NOAA Technical Memorandum NMFS-NE-209

Impacts to Marine Fisheries Habitat from Nonfishing Activities in the Northeastern United States

US DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
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Impacts to Marine Fisheries Habitat from Nonfishing Activities in the Northeastern United States

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PREFACE

The genesis of this report was a technical workshop held in Mystic, CT, on January 10-12, 2005 entitled "Workshop on Impacts to Coastal Fishery Habitat from Nonfishing Activities." The workshop and report were conceived by the Northeast Region Essential Fish Habitat Steering Committee which is composed of representatives from NOAA National Marine Fisheries Service Northeast Regional Office (NERO), NOAA National Marine Fisheries Service Northeast Fisheries Science Center (NEFSC), New England Fishery Management Council (NEFMC), Mid-Atlantic Fishery Management Council (MAFMC), and the Atlantic States Marine Fisheries Commission (ASMFC). The workshop was sponsored jointly by NOAA National Marine Fisheries Service, NEFMC and ASMFC.

The original intent of the workshop was to provide the necessary information to the NEFMC and MAFMC to assist them in updating the nonfishing impact analyses within their Fishery Management Plans as required by the Essential Fish Habitat (EFH) regulations. As work progressed, we realized that this information would be extremely useful to a much larger audience of agencies, consultants, and components of the public involved in marine and aquatic habitat assessment activities, and so this comprehensive report was developed. For this reason, the scope of impact assessment for this report was expanded to include a more general approach to coastal fishery habitat and is not limited to EFH. Our goal is to ensure that the best scientific information is available for use in making sound decisions with respect to the various environmental reviews and permitting processes conducted within the marine environment.

The comprehensive nature of this report required extensive collaboration among the authors, which includes NOAA National Marine Fisheries Service staff within the NERO Habitat Conservation Division and Headquarters Office of Habitat Conservation (OHC). We would like to thank the participants of the technical workshop who graciously provided their time and expertise towards identifying and assessing the range of impacts that threaten coastal resources in the northeast region of the United States (see appendix for list of participants). We would particularly like to thank the following individuals for their advice, time, and valuable assistance in the preparation and review of this report: Claire Steimle, Northeast Fishery Science Center (NEFSC) – Library Assistance; numerous staff of the NOAA Library; numerous reviewers, including Jen Costanza, Kathi Rodrigues, Dr. David Stevenson, and David Tomey– NOAA National Marine Fisheries Service, NERO; Jeanne Hanson – NOAA National Marine Fisheries Service, Alaska Regional Office; Joanne Delaney – NOAA National Marine Sanctuaries Program; and Ruth M. Ladd –US Army Corps of Engineers, New England District. In addition, we appreciate the advice provided by the technical and editorial reviewers at the NEFSC: Donna A. Busch, Dr. Jarita Davis, Dr. Ashok Deshpande, Dr. David Dow, Laura Garner, Dr. Jon Hare, Clyde L. MacKenzie, Jr., Donald G. McMillan, Dr. Thomas Noji, Dave Packer, and Dr. Robert Reid.

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Essential Fish Habitat Steering Committee

ACRONYMS AND ABBREVIATIONS

ACZA	ammoniacal copper zinc arsenate
ANS	aquatic nuisance species
ATOC	Acoustic Thermometry of Ocean Climate
AVS	acid volatile sulfides
BMP	best management practice
BOD	biological oxygen demand
C	Celsius
CCA	chromated copper arsenate
cm	centimeters
CSOs	combined sewer overflows
CWA	Clean Water Act
dB	decibel
DC	direct current
DDE	dichlorodiphenyl dichloroethylene
DDT	dichlorodiphenyl trichloroethane
DNA	deoxyribonucleic acid
DO	dissolved oxygen
ELMR	Estuarine Living Marine Resources
EMF	electromagnetic field
EEZ	Exclusive Economic Zone
EFH	essential fish habitat
ESP	electric service platform
F	Fahrenheit
FMP	fishery management plan
ft	feet or foot
GIS	geographic information system
HAB	harmful algal bloom
HARS	Historic Area Remediation Site
HEA	Habitat Equivalency Analysis
Hz	Hertz
IPCC	Intergovernmental Panel on Climate Change
km	kilometer
L	liter
LC50	chemical concentration which causes the death of 50% of the experimental test animals
LFAS	low frequency active sonar
LNG	liquefied natural gas
LWD	large woody debris
m	meter
MARPOL	International Convention for the Prevention of Pollution from Ships
ml	milliliter
mm	millimeter
MMS	Minerals Management Service
MOA	Memorandum of Agreement
MPRSA	Marine Protection, Research, and Sanctuaries Act

