CHAPTER TWO: COASTAL DEVELOPMENT

Introduction

Urban growth and development in the United States continues to expand in coastal areas at a rate approximately four times greater than that in other areas of the country (Hanson et al. 2003). Although loss of coastal wetlands to development has decreased in the last several decades, the percentual rate of loss has remained similar to that of the 1920-1950 periods (Valiela et al. 2004). Rate of loss of coastal wetlands was estimated to be 0.2% per year from 1922-1954, while loss rates from 1982-1987 were approximately 0.18% per year (Valiela et al. 2004). The construction of urban, suburban, commercial, and industrial centers and corresponding infrastructure results in land use conversions that typically remove vegetation and create additional impervious surface. At least one study has correlated ecosystem-level changes with the addition of impervious surfaces in coastal, urbanized areas. Holland et al. (2004) found reduced abundance of stress-sensitive macroinvertebrates and altered food webs in headwater tidal creeks when impervious cover exceeded 20-30% land cover. In fact, measurable adverse changes in the physical and chemical environment were observed when the impervious cover exceeded 10-20% land cover (Holland et al. 2004). Runoff from impervious surfaces and storm sewers is the most widespread source of pollution into the nation's waterways (USEPA 1995).

This chapter discusses the various sources of anthropogenic pollution, as well as other impacts to fishery habitat associated with coastal development. This report has employed the broad definition of adverse effect provided in the essential fish habitat (EFH) regulations to include "direct or indirect physical, chemical, or biological alterations of the waters or substrate and loss of, or injury to, benthic organisms, prey species and their habitat, and other ecosystem components." (50 CFR § 600.810). For this reason, impacts to the health and physiology of the fishery resources from physical, chemical, and biological factors are included. There are a number of impacts discussed in this chapter that overlap to some degree with those in other chapters of this report. We have attempted to minimize redundant information, and references to other chapters are provided when the topic has been treated in more detail elsewhere in the report.

Discharge of Nonpoint Source Pollution and Urban Runoff

The major threats to marine and aquatic habitats are a result of increasing human population and coastal development, which contribute to an increase in anthropogenic pollutant loads. These pollutants are released into estuarine and coastal habitats by way of point and nonpoint source discharges.

The US Environmental Protection Agency (US EPA) defines "nonpoint source" as anything that does not meet the legal definition of "point source" in section 502(14) of the Clean Water Act, which refers to "discernable, confined and discrete conveyance" from which pollutants are or may be discharged (for discussions of point source pollution and discharges, see the chapters on Chemical Effects: Water Discharge Facilities and Physical Effect: Water Intake and Discharge Facilities). Nonpoint source (NPS) pollution comes from many diffuse sources. Land runoff, precipitation, atmospheric deposition, seepage, and hydrologic modification are the major contributors to NPS pollution. The general categories of NPS pollution are: sediments, nutrients, acids and salts, metals, toxic chemicals, and pathogens. While all pollutants can become toxic at high enough levels, a number of compounds can be toxic at relatively low levels. The US EPA has identified and designated these compounds as "priority pollutants." Some of these "priority pollutants" include: (1) metals, such as cadmium, copper, chromium, lead, mercury, nickel, and zinc that arise from industrial operations, mining, transportation, and agriculture use; (2) organic compounds, such as pesticides, polychlorinated biphenyl (PCB) congeners, solvents, petroleum hydrocarbons, organometallic compounds, phenols, formaldehyde, and biochemical methylation of metals in aquatic sediments; (3) dissolved gases, such as chlorine and ammonium; (4) anions, such as cyanides, fluorides, sulfides, and sulphates; and (5) acids and alkalis (USEPA 2003a).

While our understanding of the individual, cumulative, and synergistic effects of all contaminants on the coastal ecosystem are incomplete, pollution discharges may cause organisms to be more susceptible to disease or impair reproductive success (USEPA 2005). Although the effects of NPS pollution are usually lower in severity than are those of point source pollution, they may be more widespread and damaging to fish and their habitats in the long term. NPS pollution may affect sensitive life stages and processes, is often difficult to detect, and its impacts may go unnoticed for a long time. When population impacts are finally detected, they may not be tied to any one event or source, and they may be difficult to correct, clean up, or mitigate. Increasing human populations and development within coastal regions generally leads to an increase in impervious surfaces, including but not limited to roads, residential and commercial development, and parking lots. Impervious surfaces cause greater volumes of run-off and associated contaminants in aquatic and marine waters.

Urban runoff is generally difficult to control because of the intermittent nature of rainfall and runoff, the large variety of pollutant source types, and the variable nature of source loadings (Safavi 1996). The 2000 National Water Quality Inventory (USEPA 2002) reported that runoff from urban areas is the leading source of impairment in surveyed estuaries and the third largest source of impairment in surveyed lakes. Urban areas can have a chronic and insidious pollution potential that one-time events such as oil spills do not.

It is important to note that the affects of pollution on coastal fishery resources may not necessarily represent a serious, widespread threat to all species and life history stages. The severity of the threat that individual pollutants may represent for aquatic organisms depends upon the type and concentration of the chemical compound and the length of exposure for a particular species and its life history stage. For example, species that spawn in areas that are relatively deep with strong bottom currents and well-mixed water may not be as susceptible to pollution as species that inhabit shallow, inshore areas near or within enclosed bays and estuaries. Similarly, species whose egg, larval, and juvenile life history stages utilize shallow, inshore waters and rivers may be more prone to coastal pollution than are species whose early life history stages develop in offshore, pelagic waters.

Nutrient loading and eutrophication

In the northeastern United States, highly eutrophic conditions have been reported in a number of estuarine and coastal systems, including Boston Harbor, MA, Long Island Sound, NY/CT, and Chesapeake Bay, MD/VA (Bricker et al. 1999). While much of the excess nutrients within coastal waters originates from sewage treatment plants, nonpoint sources of nutrients from municipal and agricultural run-off, contaminated groundwater and sediments, septic systems, wildlife feces, and atmospheric deposition from industry and automobile emissions contribute significantly (Hanson et al. 2003; USEPA 2005). Failing septic systems contribute to NPS pollution and are a negative consequence of urban development. The US EPA estimates that 10-25% of all individual septic systems are failing at any one time, introducing feces, detergents, endocrine disruptors, and chlorine into the environment (Hanson et al. 2003). Sewage waste contains significant amounts of organic matter that cause a biochemical oxygen demand, leading to eutrophication of coastal waters (Kennish 1998) (see also the chapter on Chemical Effects: Water Discharge Facilities). O'Reilly (1994) found that extensive hypoxia in the northeastern United States has been more chronic in river-estuarine systems from Chesapeake Bay to Narragansett Bay, RI, than in systems to the north, except for episodic low dissolved oxygen in Boston Harbor/Charles River, MA, and the freshwater portion of the Merrimack River, MA/NH. The US EPA's National Coastal Condition Report II (USEPA 2004) reported similar trends in northeast coast estuaries and also noted signs of degraded water quality in estuaries north of Cape Cod, MA. Although the US EPA report found much of the Acadian Province (i.e., Maine and New Hampshire) to have good water quality conditions, it identified Great Bay, NH, as only having fair to poor conditions (USEPA 2004).

Severely eutrophic conditions may adversely affect aquatic systems in a number of ways, including: reductions in submerged aquatic vegetation (SAV) through reduced light transmittance, epiphytic growth, and increased disease susceptibility (Goldsborough 1997); mass mortality of fish and invertebrates through poor water quality; and alterations in long-term natural community dynamics. The effect of chronic, diurnally fluctuating levels of dissolved oxygen has been shown to reduce the growth of young-of-the-year winter flounder (*Pseudopleuronectes americanus*) (Bejda et al. 1992). Short and Burdick (1996) correlated eelgrass losses in Waquoit Bay, MA, with anthropogenic nutrient loading primarily as a result of an increased number of septic systems from housing developments in the watershed. The environmental effects of excess nutrients and elevated suspended sediments are the most common and significant causes of SAV decline worldwide (Orth et al. 2006).

There is evidence that nutrient overenrichment has led to increased incidence, extent, and persistence of blooms of nuisance and noxious or toxic species of phytoplankton; increased frequency, severity, spatial extent, and persistence of hypoxia; alterations in the dominant phytoplankton species and size compositions; and greatly increased turbidity of surface waters from planktonic algae (O'Reilly 1994). Heavily developed watersheds tend to have reduced stormwater storage capacity, and the various sources of nutrient input can increase the incidence, extent, and persistence of harmful algal blooms (O'Reilly 1994). See also the chapters on Introduced/Nuisance Species and Aquaculture and Chemical Effects: Water Discharge Facilities for more information on harmful algal blooms.

Introduction of pathogens

Introduction of pathogens to aquatic habitats has become more common and widespread over the last 30 years, and various factors may be responsible, including NPS pollution from highly urbanized areas (O'Reilly 1994). Urban runoff typically contains elevated levels of pathogens, including bacteria, viruses, and protozoa, often a result of introductions of bacteria from leaking septic systems, agricultural manure, domestic animals, wildlife, and other sources of NPS pollution and can lead to beach and shellfish harvesting area closures (USEPA 2005). Pathogens are generally harmful to human health through the consumption of contaminated shellfish and finfish and exposure at beaches and swimming areas (USEPA 2005). While many pathogens affecting marine organisms are associated with upland runoff, there are also naturally occurring marine pathogens that affect fish and shellfish (Shumway and Kraeuter 2000). Some naturally occurring pathogens, such as bacteria from the genus *Vibrio*, or the dinoflagellate *Pfiesteria*, can produce blooms that release toxins capable of harming fish and possibly human health under certain conditions (Buck et al. 1997; Shumway and Kraeuter 2000). Although the factors leading to the formation of blooms for these species requires additional research, nutrient enrichment of coastal waters is suspected to play a role (Buck et al. 1997).

Sedimentation and turbidity

Land runoff from coastal development can result in an unnatural influx of suspended particles from soil erosion having negative effects on riverine, nearshore, and estuarine ecosystems. Impacts from this include high turbidity levels, reduced light transmittance, and sedimentation which may lead to the loss of SAV and other benthic structure (USEPA 2005; Orth et al. 2006). Other effects include disruption in the respiration of fishes and other aquatic organisms, reduction in filtering efficiencies and respiration of invertebrates, reduction of egg buoyancy, disruption of ichthyoplankton development, reduction of growth and survival of filter feeders, and decreased foraging efficiency of sight-feeders (Messieh et al. 1991; Wilber and Clarke 2001; USEPA 2005). For example, Breitburg (1988) found the predation rates of striped bass (*Morone saxatilis*) larvae on copepods to decrease by 40% when exposed to high turbidity conditions in the laboratory. De Robertis et al. (2003) found reductions in the rate of pursuit and probability of successful prey capture in piscivorous fish at turbidity levels as low as 10 nephelometric turbidity units, while the prey consumption of two species of planktivorous fish were unaffected at this turbidity level. In another laboratory study, rainbow smelt (*Osmerus mordax*) showed signs of increased swimming activity at suspended sediment concentrations as low as 20 mg/L, suggesting fish responded to increased suspended sediment concentrations with an "alarm reaction" (Chiasson 1993).

