.F. and G.B. Griggs, 1990: Beach response to the presence of a seawall; comparison of field observations. *Shore and Beach*, **58(2)**, 11-28.

Tartig, E.K., F. Mushacke, D. Fallon, and A. Kolker, 2000: *A Wetlands Climate Change Impact Assessment for the Metropolitan East Coast Region*. Center for International Earth Science Information Network.

Tenore, K.R., 1970: The macrobenthos of the Pamlico River estuary, North Carolina. *Ecological Monographs*, **42**, 51-69.

Tilman, D. and J.A. Downing, 1994: Biodiversity and stability in grasslands. *Nature*, **367(6461)**, 363-365.

Titus, J.G., 1989: *Sea Level Rise*. EPA 230-05-89-052, U.S. Environmental Protection Agency, Washington, DC.

Titus, J.G., 2000: Does the U.S. government realize that the sea is rising? How to restructure federal programs so that wetlands can survive. *Golden Gate University Law Review*, **30(4)**, 717-778.

Titus, J.G., 2004: *Maps That Depict the Business-As-Usual Response to Sea Level Rise in the Decentralized United States of America*. paper presented at the OECD Global Forum on Sustainable Development: Development and Climate Change ENV/EPOC/GF/SD/RD(2004)9/FINAL, Paris.

Titus, J.G., 1991: Greenhouse effect and coastal wetland policy: how americans could abandon an area the size of Massachusetts at minimum cost. *Environmental Management*, **15(1)**, 39-58.

Titus, J.G., 1998: Rising seas, coastal erosion, and the takings clause: how to save wetlands and beaches without hurting property owners. *Maryland Law Review,* **57(4)**, 1279-1399.

Titus, J.G., R. Park, S.P. Leatherman, J.R. Weggel, P.W. Mausel, S. Brown, G. Gaunt, M. Trehan, and G. Yohe, 1991: Greenhouse effect and sea level rise: the cost of holding back the sea. *Coastal Management*, **19**, 171-204.

Titus, J.G. and C. Richman, 2001: Maps of lands vulnerable to sea level rise: modeled elevations along the U.S. Atlantic and Gulf Coasts. *Climate Research*, **18**, 205-228.

Turgeon, J., R. Stoks, R.A. Thum, J.M. Brown, and M.A. McPeek, 2005: Simultaneous Quaternary radiations of three damselfly clades across the holarctic. *American Naturalist*, **165(4)**, E78-E107.

Turner, R.E., J.J. Baustian, E.M. Swenson, and J.S. Spicer, 2006: Wetland sedimentation from Hurricanes Katrina and Rita. *Science*, **314(5798)**, 449-452.

Turner, R.E., W.W. Schroeder, and W.J. Wiseman, 1987: Role of stratification in the deoxygenation of Mobile Bay and adjacent shelf bottom waters. *Estuaries*, **10(1)**, 13-19.

U.S. Army Corps of Engineers, In Press: Louisiana coastal protection and restoration. *Tol be submitted to Congress*.

U.S. Climate Change Science Program, 2007: *Synthesis and Assessment Product 4.1: Coastal Elevation and Sensitivity to Sea Level Rise*. A report by the U.S. Climate Change Science Program and the Subcommittee on Global Change Research, U.S. Environmental Protection Agency.

U.S. Commission on Ocean Policy, 2004: *An Ocean Blueprint for the 21st Century*. 9, pp.15-18.

U.S. Congress, 1973: Endangered Species Act. **7 U.S.C. §136**.

U.S. Congress, 1980: Guidelines for specification of disposal sites for dredged or fill material.

U.S. Congress, 1987: Clean Water Act (amended 1987).

U.S. Congress, 1996: Magnuson-Stevens Fishery Conservation and Management Act. **Public Law 94-265**.

U.S. Congress, 2000: Estuary Restoration Act. **106-457**.

U.S. Congress, 2002: Federal Water Pollution Control Act (Clean Water Act). 33 U.S.C. 1251 - 1376, 320.

U.S. Environmental Protection Agency, 1989: *The Potential Effects of Global Climate Change on the United States: Report to Congress*. EPA-230-05-89-052, Office of Policy, Planning, and Evaluation, US Environmental Protection Agency.

U.S. Environmental Protection Agency, 2006: The National Estuary Program: a ten year perspective. U.S.Environmental Protection Agency Website, http://www.epa.gov/owow/estuaries/aniv.htm, accessed on 4-6-2007.