MSA	Magnuson-Stevens Fishery Conservation and Management Act
MSD	marine sanitation device
NATO	North Atlantic Treaty Organization
NEFMC	New England Fishery Management Council
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NPDES	National Pollutant Discharge Elimination System
NPS	nonpoint source
NS&T	National Status and Trends
NRC	National Research Council
OCS	Outer Continental Shelf
PAH	polycyclic aromatic hydrocarbons
PCB	polychlorinated biphenyl
pH	the measure of acidity or alkalinity of a solution
POP	persistent organic pollutant
PPCP	pharmaceuticals and personal care products
ppt	parts per thousand
s	second
SAV	submerged aquatic vegetation
SCUBA	self-contained underwater breathing apparatus
SURTASS	Surveillance Towed Array Sensor System
TBT	tributyltin
THC	thermohaline circulation
TOC	total organic carbon
TOY	time-of-year
US ACE	United States Army Corps of Engineers
US EPA	United States Environmental Protection Agency
μA	microamp
μg	micrograms
μV	microvolt

GLOSSARY OF TERMS

alevins	young salmonid fish distinguished by an attached yolk sac
alkalinity	the quantitative capacity of water to neutralize an acid
amnesic shellfish poisoning	caused by domoic acid, an amino acid, as the contaminant of shellfish
anadromous	migrating from the sea to fresh water to spawn
anoxia	complete absence of oxygen in aquatic habitats
anthropogenic	effects, processes, or materials that are derived from human activities
aquatic nuisance species	introduced (nonnative) organisms that produce harmful impacts on aquatic natural resources
autotrophic	a class of organism that produces organic compounds from carbon dioxide as a carbon source, by using either light or reactions of inorganic chemical compounds, as a source of energy; also known as a producer in a food chain
beach nourishment	the replacement of sand on an eroded beach from an outside source such as an offshore sand deposit, an inlet tidal delta, or an upland sand quarry
benthic	in or associated with the seafloor
benthos	organisms living on, in, or near the bottom of water bodies
bioaccumulation	the accumulation of substances, such as pesticides, methylmercury, or other organic chemicals in an organism or part of an organism
biocide	a chemical substance capable of killing different forms of living organisms (e.g., pesticide)
borrow pit	an excavation dug to provide material for fill elsewhere; used in aggregate or mineral mining and in beach nourishment
carcinogenic substance	cancer causing agent
catadromous	migrating from fresh water to the sea to spawn
climax community	a community of organisms the composition of which is more or less stable and in equilibrium with existing natural environmental conditions

creosote	a brownish oily liquid consisting chiefly of aromatic hydrocarbons obtained by distillation of coal tar and used especially as a wood preservative
cytolysis	the dissolution or destruction of a cell
demersal	dwelling at or near the bottom of a body of water
denitrification	the process of reducing nitrate and nitrite (highly oxidized forms of nitrogen available for consumption by many groups of organisms) into gaseous nitrogen
desalination	any of several processes that remove the excess salt and other minerals from water in order to obtain fresh water suitable for consumption or irrigation
diadromous	migratory between fresh and salt waters
diel	occurring on a daily basis, such as vertical migrations in some copepods and fish
dissolved oxygen	a measure of the amount of gaseous oxygen dissolved in an aqueous solution
echolocation	the biological sonar used by dolphins and whales for navigation and foraging
ecosystem	a natural unit consisting of all plants, animals, and microorganisms in an area functioning together with all the nonliving physical factors of the environment
endocrine disruptor	an exogenous (outside the body) agent that interferes with the production, release, transport, metabolism, binding, action, or elimination of natural hormones in the body responsible for the maintenance of homeostasis and the regulation of developmental processes
entrainment	the voluntary or involuntary movement of aquatic organisms from the parent water body into a surface diversion or through, under, or around screens, resulting in the loss of the organisms from the population
epibiota	attached plants and animals that settle and grow on natural or artificial surfaces
epipelagic	part of the open ocean comprising the water column from the surface down to approximately 200 meters
estrogenic substances	compounds that mimic female steroid hormones or inhibit male steroid hormones