Release of petroleum products

Petroleum products consist of thousands of chemical compounds that can be toxic to marine life including polycyclic aromatic hydrocarbons (PAH), which can be particularly damaging to marine biota because of their extreme toxicity, rapid uptake, and persistence in the environment (Kennish 1998). PAH have been found to be significantly higher in urbanized watersheds when compared to nonurbanized watersheds (Fulton et al. 1993). By far, the largest amount of petroleum released through human activity comes from the use of petroleum products (e.g., cars, boats, paved urban areas, and two-stroke engines) (ASMFC 2004). Most of the petroleum consumption activities are land-based; however, rivers and storm and wastewater streams carry the petroleum to marine environments such as estuaries and bays. Although individual petroleum product releases are small, they are widespread and common and when combined, they contribute nearly 85% of the total petroleum pollution from human activities (ASMFC 2004).

Petroleum products can be a major stressor on inshore fish habitats. Short-term impacts include interference with the reproduction, development, growth, and behavior (e.g., spawning, feeding) of fishes, especially early life-history stages (Gould et al. 1994). PAH can degrade aquatic habitat, consequently interfering with biotic communities and may be discharged into rivers from nonpoint sources, including municipal run-off and contaminated sediments. Oil has been shown to disrupt the growth of vegetation in estuarine habitats (Lin and Mendelssohn 1996). Although oil is toxic to all marine organisms at high concentrations, certain species are more sensitive than others and, in general, the early life stages (i.e., eggs and larvae) of organisms are most sensitive (Gould et al. 1994; Rice et al. 2000).

Oil spills may cover and degrade coastal habitats and associated benthic communities or may produce a slick on the surface waters which disrupts the pelagic community. The water column may be polluted with oil as a result of wave action and currents dispersing the oil. Benthic habitat and the shoreline can be covered and saturated with oil, leading to the protracted damage of aquatic communities, including the disruption of population dynamics. Oil can persist in sediments for decades after the initial contamination, causing disruption of physiological and metabolic processes of demersal fishes (Vandermeulen and Mossman 1996). These changes may lead to disruption of community organization and dynamics in affected regions and permanently diminish fishery habitat. Carcinogenic and mutagenic properties of oil compounds have been identified (Larsen 1992; Gould et al. 1994). For more detail on oil spills, see the chapter on Energy-related Activities.

Alteration of water alkalinity

Fishery resources are known to be sensitive to changes in water alkalinity. Rivers and the brackish waters of estuaries are especially sensitive to acidic effluents because of the lower buffering capacity of freshwater as compared to that of salt water. The influx of pH altering flows to aquatic habitats can hinder the sustainability of fisheries. Municipal run-off, contaminated groundwater, and atmospheric deposition are potential nonpoint sources of acid influx to aquatic habitats. Acidification may disrupt or prevent reproduction, development, and growth of fish (USFWS and NMFS 1999). Osmoregulatory problems in Atlantic salmon (*Salmo salar*) smolts have been demonstrated to be related to habitats with low pH (Staurnes et al. 1996). Low pH in estuarine waters has been shown to cause cellular changes in the muscle tissues of Atlantic herring (*Clupea harengus*), which may lead to a reduction in swimming ability (Bahgat et al. 1989).

Alteration of temperature regimes

Alteration of natural temperature regimes can occur in riverine and estuarine ecosystems because of land runoff from urbanized areas. Radiant heating from impervious surfaces, such as concrete and asphalt can increase the water temperature of streams, rivers, and bays. The removal of shoreline and riparian vegetation can reduce shading effects and raise the water temperature of creeks and ponds that drain into larger water bodies. Temperature influences biochemical processes, behavior (e.g., migration), and physiology of aquatic organisms (Blaxter 1969), and long-term thermal pollution may change natural community dynamics.

Because warmer water holds less oxygen than colder water does, increased water temperatures reduce the dissolved oxygen concentration in bodies of water that are not well mixed. This may exacerbate nutrient-enrichment and eutrophication conditions that already exist in many estuaries and marine waters in the northeastern United States. In addition, increased water temperatures in the upper strata of the water column can increase water column stratification, which inhibits the diffusion of oxygen into deeper water leading to reduced (hypoxic) or depleted (anoxic) dissolved oxygen concentrations in estuaries with excess nutrients (Kennedy et al. 2002). Stratification could also affect primary and secondary productivity by suppressing nutrient upwelling and mixing in the upper regions of the water column, potentially altering the composition of phytoplankton and zooplankton. Impacts to the base of the food chain would not only affect fisheries but could impact entire ecosystems.

Release of metals

Metal contaminants are found in the water column and can persist in the sediments of coastal habitat, including urbanized areas, as well as fairly uninhabited regions, and are a potential environmental threat (Larsen 1992; Readman et al. 1993; Buchholtz ten Brink et al. 1996). High levels of metals, such as mercury, copper, lead, and arsenic, are found in the sediments of New England estuaries because of past industrial activity (Larsen 1992) and may be released into the water column during navigation channel dredging or made available to organisms as a result of storm events. Some activities associated with shipyards and marinas have been identified as sources of metals in the sediments and surface waters of coastal areas (Milliken and Lee 1990; USEPA 2001; Amaral et al. 2005). These include copper, tin, and arsenic from boat hull painting and scraping, hull washing, and wood preservatives. Treated wood used for pilings and docks releases copper compounds that are applied to preserve the wood (Poston 2001; Weis and Weis 2002). These chemicals can become available to marine organisms through uptake by wetland vegetation, adsorption by adjacent sediments, or directly through the water column (Weis and Weis 2002). Refer to the Overwater Structures section of this chapter for more information on treated wood products and their effects on aquatic organisms. Urban stormwater runoff often contains metals from automobile and industrial facilities, such as mercury, lead (used in batteries), and nickel and cadmium (used in brake linings). Refer to the chapter on Marine Transportation for more information on channel dredging and storm water impacts from marinas and shipyards.

At low concentrations, metals may initially inhibit reproduction and development of marine organisms, but at high concentrations, they can directly contaminate or kill fish and invertebrates. Shifts in phytoplankton species composition may occur because of metal accumulation and may lead to an alteration of community structure by replacing indigenous producers with species of lesser value as a food source for consumers (NEFMC 1998). Metals are known to produce a number of toxic effects on marine fish species, including skeletal deformities in Atlantic cod (*Gadus morhua*) from cadmium exposure (Lang and Dethlefsen 1987), larval developmental deformities in haddock (*Melanogrammus aeglefinus*) from copper exposure (Bodammer 1981), and reduced viable hatch rates in winter flounder embryos and increased larval mortality from silver exposure (Klein-MacPhee et al. 1984). Laboratory experiments have shown high mortality of Atlantic herring eggs and larvae at copper concentrations of 30 µg/L and 1,000 µg/L, respectively, and vertical migration of larvae was impaired at copper concentrations of greater than 300 µg/L (Blaxter 1977). Copper may also bioaccumulate in bacteria and phytoplankton (Milliken and Lee 1990). Metals have been implicated in disrupting endocrine secretions of aquatic organisms, potentially disrupting natural physiological processes (Brodeur et al. 1997; Thurberg and Gould 2005). Refer to the Chemical Effects: Water Discharge Facilities chapter for a broader discussion on endocrine-disrupting chemicals. While long-term impacts do not appear significant in most marine organisms, metals can move upward through trophic levels and accumulate in fish (bioaccumulation) at levels that can eventually cause health problems in human consumers (NEFMC 1998). See also Global Effects and Other Impacts chapter for mercury loading/bioaccumulation via the atmosphere.

Release of radioactive wastes

Radioactive wastes may be a potential threat to aquatic habitats used by fish and shellfish species. Fishery resources may accumulate radioactive isotopes in tissues that could lead to negative effects on the resource and consumers (ICES 1991). Potential sources of radioactive wastes are urban stormwater runoff, municipal landfills, atmospheric deposition, contaminated groundwater, and sediments (e.g., past offshore dumping locations [NEFMC 1998]).

Release of toxic compounds

Many different toxic compounds, including "priority pollutants" described previously, have been found in urban runoff (USEPA 2005). The US EPA reported that at least 10% of urban runoff samples contained toxic pollutants (USEPA 2005). Organic contamination contained within urban runoff, particularly chlorinated and aromatic compounds, has been implicated in causing immuno-suppression in juvenile chinook salmon (*Oncorhynchus tshawytscha*) (Arkoosh et al. 2001). The organophosphate insecticide, malathion, has been implicated in the mass mortality of American lobsters (*Homarus americanus*) in Long Island Sound during 1999 (Balcom and Howell 2006). In addition, impairment of immune response and stress hormone production were identified as examples of the sublethal effects from exposure of this compound on American lobsters (Balcom and Howell 2006). Refer to the subsections release of metals, pesticides, and herbicides in this chapter for additional information on toxic compounds.

Release of pesticides and herbicides

Although agricultural run-off is a major source of pesticide pollution in aquatic systems, residential areas are also a notable source (see Agriculture and Silviculture chapter for a discussion on agricultural runoff of pesticides). Other sources of pesticide discharge into coastal waters include atmospheric deposition and contaminated groundwater (Meyers and Hendricks 1982). Pesticides may bioaccumulate in the ecosystem by retention in sediments and detritus then ingested by macroinvertebrates, which in turn are eaten by larger invertebrates and fish (ASMFC 1992). For example, winter flounder liver tissues taken in 1984 and 1985 in Boston and Salem Harbors in Massachusetts were found to have the two highest mean concentrations of total dichlorodiphenyl trichloroethane (DDT) found in all New England sites sampled (NOAA 1991). Samples taken of soft parts from softshelled clams (*Mya arenaria*) during the same time period indicated that Boston Harbor mussels were moderately to highly contaminated with DDT when compared to nationwide sites (NOAA 1991).

There are three basic ways that pesticides can adversely affect the health and productivity of fisheries: (1) direct toxicological impact on the health or performance of exposed fish; (2) indirect impairment of the productivity of aquatic ecosystems; and (3) loss or degradation of habitat (e.g., aquatic vegetation) that provides physical shelter for fish and invertebrates (Hanson et al. 2003).

For many marine organisms, the majority of effects from pesticide exposures are sublethal, meaning that the exposure does not directly lead to the mortality of individuals. Sublethal effects can be of concern, as they impair the physiological or behavioral performance of individual animals in ways that decrease their growth or survival, alter migratory behavior, or reduce reproductive success (Hanson et al. 2003). Early development and growth of organisms involve important physiological processes and include the endocrine, immune, nervous, and reproductive systems. Many pesticides have been shown to impair one or more of these physiological processes in fish (Moore and Waring 2001; Gould et al. 1994). For example, evidence has shown that DDT and its chief metabolic by-product, dichlorodiphenyl dichloroethylene (DDE), can act as estrogenic compounds, either by mimicking estrogen or by inhibiting androgen effectiveness (Gilbert 2000). DDT has been shown to cause deformities in winter flounder eggs and Atlantic cod embryos and larvae (Gould et al. 1994). Generally, however, the sublethal impacts of pesticides on fish health are poorly understood.