U.S. Environmental Protection Agency, 2007a: National Estuary Program: program profiles. EPA Website, http://www.epa.gov/owow/estuaries/list.htm, accessed on 5-30-2007a.

U.S. Environmental Protection Agency, 2007b: Office of water organizational chart. EPA Website, http://www.epa.gov/water/org_chart/index.htm, accessed on 5-30-2007b.

U.S. Environmental Protection Agency, 2007c: Performance indicators visualization and outreach tool introduction. EPA Website, www.epa.gov/owow/estuaries/habitat/index.html, accessed on 7-25-2007c.

U.S. Geological Survey, 1999: National water summary on wetland resources: state summary highlights. USGS, http://water.usgs.gov/nwsum/WSP2425/state_highlights_summary.html, accessed on 3-23-2007.

U.S. Geological Survey, 2005: Nonindigenous aquatic species search page. U.S.Geological Survey, http://nas.er.usgs.gov/queries/default.asp, accessed on 4-9-2007.

U.S. Geological Survey, 2006: Potential effects of elevated atmospheric carbon dioxide (CO_2) on coastal wetlands. USGS, http://www.nwrc.usgs.gov/factshts/2006-3074/2006-3074.htm, accessed on 4-1-2006.

U.S. Geological Survey, 2007: Hurricanes Katrina and Rita. USGS, http://www.nwrc.usgs.gov/hurricane/katrina.htm, accessed on 3-23-2007.

Vinebrooke, R.D., K.L. Cottingham, M.S.J. Norberg, S.I. Dodson, S.C. Maberly, and U. Sommer, 2004: Impacts of multiple stressors on biodiversity and ecosystem functioning: the role of species co-tolerance. *Oikos*, **104(3)**, 451-457.

Walters, C.J., 1986: *Adaptive Management of Renewable Resources*. McMillan, New York, New York.

Walther, G.R., E. Post, P. Convey, A. Menzel, C. Parmesan, T.J.C. Beebee, J.M. Fromentin, O. Hoegh-Guldberg, and F. Bairlein, 2002: Ecological responses to recent climate change. *Nature*, **416**, 389-395.

Ward, J.R. and K.D. Lafferty, 2004: The elusive baseline of marine disease: are diseases in ocean ecosystems increasing? *PLoS Biology*, **2(4)**, 542-547.

Whitfield, A.K., 2005: Fishes and freshwater in southern African estuaries - a review. *Aquatic Living Resources*, **18(3)**, 275-289.

Whitfield, A.K., 1994: Abundance of larval and 0+ juvenile marine fishes in the lower reaches of 3 Southern African estuaries with differing fresh-water inputs. *Marine Ecology Progress Series*, **105(3)**, 257-267.

Williams, S.L. and K. L. Heck Jr., 2001: Seagrass community ecology, [Bertness, M.D., S.D. Gaines, and M.E. Hay (eds.)]. Sinauer Associates, Inc, MA, USA, pp. 317-337.

Woerner, L.S. and C.T. Hackney, 1997: Distribution of *Juncus roemerianus* in North Carolina marshes: the importance of physical and abiotic variables. *Wetlands*, **17(2)**, 284-291.

Wolfe, D.A., 1986: *Estuarine Variability*. Academic Press, New York, NY.

Wolock, D.M. and G.J. McCabe, 1999: Explaining spatial variability in mean annual runoff in the conterminous United States. *Climate Research*, **11**, 149-159.

Worm, B., E.B. Barbier, N. Beaumont, J.E. Duffy, C. Folke, B.S. Halpern, J.B.C. Jackson, H.K. Lotze, F. Micheli, S.R. Palumbi, E. Sala, K.A. Selkoe, J.J. Stachowicz, and R. Watson, 2006: Impacts of biodiversity loss on ocean ecosystem services. *Science,* **314(5800)**, 787.

Yohe, G., J. Neumann, P. Marshall, and H. Ameden, 1996: The economic cost of greenhouse-induced sea-level rise for developed property in the United States. *Climatic Change*, **32(4)**, 387-410.

Zedler, J.B., 1993: Canopy architecture of natural and planted cordgrass marshes: Selecting habitat evaluation criteria. *Ecological Applications,* **3(1)**, 123-138.

Zervas, C., 2001: *Sea Level Variations of the United States, 1854-1999.* Technical Report NOS CO-OPS 36, US Dept. of Commerce, National Oceanic and Atmospheric Administration, National Ocean Service, 201.