eutrophication	enrichment of nutrients causing excessive plant growth that can reduce oxygen concentration and kill aquatic organisms
extirpate	to eliminate completely certain populations within the range of a given species
gas supersaturation	the overabundance of gases in turbulent water, such as at the base of a dam spillway, which can cause a fatal condition in fish
genotype	the genetic constituents in each cell of an organism
glacial till	an unsorted, unstratified mixture of fine and coarse rock debris deposited by a glacier
hardpan	a layer of hard subsoil or clay
headwater	the source of water for a river or stream
heterotrophic	a class of organism that requires organic substrates to get its carbon for growth and development; also known as a consumer in the food chain
hydrophobicity	the property of being water-repellent or tending to repel and not absorb water
hyperplasia	an increase in the number of the cells causing an organ or tissue to increase in size
hypersaline	salinity well in excess of that of sea water
hypertrophy	an increase in the size of an organ or in a select area of the tissue caused by an increase in the size of cells, while the number stays the same
hyporheic zone	saturated zone under a river or stream, composed of substrates with interstices filled with water
hypoxia	a low oxygen condition in aquatic habitats
ichthyoplankton	eggs and larvae of fish that drift in the water column
immunotoxicity	adverse effects on the functioning of the immune system that result from exposure to chemical substances
impingement	involuntary contact and entrapment of aquatic organisms on the surface of intake screens caused by the approach velocity exceeding the swimming capability of the organism
littoral zone	also called the intertidal zone, it lies between the high tide mark and the low tide mark

lotic	pertaining to running water, as opposed to lentic or still waters
macroinvertebrate	an animal lacking a backbone and visible without the aid of magnification
meroplankton	organisms that are planktonic for only a part of their life cycles, usually the larval stage
methylmercury	formed from inorganic mercury by the action of anaerobic organisms that live in aquatic systems and sediments; a bioaccumulative environmental toxin
mutagenic	agent causing genetic mutations
neurotoxic shellfish poisoning	shellfish poisoning caused by exposure to a group of polyethers called brevetoxins
oligohaline	brackish water with a salinity of 0.5 to 5.0 parts per thousand
organochlorides	a large, diverse group of organic compounds containing at least one covalently bonded chlorine atom, some of which are considered to be persistent organic pollutants and are harmful to the environment (e.g., PCB, DDT, chlordane, dioxins)
organometal	A member of a broad class of compounds whose structures contain both carbon and a metal (e.g., methylmercury and tetra-ethyl lead) - persistent and bioaccumulative environmental toxins
osmoregulation	the physiological mechanism for the maintenance of an optimal and constant fluid concentration and pressure in and around the cells
paralytic shellfish poisoning	caused by a group of toxins elaborated by planktonic algae (dinoflagellates, in most cases) upon which the shellfish feed
parr	developmental stage of young salmonid fish that follows the fry and lasts for one to three years in their native stream before becoming smolts
pelagic	associated with the water column
phytoplankton	microscopic plants that drift in the water column
planktivorous	feeding on plankton (e.g., most fish larvae and many pelagic fishes)
pycnocline	a layer of rapid change in water density with depth mainly caused by changes in water temperature and salinity
radionuclide	an atom with an unstable nucleus that can occur naturally but can also be artificially produced; also known as radioisotope

redd	an area in gravel where salmonids bury their eggs; also known as nests or gravel nests
reflective turbulence	changes in water velocity caused by wave energy reflection from solid structures in the nearshore coastal area, resulting in increased turbidity
riparian	land directly adjacent to a stream, lake, or estuary
salmonid	belonging to, or characteristic of the family salmonidae, which includes salmon, trout, and whitefish
sedimentation	the deposition by settling of suspended solids
siltation	sedimentary material consisting of very fine particles intermediate in size between sand and clay
smoltification	a suite of physiological, morphological, biochemical, and behavioral changes, including development of the silvery color of adults and a tolerance for seawater, that take place in young salmonid fish they prepare to migrate downstream and enter the sea
soil infiltration	the passage of water through the surface of the soil into the soil profile via pores or small openings
spermatogenesis	the process by which male gametes are formed in many sexually reproducing organisms
synergistic	combined effects being greater than the sum of individual effects
tailwater	an area immediately below a dam where the river water is cooler than normal and rich in nutrients
tannins	astringent, plant polyphenol compounds that bind and precipitate proteins; used in manufacturing inks and dyes
thermocline	a vertical temperature gradient in some layer of a body of water that is appreciably greater than the gradients above and below it
time-of-year restrictions	seasonal constraints for dredging to avoid or minimize impacts of sensitive periods in the life-history of an organism, such as spawning, egg development, and migration
tonne	sometimes referred to as a metric tonne, the measurement of mass equal to 1,000 kilograms
trophic level	the position that an organism occupies in a food chain

turbidity	the cloudiness or haziness of water caused by individual particles or suspended solids
volitional fish passage	any type of structure that provides fish passage over, through, or around an obstruction in a river or stream (e.g., dam) that can be successfully achieved under the fish's own power (as opposed to trap and truck methods)
xenobiotic	a chemical which is found in an organism but which is not normally produced or expected to be present in it (e.g., pollutants, such as dioxins or PCB congeners)