The direct and indirect effects that pesticides have on fish and other aquatic organisms can be a key factor in determining the impacts on the structure and function of ecosystems (Preston 2002). This factor includes impacts on primary producers (Hoagland et al. 1996) and aquatic microorganisms (DeLorenzo et al. 2001), as well as macroinvertebrates that are prey species for fish. Because pesticides are specifically designed to kill insects, it is not surprising that these chemicals are relatively toxic to insects and crustaceans that inhabit river systems and estuaries. The use of pesticides to control mosquitoes has been suggested as a potential factor in the mass mortality of American lobsters in Long Island Sound during 1999 (Balcom and Howell 2006). Recent lab studies have shown that lobsters are considerably more sensitive to the effects of the mosquito adulticide, malathion, than are any other species previously tested. Sublethal effects (i.e., impairment of immune response and stress hormone production) occur at concentrations in parts per billion and at concentrations much lower than those observed to cause lethal effects (Balcom and Howell 2006). Lab studies have shown that American lobsters have a 96-hour LC50 (i.e., Lethal

Concentration 50- the duration and chemical concentration which causes the death of 50% of the test animals) of 33.5 ppb with immunotoxicity resulting at 5 ppb, suggesting a high sensivity in this species to both lethal and sublethal toxicity effects from malathion in seawater (De Guise et al. 2004).

Herbicides may alter long-term natural community structure by hindering aquatic plant growth or destroying aquatic plants. Hindering plant growth can have notable effects on fish and invertebrate populations by limiting nursery and forage habitat. Chemicals used in herbicides may also be endocrine disrupters, exogenous chemicals that interfere with the normal function of hormones (NEFMC 1998). Coastal development and water diversion projects contribute substantial levels of herbicides entering fish and shellfish habitat. A variety of human activities such as noxious weed control in residential development and agricultural lands, right-of-way maintenance (e.g., roads, railroads, power lines), algae control in lakes and irrigation canals, and aquatic habitat restoration results in contamination from these substances.

Conservation measures and best management practices (BMPs) for discharge of nonpoint source pollution and urban runoff (adapted from Hanson et al. 2003)

1. Remove unnecessary impervious surfaces such as abandoned parking lots and buildings from riparian and shoreline areas and reestablish wetlands and native vegetation, whenever possible. Construction of new impervious surfaces should be avoided or minimized.
2. Implement BMPs for sediment control during construction and maintenance operations, including: avoiding ground disturbing activities during the wet season; minimizing the temporal and spatial extent of the disturbance; using erosion prevention and sediment control methods; maintaining natural buffers of vegetation around wetlands, streams, and drainage ways; and avoiding building activities in areas of steep slopes and areas with highly erodable soils. Whenever appropriate, recommend the use of methods such as sediment ponds, sediment traps, bioswales, or other facilities designed to slow runoff and trap sediment and nutrients (USEPA 1993).
3. Protect, enhance, and restore vegetated buffer zones along streams and wetlands that include or influence fishery habitat.
4. Manage stormwater to duplicate the natural hydrologic cycle, maintaining natural infiltration and runoff rates to the maximum extent practicable.
5. Encourage proposed residential and commercial developments to utilize municipal wastewater facilities capable of treating sewage to the maximum extent practicable. Any proposed residential developments utilizing septic systems should include modern, state of the art systems. Ensure that they are properly sited and maintained.
6. Encourage communities to implement "smart-growth" development and land-use planning that reduces urban sprawl and minimizes impervious surfaces.
7. Encourage the use of nontreated wood materials in construction near aquatic environments.
8. Incorporate integrated pest management and BMPs as part of the authorization or permitting process to ensure the reduction of pesticide contamination in fishery habitat (Scott et al. 1999).
9. Avoid the use of pesticides and herbicides in and near aquatic habitats.
10. Refrain from aerial spraying of pesticides on windy days.
11. Address nonpoint source pollution by assessing cumulative impacts of past, present, and foreseeable future development activies on aquatic habitats in the review process.

Commercial and Domestic Water Use

Freshwater withdrawn for human use from riverine environments can alter natural current and sedimentation patterns, water quality, water temperature, and associated biotic communities (NEFMC 1998). Natural freshwater flows are subject to human alteration through water diversion for agriculture and industrial uses and modifications to the watershed. An increasing demand for potable water, combined with inefficient use of freshwater resources and natural events (e.g., droughts) have led to serious ecological damage worldwide, as well as in New England (Deegan and Buchsbaum 2005). For example, the flow of the Ipswich River in Massachusetts has been reduced to about one-half historical levels because of water withdrawals for human uses and about one-half of the native fish species on the river have been eliminated or greatly reduced (Bowling and Mackin 2003). Water withdrawal for freshwater drinking supply, power plant coolant systems, and irrigation occurs along urban and suburban areas, causing potential detrimental effects on aquatic habitats. The water withdrawal limits the amount of freshwater flowing into estuaries, which can affect the health and productivity of the ecosystem. For example, diversion of freshwater leading to increased salinities can result in oysters relocating upstream where less suitable habitat may be available and in areas subjected to higher levels of pollution (MacKenzie 2007). Urbanization leads to increases in the amount of impervious surface (e.g., roads and parking lots), which causes water to flow off the land more quickly than if the land was undeveloped and forested, reducing the natural recharge of groundwater. Alteration of the natural hydroperiod can affect circulation patterns in estuarine systems, leading to both short-term and long-term changes (Deegan and Buchsbaum 2005). In addition, the use of desalinization plants to meet industrial and municipal water needs may further alter chemical and physical environments by discharging hypersaline water into the aquatic ecosystem. Refer to the chapters on Physical Effects: Water Intake and Discharge Facilities and Alteration of Freshwater Systems for additional information on domestic and commercial freshwater usage.

Conservation measures and best management practices for commercial and domestic water use (adapted from Hanson et al. 2003)

1. Ensure that the design of water diversion projects provide adequate passage, water quality, and proper timing of water flows for all life history stages of anadromous fish and that they maintain and restore adequate channel, floodplain, riparian, and estuarine conditions.
2. Incorporate juvenile and adult fish passage facilities on water diversion projects.
3. Seasonal restrictions should be used to avoid impacts to habitat during species critical life history stages (e.g., spawning and egg development periods). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.

Road Construction and Operation

The building and maintenance of roads can affect aquatic habitats by increasing rates of erosion, debris slides, landslides, sedimentation, introduction of exotic species, and degradation of water quality (Furniss et al. 1991; Hanson et al. 2003). Paved and dirt roads introduce an impervious or semipervious surface into the landscape, which intercepts rain and increases runoff, carrying soil, sand, and other sediments (Ziegler et al. 2001) and oil-based materials more quickly into aquatic habitats. Roads constructed near streams, wetlands, and other sensitive areas may cause sedimentation in these habitats and further diminish flood plain storage capacity, subsequently increasing the need for dredging in those systems. Sedimentation and the release of contaminants into aquatic habitats can be acute following heavy rain and snow and as a result of improper road maintenance activities. Even carefully designed and constructed roads can be a source of sediment and pollutants if they are not properly maintained (Hanson et al. 2003).

The effects of roads on aquatic habitat include: (1) contaminant releases; (2) increased release of sediments; (3) reduced dissolved oxygen; (4) changes in water temperature; (5) elimination or introduction of migration barriers; (6) changes in stream flow; (7) introduction of nonnative plant species; (8) altered salinity regimes; and (9) changes in channel configuration.

Contaminant releases

Roads constructed near or adjacent to aquatic habitats can be a source of chemical contaminants, such as deicing chemicals, road salt, fertilizers, and herbicides to control roadside vegetation and petroleum products from vehicles or from the road asphalt itself (Furniss et al. 1991).

Nationally, an estimated 18 million tons of deicing salt, primarily sodium and calcium chlorides, are used each year and state and local governments spend approximately $10 million annually to remediate road salt contamination (USEPA 2005). Road salts dissolve and enter adjacent soils, groundwater, and surface waters through runoff, which can cause toxicity in plants, fish, and other aquatic organisms. These effects are particularly pronounced in smaller water bodies adjacent to salted areas. Stormwater runoff from roads can contain oil, grease, and other hydrocarbons from asphalt, wearing of tires, deposition from automobile exhaust, and oiling of roadsides and unpaved roads with crankcase oil (USEPA 2005). Refer to the Discharge of Nonpoint Source Pollution and Urban Runoff section of this chapter for information on impacts from stormwater runoff.

Sedimentation, siltation, and turbidity

The rate of soil erosion around roads is primarily a function of storm intensity, surfacing material, road slope, and traffic levels (Hanson et al. 2003). In addition, road maintenance activities such as road sanding to prevent icing and road repair can also cause sedimentation in adjacent aquatic habitats. For roads located in steep terrain, mass soil movement triggered by roads can last for decades after roads are built (Furniss et al. 1991). Surface erosion results in increased deposition of fine sediments (Bilby et al. 1989; MacDonald et al. 2001; Ziegler et al. 2001), which has been linked to a decrease in salmon fry emergence, decreased juvenile densities, and increased predation in some species of salmon (Koski 1981).

Reduced dissolved oxygen

The introduction of stormwater runoff from roads can increase the organic loads in adjacent streams and rivers, increasing the biological oxygen demand and reducing dissolved oxygen concentrations. Reduced dissolved oxygen concentrations can cause direct mortality of aquatic organisms or result in sub-acute effects such as reduced growth and reproductive success. Bejda et al. (1992) found that the growth of juvenile winter flounder was significantly reduced when dissolved oxygen (DO) levels were maintained at 2.2 mg/L or when DO varied diurnally between 2.5 and 6.4 mg/L for a period of 11 weeks.

Loss and alteration of vegetation and altered temperature regimes

Roads located near streams often involve the removal of riparian vegetation for construction and safety and maintenance. Roads built adjacent to streams result in changes in water temperature and increased sunlight reaching the stream as riparian vegetation is removed and/or altered in composition (Hanson et al. 2003). Roads can also alter natural temperature regimes in riverine and estuarine ecosystems because of radiant heating effect from the road surfaces. Riparian vegetation is an important component of rearing habitat for coldwater species, such as salmonids, providing shade for maintaining cool water temperatures, food supply, and channel stability and structure (Furniss et al. 1991).

Temperature effects biochemical processes, behavior (e.g., migration), and physiology of aquatic organisms (Blaxter 1969), and long-term thermal pollution may change natural community dynamics. In addition, increased water temperatures can reduce the dissolved oxygen concentration in bodies of water that are not well mixed. This may exacerbate eutrophication conditions that already exist in many estuaries and marine waters in the northeastern United States.