Zimmerman, R.J., T. J. Minello, and L. P. Rozas, 2000: Salt marsh linkages to productivity of penaeid shrimps and blue crabs in the northern Gulf of Mexico, In: *Concepts and Controversies in Tidal Marsh Ecology*, [Weinstein, M.P. and D.A. Kreeger (eds.)]. pp. 293-314.

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7.9 Boxes

Box 7.1. Ecosystem services provided by coastal wetlands adapted by Peterson *et al.* (In Press) adapted from the Millennium Ecosystem Assessment (2005).

1. Habitat and food web support
    - High production at base of food chain
        - Vascular plants
        - Microphytobenthos
        - Microbial decomposers
        - Benthic and phytal invertebrates (herbivores and detritivores)
    - Refuge and foraging grounds for small fishes and crustaceans
    - Feeding grounds for larger crabs and fishes during high water
    - Habitat for wildlife (birds, mammals, reptiles)
2. Buffer against storm wave damage
3. Shoreline stabilization
4. Hydrologic processing
    - Flood water storage
5. Water quality
    - Sediment trapping
    - Nutrient cycling
    - Chemical and metal retention
    - Pathogen removal
6. Biodiversity preservation
7. Carbon storage
8. Socioeconomic services to humans
    - Aesthetics
    - Natural heritage
    - Ecotourism
    - Education
    - Psychological health

Box 7.2. Estuarine properties and the climate-driven processes that affect them. The order of the properties and processes is a subjective ranking of the importance of the property and the severity of the particular process.

Semi-enclosed geomorphology is affected by:

- sea level rise – (Rahmstorf, 2007)
- storm intensity – (Emanuel, 2005)
- storm frequency – (Emanuel, 2005)
- storm duration – (Emanuel, 2005)
- sediment delivery – (Cloern *et al.*, 1983)

Fresh water inflow is affected by:

- watershed precipitation – (Arora, Chiew, and Grayson, 2000)
- system-wide evapotranspiration – (Arora, Chiew, and Grayson, 2000)
- timing of maximum runoff – (Ramus *et al.*, 2003)
- ground water delivery – (Wolock and McCabe, 1999)

Water column mixing is affected by:

- strength of temperature-driven stratification – (Li, Gargett, and Denman, 2000)
- strength of salinity-driven stratification – (Li, Gargett, and Denman, 2000)

Water temperature is affected by:

- air temperature via sensible heat flux – (Lyman, Willis, and Johnson, 2006)
- insolation via radiant heat flux – (Lyman, Willis, and Johnson, 2006)
- temperature of fresh water runoff – (Arora, Chiew, and Grayson, 2000)
- temperature of ocean seawater advected into the estuary – (Lyman, Willis, and Johnson, 2006)

Salinity is affected by:

- exchange with the ocean – (Griffin and LeBlond, 1990)
- evaporation from estuary or lagoon – Titus (1989)

Box 7.3. “Novel” stressors resulting from climate change, together with a listing of potential biological responses to these stressors. The most important of these changes are highlighted in the main text. Not included are increases in sea levels and modifications in geomorphology of estuarine basins (barrier island disintegration), which are of utmost importance but act through complex interactions with other factors, as explained in the text.

Temperature increases, acting through thermal physiology, may cause:
- altered species (fauna and flora) distributions, including expanding ranges for tropical species currently limited by winter temperatures and contracting ranges due to increased mortality via summer temperatures
- altered species interactions and metabolic activity
- altered reproductive and migration timing
- increased microbial metabolic rates driving increased hypoxia/anoxia
- increased desiccation lethality to intertidal organisms
- increased roles of disease and parasitism
- all of the above open niches for invasive species

Timing of seasonal temperature changes, acting through phenology, disrupts:
- predator and prey availability
- food and reproductive pulses
- runoff cycle and upstream migration
- temperature-driven behavior from photoperiod-driven behavior
- biological ocean – estuary exchanges (especially of larvae and juveniles)

CO_2 increases drive acidification (lowered pH), forcing:
- reduced carbonate deposition in marine taxa
- greatly increased coral reef dieoff
- reduced photosynthetic rates
- increased trace metal toxicity