INTRODUCTION

Report Purpose

This report stems from a workshop entitled "Technical Workshop on Impacts to Coastal Fishery Habitat from Nonfishing Activities," which was held January 10 – 12, 2005 in Mystic, CT. The workshop convened a group of experts in the field of environmental, marine habitat, and fisheries impact assessment from federal and state government agencies. The goals of the workshop were to: (1) describe known and potential adverse effects of human induced, nonfishing activities on fisheries habitats; (2) create a matrix of the degree of impacts associated with various activities in riverine, estuarine, and marine habitats; and (3) develop a suite of best management practices (BMPs) and conservation recommendations that could be used to avoid or minimize adverse impacts to fisheries habitats. Refer to Chapter One-Technical Workshop on Impacts to Coastal Fisheries Habitat from Nonfishing Activities, for a detailed summary of the technical workshop.

The general purpose and goals of this report are to:

1. Identify human activities that may adversely impact Essential Fish Habitat (EFH) and other coastal fishery habitat. As Stevenson et al. (2004) characterized the impacts to EFH from fishing activities in the northeast region, the focus of this report is on nonfishing activities.
2. Review and characterize existing scientific information regarding human-induced impacts to EFH and other coastal fishery habitat.
3. Provide BMPs and conservation measures that can be implemented for specific types of activities that avoid or minimize adverse impacts to EFH and other coastal fishery habitat.
4. Provide a comprehensive reference document for use by federal and state marine resource managers, permitting agencies, professionals engaged in marine habitat assessment activities, the regulated community, and the public.
5. Ensure that the best scientific information is available for use in making sound decisions with respect to project planning, environmental assessment, and permitting.

The National Oceanic and Atmospheric Administration's (NOAA) National Marine Fisheries Service is mandated to protect and conserve fishery resources, an activity which includes engaging in consultation with federal agencies on actions that may adversely affect NOAA's trust resources. It is anticipated that the information in this report will be used to assist federal agencies and their consultants in the preparation of impact assessments for EFH and other NOAA's trust resources. In addition, this report will assist National Marine Fisheries Service habitat specialists in: (1) reviewing proposed projects; (2) considering potential impacts that may adversely affect NOAA's trust resources; and (3) providing consistent and scientifically supported conservation recommendations. This report will also provide insight for the public and the regulated community on the issues of concern to National Marine Fisheries Service along with approaches to design and implementation of projects that avoid and minimize adverse effects to fish habitat.

Organization of the Report

The document is organized by activities that may potentially impact EFH and other fishery habitat occurring in riverine, estuarine/coastal, and marine/offshore areas. Chapter One describes the technical workshop that was conducted and presents the results of those discussions and habitat impact evaluations. The major activities that were identified as impacting these three habitat areas include:

- coastal development
- energy-related activities
- alteration of freshwater systems
- marine transportation
- offshore dredging and disposal
- physical and chemical effects of water intake and discharge facilities
- agriculture and silviculture
- introduced/nuisance species and aquaculture
- global effects and other impacts

Each subsequent chapter characterizes impacts associated with the major activities listed above. Each chapter describes the adverse effects of various activities on fishery habitat and the species associated with those habitats, provides the scientific references to support those findings, and concludes with best management practices or conservation recommendations that could be implemented to avoid or minimize those particular adverse effects. Although the activities and effects identified in the technical workshop are reflected in the appropriate chapter, the reader may notice some minor variation in the order and content if the chapter author(s) failed to locate information in the literature on a specific topic or believed additional discussion of effects were warranted. The preparers of this report have attempted to summarize the current knowledge of impacts and effects from existing and potential activities in the coastal areas of the northeast region of the United States. However, the reader should not consider the information in the report as comprehensive for all activities and impacts on fishery habitats. For more detailed analyses and understanding, the reader should refer to the cited references and the most current literature regarding specific activities and impacts.

The BMPs and conservation measures provided in this report are designed to minimize or avoid the adverse effects of human activities on fishery habitat and to promote the conservation and enhancement of fishery habitat. The BMPs and conservation measures provided in this report reflect many of the conservation principals recommended in Hanson et al. (2003). These general principles include: (1) nonwater-dependent actions should not be located in fishery habitat if such actions may have adverse impacts on those resources; (2) activities that may result in significant adverse affects on fishery habitat should be avoided where less environmentally harmful alternatives are available; (3) if alternatives do not exist, the impacts of these actions should be minimized; and (4) environmentally sound engineering and management practices should be employed for all actions that may adversely affect fishery habitat.