Impaired fish passage

Roads can also reduce or eliminate upstream and downstream fish passage through improperly placed culverts at road-stream crossings (Belford and Gould 1989; Clancy and Reichmuth 1990; Evans and Johnston 1980; Furniss et al. 1991). Improperly designed stream crossings adversely effect fish and aquatic organisms by blocking access to spawning, rearing, and nursery habitat because of: (1) perched culverts constructed with the bottom of the structure above the level of the stream, effectively acting as dams and physically blocking passage; and (2) hydraulic barriers to passage are created by undersized culverts which constrict the flow and create excessive water velocities (Evans and Johnston 1980; Belford and Gould 1989; Furniss et al. 1991; Jackson 2003). Smooth-bore liners made from high density plastic help meet the goal of passing water and protecting roadways from flooding, but they greatly increase flow velocities through the passage. Culverts can be plugged by debris or overtopped by high flows. Road damage, channel realignment, and extreme sedimentation from roads can cause stream flow to become too shallow for upstream fish movement (Furniss et al. 1991). Additional information on impaired fish passage is discussed in the Alteration of Freshwater Systems chapter of this report.

Introduction of exotic invasive species

Roads can be the first point of entry for nonnative, opportunistic grass species that are seeded along road cuts or introduced from seeds transported by tires and shoes (Greenberg et al. 1997; Lonsdale and Lane 1994). Nonnative plants may be able to move away from the roadside and into aquatic sites, where they may out-compete native species and alter the structure and function of the aquatic ecosystem (see also the chapter on Introduced/Nuisance Species and Aquaculture).

Altered hydrological regimes

Roads can result in adverse effects to hydrologic processes. They intercept rainfall directly on the road surface, in road cut banks, and as subsurface water moving down the hillslope; they also concentrate flow, either on the road surfaces or in adjacent ditches or channels (Hanson et al. 2003). Roads can divert or reroute water from flow paths that would otherwise be taken if the road were not present (Furniss et al. 1991). The hydrology of riverine and estuarine systems can be affected by fragmentation of the habitat caused by the construction of roads and culverts (Niering 1988; Mitsch and Gosselink 1993). These structures also reduce natural tidal flushing and interfere with natural sediment-transport processes, all of which are important functions that maintain the integrity of coastal wetlands (Tyrrell 2005). As discussed previously, roads can alter flood plain storage patterns. These hydrological changes may lead to increased erosion and sedimentation in adjacent streams.

Altered hydrology and flood plain storage patterns around estuaries can effect water residence time, temperature, and salinity and increase vertical stratification of the water column, which inhibits the diffusion of oxygen into deeper water leading to reduced (hypoxic) or depleted (anoxic) dissolved oxygen concentrations (Kennedy et al. 2002).

Altered tidal and salinity regimes

As discussed above, roads can alter hydrologic processes by rerouting flow paths and concentrating stormwater flow towards salt marsh and tidal creeks. Together with the removal of vegetation adjacent to roads, a large and rapid influx of freshwater can alter the salinity regime and species composition of estuarine habitats. Roads and culverts can also restrict the flow in tidal creeks, lowering the head-of-tide, altering the estuarine community, and restricting the access of anadromous fish.

Altered stream morphology

The geometry of a stream is affected by the amount of water and sediment that the stream carries. These factors may be altered by roads and stream crossings. Adjustments to stream morphology are usually detrimental to fish habitat (Furniss et al. 1991). Alteration of stream morphology can change stream velocity and increase sedimentation of the streambed, which can have adverse effects on spawning and migration of anadromous fish.

Conservation measures and best management practices for road construction and operation (adapted from Hanson et al. 2003)

1. Roads should be sited to avoid sensitive areas such as streams, wetlands, and steep slopes.
2. Build bridges for crossing aquatic environments, rather than utilizing culverts, whenever possible. If culverts must be used, they should be sized, constructed, and maintained to match the gradient, flow characteristics, and width of the stream so as to accommodate a 100-year flood event, but equally to provide for seasonal migratory passage of adult and juvenile fish.
3. Design bridge abutments to minimize disturbances to stream banks, and place abutments outside of the floodplain whenever possible.
4. Specify erosion control measures in road construction plans.
5. Avoid side casting of road materials into streams.
6. Use only native vegetation in stabilization plantings.
7. Use seasonal restrictions to avoid impacts to habitat during species critical life history stages (e.g., spawning and egg development periods). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
8. Maintain roadway and associated stormwater collection systems properly.
9. Control the practice of roadway sanding and the use of deicing chemicals during the winter to minimize sedimentation and introduction of contaminants into nearby aquatic habitats. Sweep and remove sand after winter to reduce sediment loading in streams and wetlands.
10. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in the review process for road construction projects.

Flood Control/Shoreline Protection

As human populations in coastal areas grow, development pressure increases and structures are often constructed along the coastline to prevent erosion and stabilize shorelines. The protection of coastal development and human communities from flooding can result in varying degrees of change in the physical, chemical, and biological characteristics of existing shoreline and riparian habitat. Attempts to protect "soft" shorelines such as beaches to reduce shoreline erosion are inevitable consequences of coastal development. Structures placed for coastal shoreline protection include breakwaters, jetties and groins, concrete or wood seawalls, rip-rap revetments (sloping piles of rock placed against the toe of the dune or bluff in danger of erosion from wave action), dynamic cobble revetments (natural cobble placed on an eroding beach to dissipate wave energy and prevent sand loss), and sandbags (Hanson et al. 2003). These structures are designed to slow or stop the shoreline from eroding, but in many cases the opposite occurs as erosion rates increase along the adjacent areas. Many shoreline "hardening" structures, such as seawalls and jetties, tend to reduce the complexity of habitats and the amount of intertidal habitats (Williams and Thom 2001). Generally, "soft" shoreline stabilization approaches (e.g., beach nourishment, vegetative plantings) have fewer adverse effects on hydrology and habitats.

Flood control measures in low-lying coastal areas include dikes, ditches, tide gates, and stream channelization. These measures are generally designed to direct water away from flood prone areas and, in the case of tide gates, prevent tidal water and storm surge from entering these areas. Adjacent aquatic habitat can become altered, and short- and long-term impacts to local fish and shellfish populations may be associated with the presence of the erosion control structures. Coastal marshes typically have a gradient of fresh to salt tolerant vegetation. These coastal wetland systems drain freshwater through tidal creeks that eventually empty into the bay or estuary. The use of water control structures can have long-term adverse effects on tidal marsh and estuarine habitats by interfering with the exchange of fresh and brackish water within the marsh habitat.

Altered hydrological regimes

Water control structures within marsh habitats intercept and carry away freshwater drainage, block freshwater from flowing across seaward portions of the marsh, increase the speed of runoff of freshwater to the bay or estuary, lower the water table, permit saltwater intrusion into the marsh proper, and create migration barriers for aquatic species (Hanson et al. 2003). In deep channels where anoxic conditions prevail, large quantities of hydrogen sulfide may be produced that are toxic to marsh grasses and other aquatic life. Long-term effects of flood control on tidal marshes include land subsidence (sometimes even submergence), soil compaction, conversion to terrestrial vegetation, reduced invertebrate populations, and general loss of productive wetland characteristics (Hanson et al. 2003). Alteration of the hydrology of coastal salt marshes can reduce estuarine productivity, restrict suitable habitat for aquatic species, and result in salinity extremes during droughts and floods.

Altered temperature regimes

Shoreline modifications, including the construction of seawalls and bulkheads, invariably involve the removal of shoreline vegetation which eliminates shading and can cause increased water temperatures in rivers and the nearshore intertidal zone (Williams and Thom 2001). Conversely, increased shading from seawalls and bulkheads constructed along shorelines may unnaturally reduce local light levels and primary production rates and reduce water temperatures of the water column adjacent to the structures (Williams and Thom 2001). Tide gates prevent or reduce tidal flushing to an area, causing stagnant water behind the structure and increased water temperature regimes (Williams and Thom 2001). Breakwaters and jetties can also alter hydrological processes which may result in altered fluctuations of nearshore temperature (Williams and Thom 2001).

Reduced dissolved oxygen

Breakwaters and jetties affect nearshore hydrological processes, as well as river flow and tidal currents when these structures are placed at the mouth of rivers and estuaries (Williams and Thom 2001). This can alter the timing and volume of water exchange to rivers, bays, and estuaries and result in reductions in water circulation and dissolved oxygen concentrations for some areas, particularly when combined with eutrophic conditions. Flood control structures, such as tide gates, dikes, and ditches, can restrict the exchange of water within wetlands, which can create stagnant conditions and reduce dissolved oxygen concentrations (Spence et al. 1996; Williams and Thom 2001).

Altered sediment transport and increased erosion/accretion

As discussed above, shoreline stabilization structures such as breakwaters, jetties, and groins affect nearshore hydrological processes which can alter wave energy and current patterns that, in turn, can affect littoral drift and longshore sediment transport (Williams and Thom 2001). In comparisons between natural and seawalled shorelines, Bozek and Burdick (2005) found no statistically significant effects on several salt marsh processes in Great Bay, NH. However, at high-energy sites, the authors found trends indicating greater sediment movement and winnowing of fine grain sediments adjacent to seawalls (Bozek and Burdick 2005).

These structures can also impact sediment budgets in estuaries and rivers. Alterations to sediment transport can affect bottom habitats, beach formation, and sand dune size (Williams and Thom 2001). Hardened shorelines, from the construction of seawalls, groins, and revetments, directly affect nearshore sediment transport by impounding natural sediment sources. Shoreline structures can cause beach erosion and accretion in adjacent areas. Long-term, chronic impacts may result in a reduction of intertidal habitat, bottom complexity, and associated soft-bottom plant and animal communities (Williams and Thom 2001). In tidal marshes, floodgates and dikes restrict sediment transport which is a natural part of the marsh accretion process. The use of these structures can result in subsidence of the marsh and loss of salt marsh vegetation.

Alteration and loss of benthic and intertidal habitat

As discussed above, breakwaters, jetties, and groins can affect nearshore hydrological processes, such as wave energy and current patterns and, in turn, can have detrimental impacts on benthic habitats. Increased sedimentation as a result of reflective turbulence (changes in water velocity caused by wave energy reflection from solid structures in the nearshore coastal area) and turbidity can reduce or eliminate vegetated shallows (Williams and Thom 2001). In addition, these structures can alter the geomorphology of existing habitats, resulting in a large-scale replacement of soft-bottom, deepwater habitat with shallow and intertidal, hard structure habitats (Williams and Thom 2001). Alterations to the shoreline as a result of bulkhead and other hard shoreline structures can increase wave energy seaward of the armoring, causing scouring of bottom sediments and loss of salt marsh vegetation.

Altered stream morphology

Flood and erosion control structures such as bulkheads, levees, and dikes built along streams and rivers, as well as the canalization of streams and rivers, result in simplified riverine habitat and a reduction in pools and riffles that provide habitat for fish (Spence et al. 1996). In addition, altered stream hydrology and morphology can change sediment grain size and reduce the organic matter available to small organisms that serve as prey for larger species in the food web (Williams and Thom 2001).