Box 7.4. Adaptation Options for Resource Managers

National Estuaries Program:
Adaptation Options for Resource Managers

- ✓ Protect the water quality of tidal marshes with oyster breakwaters and rock sills.
- ✓ Use "managed alignment" to reorient existing engineering structures affecting rivers, estuaries, and the coastlines.
- ✓ Preserve the structural complexity of vegetation in tidal marshes, seagrass meadows, and mangroves.
- ✓ Adapt protections of important biogeochemical zones and critical habitats as the locations of these areas change.
- ✓ Prohibit bulkheads and other engineered structures to preserve or delay the loss of important shallow-water habitats by permitting their inland migration as sea levels rise.
- ✓ Connect landscapes with corridors to enable migrations to sustain biodiversity across the landscape.
- ✓ Conduct integrated management of nutrient sources and wetland treatment of nutrients to limit hypoxia and eutrophication.
- ✓ Manage water resources to ensure sustainable use in the face of changing recharge rates and saltwater infiltration.
- ✓ Maintain high genetic diversity through strategies such as the establishment of reserves specifically for this purpose.
- ✓ Maintain complexity of salt marsh landscapes, especially preserving marsh edge environments.
- ✓ Restore the vegetational layering and structure of tidal marshes, seagrass meadows, and mangroves to stabilize estuary function.
- ✓ Restore native species and remove invasive non-natives to improve marsh characteristics that promote propagation and production of fish and wildlife.
- ✓ Direct restoration programs to places where the restored ecosystem has room to retreat as sea level rises.

Box 7.5. Storms as opportunities for management change

Catastrophic events provide management opportunities for increasing ecological and human resilience to climate change. Comprehensive planning could be initiated at federal, tribal, state, and local levels after major storm events to avoid future loss of life and property, and at the same time protect many environmental assets and ecosystem services in the interest of the public trust. Examples of proactive management activities include:

- Planning to prevent rebuilding in hazardous areas of high flood risk and storm damage.
- Establishing setbacks and buffer widths based on reliable projections of future erosion and sea level rise, and implementing them rapidly after natural disasters.
- Prohibiting development subsidies (e.g., federal flood insurance and infrastructure development grants) to estuarine and coastal shorelines at high risk.
- Modifying local land use plans to influence redevelopment after storms and direct it into less risky areas.
- Using funds from land trusts and programs to protect water quality, habitat, and fisheries to purchase the most risky shorelines of high resource value.

Box 7.6 Responding to the risk of coastal property loss

The practice of protecting coastal property and infrastructure with hard engineered structures, such as bulkheads, prevents marshes and beaches from migrating inland as the sea level rises. Ultimately, many marshes and beaches seaward of bulkheads will disappear as sea level rises (Titus, 1991).

Coastal marshes have kept pace with the slow rate of sea level rise that has characterized the last several thousand years. Thus, the area of marsh has expanded over time as new lands have been inundated. If in the future, sea level rises faster than the ability of the marsh to keep pace, the marsh area will contract. Construction of bulkheads to protect economic development may prevent new marsh from forming and result in a total loss of marsh in some areas.

Beach nourishment may also contribute to the loss of salt marsh on coastal barriers, because it prevents natural processes of coastal barrier recession through overwash. Overwash of sediments to the estuarine shoreline is a process that extends and revitalizes salt marsh on the protected side of coastal barriers.

Box 7.7 Estuarine water quality and climate change

Climate change may lead to changes in estuarine water quality, which in turn may affect many of the vital ecosystem services offered by estuaries.

- Changes in nutrient concentrations and light penetration into estuarine waters may affect productivity of submerged aquatic vegetation, which provides a range of services such as nursery habitat for fish species, sediment stabilization, and nutrient uptake.
- Changes in water quality may affect oxygen demand as well as directly affecting availability of dissolved oxygen. An increase in freshwater discharge to estuaries may lead to increased frequency, scope, and duration of bottom-water hypoxia arising from stronger stratification of the estuarine water column and greater microbial oxygen demand at higher temperatures.

Box 7.8. CCMP objectives for the Albemarle-Pamlico National Estuary Program (Albemarle-Pamlico National Estuary Program, 1994)

Water Quality Plan
GOAL: Restore, maintain or enhance water quality in the Albemarle-Pamlico region so that it is fit for fish, wildlife and recreation.

- Objective A: Implement a comprehensive basinwide approach to water quality management
- Objective B: Reduce sediments, nutrients and toxicants from nonpoint sources
- Objective C: Reduce pollution from point sources, such as wastewater treatment facilities and industry
- Objective D: Reduce the risk of toxic contamination to aquatic life and human health
- Objective E: Evaluate indicators of environmental stress in the estuary and develop new techniques to better assess water quality degradation

Vital Habitats Plan
GOAL: Conserve and Protect Vital Fish and Wildlife Habitats and Maintain the Natural Heritage of the Albemarle-Pamlico Sounds Region.