Impacts to riparian habitat

As discussed above, shoreline modifications such as the construction of seawalls and bulkheads, involve the removal of shoreline vegetation which eliminates shading and can cause increased water temperatures in rivers and the nearshore, intertidal zone (Williams and Thom 2001). The loss of riparian vegetation reduces the forage and cover for aquatic organisms and the input of large woody debris and smaller organic detritus, including leaves (Spence et al. 1996).

Impaired fish passage

Tide gates and other flood control structures can eliminate or restrict access of fish to salt marshes. Tide gates can create physical barriers for estuarine fish species that utilize salt marsh wetlands for feeding and early development. High flow rates at tide gates or culvert openings can prevent small fish from accessing critical marsh and freshwater habitat. In some cases, fish can become trapped behind tide gates, preventing them from accessing deeper water and potentially stranding them during periods of low water (Williams and Thom 2001).

Alteration of natural communities

Armoring of shorelines to prevent erosion and maintain or create shoreline real estate simplifies habitats, reduces the amount of intertidal habitat, and negatively affects nearshore processes and the ecology of coastal species (Williams and Thom 2001). For example, Chapman (2003) found a paucity of mobile species associated with seawalls in a tropical estuary, compared with surrounding areas. In that study, approximately 50% of taxa found on natural rocky shorelines were absent on constructed seawall, and seawalls were found to have a diminished proportion of rare taxa. Alterations to the shoreline from hydraulic action include increased energy seaward of the armoring from reflected wave energy, narrowing of the dry beach, coarsening of the substrate, steepening of the beach slope, reduction of sediment storage capacity, a loss of organic debris, and a reduction of downdrift sediment (Williams and Thom 2001). Bozek and Burdick (2005) found no statistically significant effects of seawalls on salt marsh processes in Great Bay, NH; however, their data indicated seawalls tended to eliminate the high-diversity vegetative zones at the upper border of the salt marsh. Installation of breakwaters and jetties can result in community changes, including burial or removal of resident biota, changes in the habitat structure, alteration in prey and predator interaction, and physical obstructions that can alter the recruitment patterns of larvae (Williams and Thom 2001).

Reduced ability to counter sea-level rise

The effect of shoreline erosion and land subsidence will likely be exacerbated by sea-level rise because of global climate change. Sea level rose 10-20 cm (4-8 inches) in the 20th century and may rise another 18-59 cm (7-23 inches) by 2100 (IPCC 2007). As sea levels continue to rise, salt marshes, mudflats, and coastal shallows must be able to shift horizontally without interruption from natural or manmade barriers (Bigford 1991). Hard structures, such as seawalls, bulkheads, and jetties may inhibit the shoreward migration of marsh wetlands (Kelley 1992) and SAV beds (Orth et al. 2006). In addition, global climate change is expected to cause greater precipitation and more intense storms in the mid-high latitudes in the northern hemisphere (Nedeau 2004). Along with rising sea levels, these factors may exacerbate coastal erosion and increase the apparent need for shoreline protection. See Global Effects and Other Impacts chapter for more information on global climate change.

Conservation measures and best management practices for flood control/shoreline protection (adapted from Hanson et al. 2003)

1. Avoid or minimize the loss of coastal wetlands as much as possible, including encouraging coastal wetland habitat preservation. Preservation of coastal upland buffers between buildings and wetlands may allow for the inland migration of wetlands as sea levels rise.
2. Avoid the diking and draining of tidal marshlands and estuaries, whenever possible.
3. Use "soft" approaches (such as beach nourishment, vegetative plantings, and placement of large woody debris), in lieu of "hard" shoreline stabilization and modifications (such as concrete bulkheads and seawalls, concrete or rock revetments), whenever possible.
4. Ensure that the hydrodynamics and sedimentation patterns are properly modeled and that the design avoids erosion to adjacent properties when "hard" shoreline stabilization is deemed necessary.
5. Include efforts to preserve and enhance fishery habitat (e.g., provide new gravel for spawning or nursery habitats; remove barriers to natural fish passage; and use of weirs, grade control structures, and low flow channels to provide the proper depth and velocity for fish) to offset impacts from proposed riparian habitat and stream modifications.
6. Construct a low-flow channel to facilitate fish passage and help maintain water temperature in reaches where water velocities require armoring of the riverbed.
7. Replace in-stream fish habitat by installing boulders, rock weirs, and woody debris and by planting riverine aquatic cover vegetation to provide shade and habitat.
8. Avoid installing new water control structures in tidal marshes and freshwater streams. If the installation of new structures cannot be avoided, ensure that they are designed to allow optimal fish passage and natural water circulation.
9. Ensure water control structures are monitored for potential alteration of water temperature, dissolved oxygen concentration, and other parameters.
10. Use seasonal restrictions to avoid impacts to habitat during species critical life history stages (e.g., spawning, egg, and larval development periods). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
11. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in the review process for flood control and shoreline protection projects.

Beach Nourishment

Beach nourishment, the process of mechanically or hydraulically placing sediments (i.e., sand and gravel) directly on an eroding shore to restore or form a protective or desired recreational beach, has been steadily increasing along the eastern US coastline since the 1960s (Greene 2002). Beaches and shorelines are dynamic, constantly eroding and accreting because of exposure to waves, currents, and wind. Beach nourishment serves as a "soft," sacrificial barrier to protect the beach and property along the coast from storm and flood damage. Between 1923 and 2004, it is estimated that approximately 515 million cubic yards of beach sediment have been deposited on the

US east coast barrier island shoreline from Maine to Florida, including 966 instances of beach nourishment at 343 locations (Valverde et al. 1999; PSDS 2005).

Beach nourishment as a protective measure against coastal flooding and storm damage may be considered less of an impact to marine organisms and fishery habitat than are most "hard" structure solutions discussed in the previous section. However, beach nourishment can have a number of short- and long-term impacts on fishery resources, including displacing benthic organisms during and after nourishment, interference with respiration and feeding in finfish and filter feeding invertebrates, temporary removal of benthic prey, burial of habitat that serves as foraging and shelter sites, potential burial of demersal and benthic species, and mortality of species at vulnerable life stages, such as eggs, larvae, and juveniles (Greene 2002). Sand or cobble material needed for beach nourishment is generally dredged from offshore areas, referred to as borrow or mining sites, and either hydraulically pumped through pipes or loaded onto barges for transfer and placement on the beach. Fish and invertebrates in and around the borrow site can be subjected to entrainment, sedimentation, and increased turbidity during the dredging and transport of the beach material. In addition, the creation of borrow pits may alter the bottom topography and sediment transport processes in offshore habitats and form depressions with low-dissolved oxygen. Nourished beaches seldom last as long as natural beaches, and natural coastal processes erode the replenished sand, requiring additional nourishment of those beaches (Pilkey and Dixon 1996). The life span of a nourished beach can be highly variable and primarily dependent upon storm intensity and frequency following the completion of a project. According to Pilkey and Dixon (1996), the life span of most nourished beaches is 2-5 years. Beach nourishment projects are often conducted at a high economic cost, and they can represent a long-term and cumulative impact on the marine biological community.

Increased global precipitation, more intense storms, and sea level rise predicted for the mid-high latitudes in the northern hemisphere because of global climate change will likely exacerbate erosional forces on beaches (Nedeau 2004) and increase the frequency of beach renourishment to protect eroding shoreline. See Global Effects and Other Impacts chapter for more on global climate change.

Altered hydrological regimes

Sand removed from borrow sites can potentially affect the geomorphology of offshore sand bars and shoals that absorb incoming waves, causing greater wave energy and/or change refraction patterns (Greene 2002). This may increase the erosion rate at the nourished beach and adjacent, nonnourished beaches. In addition, nourished beaches tend to have altered sediment grain size, shape, and distribution across the beach, which can lead to changes in the hydrodynamic patterns in the intertidal beach zone (Pilkey and Dixon 1996; Greene 2002).

In addition, the conditions in deeply excavated borrow pits can become anaerobic during certain times of the year. The dissolved oxygen concentration within these deep pits can be depressed to a level that adversely affects the ability of fish and invertebrates to utilize the area for spawning, feeding, and development (Pacheco 1984). For example, construction grade aggregate removal in Raritan Bay, NJ, Long Island Sound, and the intercoastal waterway in New Jersey have left deep pits and large depressions that are more than twice the depth of the surrounding area. The pits have remained chemically, physically, and biologically unstable with limited biological diversity for more than five decades. These borrow pits in Raritan Bay were found to possess depressed benthic communities and elevated levels of highly hydrated and organically enriched sediments (Pacheco 1984).

Altered sediment transport

Longshore transport of sediments may be affected by the creation of borrow pits, which can be deep depressions taking several years to refill and can alter the nearshore sediment budget (Greene 2002). Longshore sediment transport may also be affected in the nearshore environment if material placed on the beach is not compatible with natural or historic material. In addition, nearshore rock groins are sometimes constructed in order to reduce erosion of the nourished beach, which alters the downdrift of sediment and may starve adjacent beaches of sand.

Alteration/loss of benthic habitat

Sand infauna and sessile benthic organisms in the path of dredging equipment at the borrow site are generally removed and killed during mining. In addition, some mobile organisms, such as crustaceans and larval and juvenile fish, can be entrained by the dredge equipment. Following mining, species diversity of benthic infaunal organisms within borrow pits drops precipitously, but recolonization in sandy sediments typically occurs through larval transport and migration of postsettlement life-stages (i.e., juveniles and adults) (Greene 2002).

Benthic fauna at the beach site will be killed by burial following nourishment unless an organism is capable of burrowing through the overburden of sand (Greene 2002). Several factors determine survival of beach invertebrate fauna, including the ability for vertical migration through the sand overburden and the recruitment potential of larvae, juveniles, and adult organisms from adjacent areas (Greene 2002). Peterson et al. (2000) found an 86-99% reduction in the abundance of dominant species of beach macro-invertebrates ten weeks after nourishment on a North Carolina beach. These observations were made between the months of June and July, when the abundances of beach macro-invertebrates are typically at their maximum and providing the important ecosystem service of feeding abundant surf fishes and ghost crabs (Peterson et al. 2000).

Alteration of natural communities

The recovery of the benthic infauna at a borrow site is dependent upon a number of factors, including the amount of material removed, the fauna present at the site and surrounding area prior to dredging, and the degree of sedimentation that occurs following dredging (Greene 2002). For sand habitats, the recovery time of benthic infauna within borrow sites has been reported to be as rapid as less than one year, while other studies have indicated recovery may take greater than five years (Greene 2002). Some differences in recovery time may be attributed to the fact that most benthic infauna recolonization studies look at abundance of individuals but fail to measure trophic level changes and the life history of individuals in the samples (Greene 2002). The postdredging benthic community may function very differently than does the predredging community. The borrow pits may require several years to refill with sediment and may contain a greater silt content than do the surrounding areas (Greene 2002). Generally, the degree of alteration of the sediment composition appears to be the largest factor in determining long-term impact at a borrow site (Greene 2002). The dissolved oxygen concentration within borrow pits can be depressed to a level that adversely affects the ability of fish and invertebrates to utilize the area for spawning, feeding, and development (Pacheco 1984).

Similar to the findings on the recovery of benthic infauna at borrow sites, results of studies assessing the recovery of organisms at nourished beaches are highly variable (Greene 2002). While some studies conclude that beach infauna populations may recover to predredging levels between two to seven months, other studies suggest recovery times are much longer (Greene 2002). Peterson et al. (2000) found a large reduction in prey abundance and body size of benthic macroinvertebrates at a nourished intertidal beach that likely translated to trophic level impacts on surf zone fishes and shorebirds.

Increased sedimentation/turbidity

High turbidity in the water column and sedimentation on adjacent benthic habitats can result from resuspension of sediment at the discharge pipe and from sediment winnowing from the nourished beach into the surf zone. In addition, turbidity can also increase between the borrow site and the target beach when sand is lost during hopper loading, from leaks in the pipelines carrying sand to the beach, and from the dredging activity at the borrow site itself. High turbidity and suspended sediments can be persistent in the nearshore waters long after a beach is nourished if mud balls, silt, and clays are present in the mined sediment (Greene 2002).

Generally, the severity of the effects of suspended sediments on aquatic organisms increases as a function of sediment concentration and the duration of exposure (Newcombe and Jensen 1996). Some of the effects of suspended sediments on marine organisms can include altered foraging patterns and success (Breitburg 1988), gill abrasion and reduced respiratory functions, and death (Wilber and Clark 2001). The sensitivity of species to suspended sediments is highly variable and dependent upon the nature of the sediment and the life history stage of the species. The eggs and larval stages of marine and estuarine fish are generally highly sensitive to suspended sediment exposures compared to some freshwater taxa studied (Wilber and Clark 2001). Sedimentation from beach nourishment may also have adverse effects on invertebrates that serve as prey for fish (Greene 2002). Refer to the Marine Transportation and Offshore Dredging and Disposal chapters for more information regarding turbidity and sedimentation impacts on aquatic organisms.

Conservation measures and best management practices for beach nourishment (adapted from Hanson et al. 2003)

1. Avoid sand mining in areas containing sensitive marine benthic habitats (e.g., spawning and feeding sites, hard bottom, cobble/gravel substrate, shellfish beds).
2. Avoid beach nourishment in areas containing sensitive marine benthic habitats adjacent to the beach (e.g., spawning and feeding sites, hard bottom, cobble/gravel substrate).
3. Conduct beach nourishment during the winter and early spring, when productivity for benthic infauna is at a minimum; this may minimize the impacts for some beach sites.
3. Assess source material for compatibility with that of material to be placed on beach (e.g., grain size and shape, color). Slope of nourished beach should mimic the natural beach profile.
4. Use upland beach material sources, if compatible, to avoid impacts associated with offshore sand mining.
5. Preserve, enhance, or create beach dune and native dune vegetation in order to provide natural beach habitat and reduce the need for nourishment.
6. Monitor turbidity during operations, and cease operations if turbidity exceeds predetermined threshold levels at the beach and borrow sites.
7. Implement seasonal restrictions to avoid impacts to habitat during species critical life history stages (e.g., spawning season, egg, and larval development period). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
8. Address the cumulative impacts of past, present, and foreseeable future development activies on aquatic habitats by considering them in the review process for beach nourishment projects.

Wetland Dredging and Filling

The dredging and filling of coastal wetlands for commercial and residential development, port, and harbor development directly removes important wetland habitat and alters the habitat surrounding the developed area. Even development projects that appear to have minimal individual wetland impacts can have significant cumulative effects on the aquatic ecosystem. This section discusses the impacts on fishery habitat from dredging and filling freshwater and tidal wetlands for development purposes. Additional information on dredging and filling in freshwater wetlands and rivers and streams is provided in the chapter on Alteration of Freshwater Systems, and dredging and disposal of dredge material in subtidal habitats (e.g., navigation channel dredging and marine mining) have been addressed in the chapters on Marine Transportation and Offshore Dredging and Disposal. The primary impacts to fishery habitat from the introduction of fill material in or adjacent to wetlands include: (1) physical loss of habitat; (2) loss or impairment of wetland functions; and (3) changes in hydrologic patterns.

The discharge of dredge and fill materials are regulated under Section 404 of the Clean Water Act (CWA) of 1972 for all "waters of the United States," which include both freshwater and tidal wetlands. Some of the types of discharge of fill material covered under Section 404 of the CWA include: (1) placement of fill that is necessary to the construction of a structure or impoundment; (2) site development fills for recreational, industrial, commercial, or residential uses; (3) causeway or road fills, dams, or dikes; (4) artificial islands; (5) property protection and/or reclamation devices such as riprap, groins, seawalls, breakwaters, and revetments; (6) beach nourishment; (7) levees; (8) fill for structures such as sewage treatment facilities, intake and outfall pipes associated with power plants and subaqueous utility lines; and (9) artificial reefs.

Loss and alteration of wetland vegetation

Salt marsh wetlands serve as habitat for early life history stages of many fish species, as well as shellfish, crabs, and shrimp, which use the physical structure of the marsh grasses as refuge from predators (Tyrrell 2005). Smaller fish, such as mummichog (*Fundulus heteroclitus*), Atlantic silverside (*Menidia menidia*), sticklebacks (*Gasterosteids*, spp.), and sheepshead minnow (*Cyprinidon variegates*), rely on salt marshes for parts of their life cycles. These species form the prey base of many larger, commercially important species such as a number of flounder species, black sea bass (*Centropristis striata*), and bluefish (*Pomatomus saltatrix*) (Collette and Klein-MacPhee 2002).

Filling wetlands removes productive habitat and eliminates the important functions that both aquatic and many terrestrial organisms depend upon. For example, the loss of wetland habitats reduces the production of detritus, an important food source for aquatic invertebrates; alters the uptake and release of nutrients to and from adjacent aquatic and terrestrial systems; reduces wetland vegetation, an important source of food for fish, invertebrates, and water fowl; hinders physiological processes in aquatic organisms (e.g., photosynthesis, respiration) caused by degraded water quality and increased turbidity and sedimentation; alters hydrological dynamics, including flood control and groundwater recharge; reduces filtration and absorption of pollutants from uplands; and alters atmospheric functions, such as nitrogen and oxygen cycles (Niering 1988; Mitsch and Gosselink 1993).

Altered hydrological regimes

The discharge of dredged or fill material into aquatic habitats can modify current patterns and water circulation by obstructing the flow or by changing the direction or velocity of water flow and circulation. As a result, adverse changes can occur in the location, structure, and dynamics of aquatic communities; shoreline and substrate erosion and deposition rates; the deposition of suspended particulates; the rate and extent of mixing of dissolved and suspended components of the water body; and water stratification (Hanson et al. 2003). Altering the hydrology of wetlands can affect the water table, groundwater discharge, and soil salinity, causing a shift in vegetation patterns and quality of the habitat. Hydrology can be affected by fragmenting the habitat caused by the construction of roads and residential development or by building bulkheads, dikes, levees, and other structures designed to prevent or remove floodwater from the land around the wetlands (Niering 1988; Mitsch and Gosselink 1993). These structures also reduce natural tidal flushing and interfere with natural sediment-transport processes, all of which are important functions that maintain the integrity of the marsh habitat (Tyrrell 2005). Altered hydrodynamics can affect estuarine circulation, including short-term (diel) and longer term (seasonal or annual) changes (Deegan and Buchsbaum 2005). Alteration of the hydrology and soils of salt marsh wetlands has led to the invasion of an exotic haplotype of the common reed (*Phragmites australis*), which has spread dramatically and degraded salt marsh habitats along the Atlantic coast (Posey et al. 2003; Tyrrell 2005).

Loss of flood storage capacity

Coastal wetlands absorb and store rain and urban runoff, buffering upland development from floods. In addition, coastal marshes provide a physical barrier that protects upland development from storm surge. As a result, the loss and alteration of coastal wetlands can cause upland development to be more prone to flooding from storms and heavy rains. Furthermore, altering the hydrological regimes of wetlands through construction of dikes, levees, and tide gates can redirect floodwater towards rivers and estuaries and bypass the natural flood storage functions of coastal wetlands.

Altered current patterns

Replacing wetlands with roads, buildings, and other impervious surfaces increases the volume and intensity of storm water runoff, which can accelerate the rate of coastal erosion. Placing dredge material onto intertidal mud habitats can dramatically alter tidal flow. These effects can change the geomorphology and current patterns of rivers and estuaries and adversely affect habitat suitability for certain species. For example, counter current flows set up by freshwater discharges into estuaries are important for larvae and juvenile fish entering those estuaries. Behavioral adaptations of marine and estuarine species allow larvae and early juveniles to concentrate in estuaries (Deegan and Buchsbaum 2005).

Altered temperature regimes

The loss of riparian and salt marsh vegetation can increase the amount of solar radiation reaching streams and rivers and results in an increase in the water temperatures of those water bodies (Moring 2005). Replacing coastal wetlands with impervious surfaces such as asphalt, which absorb more solar radiation than does vegetation, tends to raise the water temperature in adjacent aquatic environments. Altered temperature regimes have the ability to affect the distribution; growth rates; survival; migration patterns; egg maturation and incubation success; competitive ability; and resistance to parasites, diseases, and pollutants of aquatic organisms (USEPA 2003b). In freshwater habitats of the northeastern United States, the temperature regimes of cold-water fish such as salmon, smelt, and trout may be exceeded, leading to local extirpation of these species (Moring 2005). The removal of riparian vegetation can also have the effect of lowering water temperatures during winter, which can increase the formation of ice and delay the development of incubating fish eggs and alevins in salmonids (Hanson et al. 2003).

Release of rutrients/eutrophication

When functioning properly, riparian and tidal wetlands support denitrification of nitrate-contaminated groundwater. While sediment particles can bind to some nutrients, resuspension of sediments following a disturbance tends to cause a rapid release of nutrients to the water column (Lohrer and Wetz 2003). Coastal wetlands reduce the risk of eutrophication in estuaries and nearby coastal waters (Tyrrell 2005) by absorbing nutrients in groundwater and storm water. Eliminating or degrading coastal wetlands through dredge and fill activities can eliminate these important wetland functions and adversely affect estuarine and marine ecosystems.

Release of contaminants

The removal of wetlands eliminates an important wetland function: pollution filtration (Niering 1988; Mitsch and Gosselink 1993). Wetlands are capable of absorbing metals, pesticides, excess nutrients, oxygen-consuming substances, and other pollutants that would otherwise be transported directly to aquatic environments. In addition, dredging and filling of wetlands can release contaminants that have accumulated in the sediments into adjacent aquatic habitats.

Increased sedimentation/turbidity

When functioning properly, riparian and tidal wetlands filter sediment and runoff from floodplain development. Siltation, sedimentation, and turbidity impacts on riverine and estuarine habitats can be worsened by the loss and replacement of wetlands with impervious surfaces. Suspended sediments in aquatic environments reduce the availability of sunlight to aquatic plants, cover fish spawning areas and food supply, interfere with filtering capacity of filter feeders, and can clog and harm the gills of fish (USEPA 2003b).