- Objective A: Promote regional planning to protect and restore the natural heritage of the A/P Sounds region
- Objective B: Promote the responsible stewardship, protection and conservation of valuable natural areas in the A/P Sounds region
- Objective C: Maintain, restore and enhance vital habitat functions to ensure the survival of wildlife and fisheries

Fisheries Plan
GOAL: Restore or Maintain Fisheries and Provide for Their Long-Term, Sustainable Use, Both Commercial and Recreational.

- Objective A: Control overfishing by developing and implementing fishery management plans for all important estuarine species
- Objective B: Promote the use of best fishing practices that reduce bycatch and impacts on fisheries habitats

Stewardship Plan
GOAL: Promote Responsible Stewardship of the Natural Resources of the Albemarle-Pamlico Sounds Region.

- Objective A: Promote local and regional planning that protects the environment and allows for economic growth
- Objective B: Increase public understanding of environmental issues and citizen involvement in environmental policy making
- Objective C: Ensure that students, particularly in grades K-5, are exposed to science and environmental education

Implementation Plan
GOAL: Implement the Comprehensive Conservation and Management Plan in a way that protects environmental quality while using the most cost-effective and equitable strategies.

- Objective A: Coordinate public agencies involved in resource management and environmental protection to implement the recommendations of the CCMP
- Objective B: Assess the progress and success of implementing CCMP recommendations and the status of environmental quality in the Albemarle-Pamlico Sounds region.

7.10 Tables

Table 7.1. The major stressors currently acting on estuaries and their expected impacts on management goals, as determined by consensus opinion of the contributing authors. Evidence is mounting that sea level rise is already having direct and indirect impacts on estuaries (*e.g.*, Galbraith *et al.*, 2002), but because this factor has not yet been widely integrated into management, we do not list it here despite its dominating significance in future decades.

Stressor	Water Quality	Fisheries	Habitat	Human Value & Welfare	Water Quantity
Excess Nutrients	negative	positive then negative	positive then negative	positive then negative	
Sediments	negative	positive **or** negative	positive **or** negative	negative	
Pathogens	negative	negative		negative	
Oyster Loss & Habitat Destruction	negative	negative	negative	negative	
Benthic Habitat Disturbance	negative	positive **or** negative	positive then negative	negative	
Wetland Habitat Loss from Development	negative	negative	negative	positive **or** negative	positive **or** negative
Toxics	negative	negative	negative	negative	
Invasive Species	positive **or** negative	positive **or** negative	positive **or** negative	positive **or** negative	
Thermal Pollution	positive then negative **or** down	positive then negative	pos then negative **or** down	positive then negative	
BOD	negative	negative	negative	negative	

Table 7.2. Percentage change in oceanic properties or processes as a result of climate change forcing by 2050. This table is adapted from Sarmiento *et al.* (2004). Physical changes used as inputs to the biological model are the mean of six global AOCGMs from various laboratories around the world. The AOCGMs were all forced by the IPCC IS92a scenario, which has atmospheric CO_2 doubling by 2050.

	Percentage Change by 2050 due to Climate Change Forcing					
Domain	Mixed layer	Upwelling volume	Vertical stratification	Growing season	Chlorophyll concentration	Primary productivity
marginal ice zone	-41	-10	+17	-14	+11	+18
subpolar gyre, seasonally stratified	-22	+1	+11	+6	+10	+14
subtropical gyre, seasonally stratified	-12	-6	+13	+2	+5	+5
subtropical gyre, permanently stratified	nd	-7	+8	0	+3	-3
low-latitude and equatorial upwelling	nd	-6	+11	0	+6	+9

Table 7.3. Factors that control the occurrence of estuarine hypoxia and the climate change-related impacts that are likely to affect them.

Factor	Climate-Related Forcing
Water temperature	Δ T
River discharge	Δ precipitation
N&P loading	Δ T, Δ precipitation
Stratification	Δ T, Δ precipitation, Δ RSL*
Wind	Δ weather patterns, Δ tropical storms
Organic carbon source	Δ T, Δ precipitation, Δ RSL*

*RSL = relative sea level

7.11 Figures

Figure 7.1. Organization of the NEP system (U.S. Environmental Protection Agency, 2007b).

Figure 7.2. Timeline of National Estuaries Program formation (U.S. Environmental Protection Agency, 2007a).

Figure 7.3. The Albermarle-Pamlico National Estuary Program region (Albemarle-Pamlico National Estuary Program, 2007).

Figure 7.4. Feedbacks between nutrient and sediment exchange and primary production in the benthos and water column. A plus symbol indicates enhancement and a minus symbol suppression.