Loss of fishery productivity

Hydrological modifications from dredge and fill activities and general coastal development are known to increase the amount of run-off entering the aquatic environment and may contribute to the reduced productivity of fishery resources. Many wetland dependent species, such as mummichog, Atlantic silverside, sticklebacks, and sheepshead minnow, are important prey for larger, commercially important species such as a number of flounder species, black sea bass, and bluefish (Collette and Klein-MacPhee 2002). Although there have been sharp declines or collapses of many estuarine-dependent fisheries in the United States, attributing reductions in fishery productivity directly to losses of wetland habitat can be complicated (Deegan and Buchsbaum 2005). Recent wetland losses can be quantified for discrete regions and the nation as a whole; however, a number of other factors, such as overfishing, cultural eutrophication, and altered input of freshwater caused by flood control structures, probably all contribute to a reduction in the productivity of fisheries. Since the implementation of the Clean Water Act in 1972, the major problems for coastal habitats have changed from outright destruction to more subtle types of degradation, such as cultural eutrophication (Deegan and Buchsbaum 2005).

Introduction of invasive species

A nonnative haplotype of the common reed, *Phragmites australis,* has expanded its range along the entire east coast of the United States, primarily in wetland habitats disturbed by nutrient loading and hydrological alterations of salt marsh wetlands (Posey et al. 2003). *Phragmites* is tolerant of low-salinity conditions in salt marshes, which can occur with tidal restrictions from the construction of tide gates, bulkheads, and dikes. Under these conditions, *Phragmites* can out-compete native salt marsh vegetation such as *Spartina* sp. (Burdick et al. 2001; Deegan and Buchsbaum 2005). Salt marshes that are dominated by *Phragmites* may have reduced function and productivity compared to that of salt marshes consisting of native marsh vegetation (Tyrrell 2005).

Conservation measures and best management practices for wetland dredging and filling (adapted from Hanson et al. 2003)

1. Apply a sequence of measures to avoid, minimize, and mitigate adverse impacts in wetlands to all proposed dredging projects. Dredging and filling within wetlands should be avoided to the maximum extent practicable.
2. Consider only "water-dependent" dredge and fill projects in wetlands and only after upland alternatives have been investigated.
3. Do not dispose dredge material in wetlands, and ensure that these materials meet or exceed applicable state and/or federal water quality standards.
4. Identify and characterize fishery habitat functions/services in the project areas prior to any dredge and fill activities.
5. Identify the direct and indirect affects of wetland fills on fishery habitat during proposed project reviews, including alterations of hydrology and water quality as a result of the proposed project.
6. Assess the cumulative impact from past, current, and all reasonably foreseeable future dredge and fill operations that impact aquatic habitats via federal, state, and local resource management and permitting processes.
7. Use seasonal restrictions to avoid impacts to habitat during species critical life history stages (e.g., spawning and egg development periods). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
8. Undertake activities in wetlands, if required, using only low ground pressure vehicles.
9. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in the review process for wetland dredge and fill projects.

Overwater Structures

With increasing coastal development comes a concomitant interest in the construction and operation of waterfront facilities, the use of coastal waterways, and the environmental implications of these activities (Barr 1993). Overwater structures include commercial and residential piers and docks, floating breakwaters, moored barges, rafts, booms, and mooring buoys. These structures are typically located from intertidal areas to areas of water depths approximately 15 m below mean low water (i.e., the shallow subtidal zone). Light, wave energy, substrate type, depth, and water quality are the primary factors controlling the plant and animal assemblages found at a particular site. Overwater structures and associated use activities can alter these factors and interfere with key ecological functions such as spawning, rearing, and the use of refugia. Site-specific factors (e.g., water clarity, current, depth) and the type and use of a given overwater structure determine the occurrence and magnitude of these impacts (Hanson et al. 2003).

Shading impacts to vegetation

Overwater structures create shade which reduces the light levels below the structure. Shading from overwater structures can reduce prey organism abundance and the complexity of the habitat by reducing aquatic vegetation and phytoplankton abundance (Haas et al. 2002). The size, shape, and intensity of the shadow cast by a particular structure are dependent upon its height, width, construction materials, and orientation. In field studies conducted in Massachusetts, the most significant factors affecting shading impacts on eelgrass were the height of the structure above vegetation, orientation of the dock, and dock width (Burdick and Short 1999). High and narrow piers and docks produce narrower and more diffuse shadows than do low and wide structures. Increasing the numbers of pilings used to support a pier increases the shade cast by pilings on the under-pier environment. In addition, less light is reflected underneath structures built with light-absorbing materials (e.g., wood) than from structures built with light-reflecting materials (e.g., concrete or steel). Under-pier light levels have been found to fall below threshold amounts for the photosynthesis of diatoms, benthic algae, eelgrass, and associated epiphytes and other autotrophs. Eelgrass and other macrophytes can be reduced or eliminated, even through partial shading of the substrate, and have little chance to recover (Kenworthy and Hauners 1991). Structures that are oriented north-south produce a shadow that moves across the bottom throughout the day, resulting in a smaller area of permanent shade than those that are oriented east-west (Burdick and Short 1999; Shafer 1999). In a report investigating effects of residential docks in south Florida, Smith and Mezich (1999) found approximately 40% of the docks surveyed had additions fixed to them (e.g., boat lifts and cradles, floating docks, finger piers). These structural additions increased the dock area (and seagrass impacts) and ranged from 16-77%, and contributed to mean seagrass impacts of 47% beyond the footprint of the dock.

Similar shading impacts to salt marsh vegetation from docks and piers have been reported. A study in Connecticut measuring the density and average plant height of salt marsh vegetation below docks and adjacent areas found a reduction in vegetative reproductive capacity caused by the presence of docks (Kearney et al. 1983). This study concluded that the height of the dock was a strong determining factor in the effects to salt marsh vegetation.

Altered hydrological regimes

Alterations to wave energy and water transport from overwater structures can impact the nearshore detrital foodweb by altering the size, distribution, and abundance of substrate and detrital materials (Hanson et al. 2003). The disruption of longshore transport can alter substrate composition and can present potential barriers to the natural processes that build spits and beaches and provide substrates required for plant propagation, fish and shellfish settlement and rearing, and forage fish spawning (Hanson et al. 2003).

Contaminant releases

Kennish (2002) identified a number of contaminants associated with overwater structures that can be released into the aquatic environment, including detergents, petroleum products, and copper. Treated wood used for pilings and docks releases contaminants into the aquatic environment. Creosote-treated wood pilings and docks commonly release PAH and other chemicals, such as ammoniacal copper zinc arsenate (ACZA) and chromated copper arsenate (CCA), which are applied to preserve the wood (Poston 2001; Weis and Weis 2002). These chemicals can become available to marine organisms through uptake by wetland vegetation, adsorption by adjacent sediments, or directly through the water column (Weis and Weis 2002). The presence of CCA in the food chain can also cause a localized reduction in species richness and diversity (Weis and Weis 2002). These preservatives are known to leach into marine waters after installation, but the rate of leaching is highly variable and dependent on many factors, including the age of the treated wood. Concrete or steel, on the other hand, are relatively inert and do not leach contaminants into the water.

Benthic habitat impacts

Additional impacts associated with overwater structures may include damage to seagrasses and substrate scour from float chains and anchors (Kennish 2002). Docks located in intertidal areas that are exposed during low tides result in vessels resting on the substrate, which may impact shellfish beds, SAV, and intertidal mudflats. Vessels operating in shallow water to access docks may cause a resuspension of bottom sediments and may physically disrupt aquatic habitats, such as bank and shoreline (Barr 1993) and SAV through "prop dredging" (Burdick and Short 1999). Barr (1993) identified a number of potential impacts to aquatic ecosystems from resuspension of sediments caused by vessel activity, including reductions in primary productivity (e.g., phytoplankton and SAV), alteration of temperature, dissolved oxygen and pH of the water, abrasion and clogging of fishes gill filaments, and reductions in egg development and the growth of some fishes and invertebrates. Glasby (1999) found that epibiota on pier pilings at marinas subject to shading were markedly different than those in surrounding rock reef habitats. Shading by overwater structures may be responsible for the observed reductions in juvenile fish populations found under piers and the reduced growth and survival of fishes held in cages under piers, when compared to open habitats (Able et al. 1998; Duffy-Anderson and Able 1999).

Increased erosion/accretion

Pilings can alter adjacent substrates with increased deposition of sediment from changes in current fields or shell material deposition from piling communities. Changes in substrate type can alter the nature of the flora and fauna native to a given site. Kearney et al. (1983) found that docks and pier walkways cause shading impacts to salt marsh vegetation, reduce plant root mat, and may lead to soil erosion in the area of the structures. In the case of pilings, native dominant communities typically associated with sand, gravel, mud, and eelgrass substrates may be replaced by communities associated with shell hash substrates (Penttila and Doty 1990; Nightingale and Simenstad 2001; Haas et al. 2002). In addition to impacts to eelgrass habitat from overwater structures, Penttila and Doty (1990) found that changes to current fields around structures caused altered sediment distribution and topography that created depressions along piling lines.

Changes in predator/prey interaction

Fish use visual cues for spatial orientation, prey capture, schooling, predator avoidance, and migration. The reduced-light conditions found under an overwater structure limit the ability of fish, especially juveniles and larvae, to perform these essential activities (Hanson et al. 2003). In addition, the use of artificial lighting on docks and piers creates unnatural nighttime conditions that can increase the susceptibility of some fish to predation and interfere with predator/prey interactions (Nightingale and Simenstad 2001).

Cumulative effects

While the effect of some individual overwater structures on fishery habitat may be minimal, the overall impact may be substantial when considered cumulatively. For example, although shading impacts on seagrasses may affect a relatively small area around overwater structures, fragmentation of seagrass beds along a highly developed shoreline or within a bay can be considerable. Fragmentation of seagrass habitat can lower the integrity of the remaining seagrass beds, leaving it more susceptible to other impacts (Burdick and Short 1999). The additive effect of these structures increases the overall magnitude of impact, reduces the ability of the habitat to support native plant and animal communities, and makes the habitat more susceptible to damage from storms and disease.

Conservation measures and best management practices for overwater structures (adapted from Hanson et al. 2003)

1. Use upland boat storage whenever possible to minimize need for overwater structures.
2. Locate overwater structures in sufficiently deep waters to avoid intertidal and shade impacts, to minimize or preclude dredging, to minimize groundings, and to avoid displacement of SAV, as determined by a preconstruction survey.
3. Design piers, docks, and floats to be multi-use facilities serving multiple homeowners in order to reduce the overall number of such structures and the nearshore habitat that is impacted.
4. Incorporate measures that increase the ambient light transmission under piers and docks. Some of these measures include: maximizing the height of the structure and minimizing the width of the structure to decrease shade footprint; grated decking material; using the fewest number of pilings necessary to support the structures to allow light into under-pier areas and minimize impacts to the substrate; and aligning piers, docks, and floats in a north-south orientation to allow the path of the sun to cross perpendicular to the length of the structure and reduce the duration of shading.
5. Encourage seasonal use of docks and off-season haul-out.
6. Avoid placing floating docks in areas supporting SAV. Locate floats in deep water to avoid light limitation and grounding impacts to the intertidal zone, and ensure that adequate water depth is available between the substrate and the bottom of the float throughout all tide cycles.
7. Incorporate float stops in dock proposals when it is impracticable or impossible to avoid placing floating docks in water deep enough to avoid contact with the bottom to avoid mechanical and/or hydraulic damage to the substrate from the float during low tides. Float stops should be designed to provide a minimum of 2 ft of clearance between the float and substrate to prevent hydraulic disturbances to the bottom. Greater clearances may be necessary in higher energy environments that experience strong wave action.
8. Conduct in-water work during the time of year when managed species and prey species are least likely to be impacted.
9. Avoid the use of treated wood timbers or pilings to the extent practicable. The use of alternative materials such as untreated wood, concrete, or steel is recommended. Concrete and steel pilings are generally considered to be less damaging, since they help reflect light under docks and generally do not release contaminants into the aquatic environment.
10. Orient artificial lighting on docks and piers such that illumination of the surrounding waters at night is avoided.
11. Address the cumulative impacts of past, present, and foreseeable future development projects on aquatic habitats by considering them in the review process for overwater structure projects.

Pile Driving and Removal

Pilings provide support for the decking of piers and docks; they function as fenders and dolphins to protect structures, support navigation markers, and are used to construct breakwaters and bulkheads. Materials used in pilings include steel, concrete, wood (both treated and untreated), plastic or a combination thereof, and they are usually driven into the substrate with impact hammers or vibratory hammers (Hanson et al. 2003). Impact hammers consist of a heavy weight that is repeatedly dropped onto the top of the pile, driving it into the substrate. Vibratory hammers utilize a combination of a stationary, heavy weight and vibration, in the plane perpendicular to the long axis of the pile, to force the pile into the substrate. While impact hammers are able to drive piles into most substrates (e.g., hardpan, glacial till), vibratory hammers are limited to softer, unconsolidated substrates (e.g., sand, mud, gravel). Piles can be removed by using a variety of methods, including vibratory hammer, direct pull, clamshell grab, or cutting/breaking the pile below the mudline. Vibratory hammers can be used to remove all types of pile, including wood, concrete, and steel. Broken stubs are often removed with a clamshell and crane. In other instances, piles may be cut or broken below the mudline, leaving the buried section in place (Hanson et al. 2003).

Sound energy impacts

Pile driving with impact hammers can generate intense underwater sound pressure waves that may adversely affect fish species and their habitats. These pressure waves have been shown to injure and kill fish (CalTrans 2001; Longmuir and Lively 2001). Injuries directly associated with pile driving include rupture of the swimbladder and internal hemorrhaging, but these have been poorly studied (CalTrans 2001).

Benthic habitat impacts

The extraction of piles can result in altered sediment composition and depressions in the bottom, which may cause erosion and loss of sediment. Bottom depressions may fill in with fine sediments and silt, changing the characteristics of the benthic habitat. Removal of piles may cause sediments to slough off and elevate the suspended sediment concentrations at the work area (Hanson et al. 2003). The subsequent sedimentation and turbidity can impact adjacent sensitive habitats, such as SAV.

Increased sedimentation/turbidity and contaminant releases

The primary adverse effect of removing piles is the suspension of sediments, which may result in harmful levels of turbidity and release of contaminants contained in those sediments. Contaminants contained within the sediments in the area of pilings can become available to aquatic plants and animals when pilings are extracted from the substrate. Sediment plumes may also be created around the pilings when they are installed, although it is usually much less than the turbidity created during removal. Some turbidity may be generated when piles are installed or removed with hydraulic jets, although this technique may not be widely used in the northeast coastal region. Vibratory pile removal tends to cause the sediments to slough off, resulting in relatively low levels of suspended sediments and contaminants (Hanson et al. 2003). Vibratory removal of piles may be preferable in some circumstances because it can be used on all types of piles, providing that they are structurally sound. Breaking or cutting the pile below the mudline may suspend only small amounts of sediment, providing the stub is left in place and little digging is required to access the pile. Direct pull or use of a clamshell to remove broken piles, however, may suspend large amounts of sediment and contaminants. When the piling is pulled from the substrate with these two methods, sediments clinging to the piling will slough off as it is raised through the water column, producing a potentially harmful plume of turbidity and/or contaminants. The use of a clamshell may suspend additional sediment if it penetrates the substrate while grabbing the piling (Hanson et al. 2003). For more information on turbidity and sedimentation, consult the chapters on Physical Effects: Water Intake and Discharge Facilities and Marine Transportation. Additional information on contaminant releases can be reviewed in the Chemical Effects: Water Discharge Facilities chapter.

Conservation measures and best management practices for pile driving and removal (adapted from Hanson et al. 2003)

1. Drive piles during low tide periods when substrates are exposed in intertidal areas.
2. Use a vibratory hammer to install piles, when possible. Under those conditions where impact hammers are required for reasons of seismic stability or substrate type, it is recommended that the pile be driven as deep as possible with a vibratory hammer prior to the use of the impact hammer.
3. Implement measures to attenuate the sound or minimize impacts to aquatic resources during piling installation. Methods to mitigate sound impacts include, but are not limited to, the following:
   a. Surround the pile with an air bubble curtain system or dewatered cofferdam.
   b. Drive piles during low water conditions for intertidal areas.
   c. Utilize appropriate work windows that avoid impacts during sensitive times of year (e.g., anadromous fish runs and spawning, larval, and juvenile development periods).
4. Remove creosote-coated piles completely rather than cutting or breaking off if the pile is structurally sound.
5. Minimize the suspension of sediments and disturbance of the substrate when removing piles. Measures to help accomplish this include, but are not limited to, the following:
   a. Remove piles with a vibratory hammer when practicable, rather than with the direct pull or clamshell method.
   b. Remove the pile slowly to allow sediment to slough off at or near the mudline.
   c. Hit or vibrate the pile first to break the bond between the sediment and pile to minimize the potential for the pile to break, as well as reduce the amount of sediment sloughing off the pile during removal.
   d. Encircle the pile or piles with a silt curtain that extends from the surface of the water to the substrate.
6. Fill all holes left by the piles with clean, native sediments, if possible.
7. Place piles on a barge equipped with a basin to contain all attached sediment and runoff water after removal. Creosote-treated timber piles should be cut into short lengths to prevent reuse, and all debris, including attached, contaminated sediments, should be disposed of in an approved upland facility.
8. Drive broken/cut stubs with a pile driver sufficiently below the mudline to prevent release of contaminants into the water column as an alternative to their removal.
9. Use seasonal restrictions to avoid impacts to habitat during species critical life history stages (e.g., spawning and egg development periods). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
10. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in the review process for pile driving projects.

Marine Debris

Marine debris is a chronic problem along much of the US coast, resulting in littered shorelines and estuaries and creating hazards for marine organisms. Marine debris consists of a large variety of anthropogenic materials such as generic litter, hazardous wastes, and discarded or lost fishing gear and can have varying degrees of negative effects on the coastal ecosystem (Hanson et al. 2003). It generally enters waterways indirectly through rivers and storm drains or by direct ocean dumping. Several laws and regulatory programs exist to prevent or control the disposal of industrial wastes and the release of marine debris from ocean sources, including commercial merchant vessels (e.g., galley waste and other trash), recreational boaters and fishermen, offshore oil and gas exploration and facilities, military and research vessels, and commercial fishing vessels (Cottingham 1988). Despite these laws and regulations, marine debris continues to adversely impact our waters (Hanson et al. 2003). See the Marine Transportation chapter for more information on marine debris.

Land-based sources of marine debris account for approximately 80% of the marine debris on the beaches and in the waters of the Gulf of Maine (Hoagland and Kite-Powell 1997), as well as other coastal areas of the United States (Hanson et al. 2003). Land-based debris can originate from a wide variety of sources, including combined sewer overflows and storm drains; storm-water runoff; landfills; solid waste disposal; manufacturing facilities; poorly maintained garbage bins; floating structures (i.e., docks and piers); and general littering of beaches, rivers, and open waters (Cottingham 1988; Hanson et al. 2003). Plastics account for 50-60% of marine debris collected from the Gulf of Maine (Hoagland and Kite-Powell 1997).

Entanglement and ingestion

Entanglement and ingestion of marine debris by marine species is known to affect individuals of at least 267 species worldwide, including 86% of all sea turtle species, 44% of all seabird species, and 43% of all marine mammal species (Laist 1997). Plastic debris may be ingested by seabirds, fish and invertebrates, sea turtles, and marine mammals, which can obstruct the animal's intestinal tract and cause infections and death (Cottingham 1988). A study of marine debris ingestion by seabirds in the southern Atlantic Ocean found that 73% of all birds sampled had ingested some type of marine debris, and plastics composed 66% of all debris occurrences (Copello and Quintana 2003).

Introduction of invasive species

Marine debris discarded from commercial cargo and recreational vessels are one of the primary methods of transporting nonindigenous marine life around the world, some of which have become invasive species that can alter the structure and function of aquatic ecosystems (Valiela 1995; Carlton 2001; Niimi 2004). Refer to the chapters on Marine Transportation, and Introduced/Nuisance Species and Aquaculture for more information on invasive species.

Contaminant releases and introduction of pathogens

The type of debris from land-based sources can include raw or partially treated sewage, litter, hazardous materials (e.g., PAH, paint, solvents), and discarded trash. The typical floatable debris from combined sewer overflows includes street litter, sewage containing viral and bacterial pathogens, pharmaceutical by-products from human excretion, and pet wastes. It may contain condoms, tampons, and contaminated hypodermic syringes, all of which can pose physical and biological threats to fishery habitat (Hanson et al. 2003). Toxic substances in plastics, for example, can persist in the environment and bioaccumulate through the food web and can kill or impair fish and invertebrates that use habitat polluted by these materials.

Conversion of habitat

Because of the wide range and diversity of sources and materials contributing to marine debris, the effects on aquatic habitats are likewise wide-ranging and diverse. Floating or suspended trash can directly affect fish and invertebrates that may consume or are entangled by the debris. Debris that settles to the bottom of rivers, estuaries, and open ocean areas may continue to cause environmental problems. Plastics and other materials with a large surface area can cover and suffocate sessile animals and plants. Debris can be transported by currents to other areas where it can become snagged and attached to benthic reefs, damaging these sensitive habitats.

Conservation measures and best management practices for marine debris (adapted from Hanson et al. 2003)

1. Require all existing and new commercial construction projects near the coast (e.g., marinas and ferry terminals, recreational facilities, boat building and repair facilities) to develop and implement refuse disposal plans.
2. Encourage proper trash disposal in coastal and ocean settings.
3. Provide resources to the public on the impact of marine debris and guidance on how to reduce or eliminate the problem.

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