The conservation measures and BMPs included with each activity present a series of practices or steps that can be undertaken to avoid or minimize impacts to fishery habitats. Not all of these suggested measures are applicable necessarily to any one project or activity that may adversely affect habitat. More specific or different measures based on the best and most current scientific information may be developed as part of the project planning or regulatory processes. The conservation recommendations and BMPs provided represent a generalized menu of the types of measures that can contribute to the conservation and protection of fishery habitat and other coastal aquatic habitats.

The final chapter contains a brief discussion of the purpose and application of compensatory mitigation used to offset adverse effects on fishery habitat. We have chosen to include a discussion on compensatory mitigation in its own chapter because its application is not generally considered a best management practice or a recommendation to conserve fishery habitat. Instead, compensatory mitigation is a method of offsetting adverse effects after they have occurred. For that reason, compensatory mitigation should be considered only after all measures to avoid and then minimize impacts have been exhausted. Compensatory mitigation should never be used as a first-line conservation measure.

Some of the impact types described in one chapter may also be found in other chapters containing similar impacts or activities. Therefore, the reader may find some redundancy in the various chapters. Because the report's focus was to describe the impacts to living marine resources and habitats associated with specific anthropogenic activities and often have similar adverse affects on living marine resources, some redundancy in the descriptions of impacts between various chapters was unavoidable.

Characterization of Habitat in the Northwest Atlantic Ocean

The general focus of this report pertains to effects on marine, estuarine, and diadromous fishes and their habitats. However, the preparers of the report have attempted to provide a broad perspective of coastal aquatic habitat and the organisms that depend upon those habitats in an ecosystem context. Although the report often refers to "fishery habitat" or "fish," the definitions of these resources should not necessarily be limited to any particular regulatory or management mandate, such as EFH. The authors have attempted to include information on known or potential impacts that may affect the ecological functions and values for habitats for all species of fish and invertebrates. Because the focus of this report is on impacts to fish and fishery habitats, we have included only limited discussions on impacts specific to marine mammals and sea turtles.

Habitats provide living things with the basic life requirements of nourishment and shelter (Stevenson et al. 2004). According to Deegan and Buchsbaum (2005), a habitat includes the physical environment, the chemical environment, and the many organisms that compose a food web. This report employs a similarly broad definition to discuss the multitude of adverse effects on habitats in the coastal northeastern United States. For example, the quality of the water in which aquatic organisms live, feed, and reproduce is a facet of their habitat, and the presence of contaminants or alterations to the water has important implications on the health of those organisms. Habitats may also provide a broader range of benefits to the ecosystem, such as the way seagrasses physically stabilize the substrate and help recirculate oxygen and nutrients (Stevenson et al. 2004). These habitats do not exist in isolation but are linked through ecological and oceanographic processes that are a part of the larger ecosystem. For example, the movement of the water plays a major role in the interconnection of habitats by transporting nutrients, food, larvae, sediments, and pollutants among them (Tyrrell 2005).

The northwest Atlantic Ocean includes a broad range of habitats with varying physical and biological properties extending from the cold waters of the Gulf of Maine south to the more temperate climate of the Mid-Atlantic Bight. In this region, the oceanographic and physical processes interact to form a network of expansively to narrowly distributed habitat types (Stevenson et al. 2004). The offshore component of this region, also known as the Northeast US Continental Shelf Ecosystem (Sherman et al. 1996), is composed of four distinct subregions: the Gulf of Maine, Georges Bank, the Mid-Atlantic Bight, and the continental slope (Stevenson et al. 2004). In addition, the region contains freshwater rivers and streams that flow towards the sea into numerous bays and estuaries that serve as important refuge and nursery areas for marine species. This report focuses on the three major systems composing this ecosystem: riverine, estuarine/nearshore, and marine/offshore environments.

The habitat classifications described by Jury et al. (1994) and adopted by NOAA as a national standard for organizing its Estuarine Living Marine Resources (ELMR) program's database are useful because they facilitate consideration of physico-chemical interactions in water quality and habitat impacts and implications for aquatic organisms. Conveniently, this approach also aligns with ambient suspended sediment and particulate loads because maximum turbidity zones of temperate, well-mixed estuaries typically coincide with low salinity regions (Herman and Heip 1999). Accordingly, this report has used the three ELMR salinity ranges developed for coastal aquatic habitats to describe "riverine" (<0.5 ppt), "estuarine/nearshore" (0.5-25.0 ppt), and "marine/offshore" (>25.0 ppt) conditions.

Riverine

Riverine habitats, located along the coast of New England and the Mid-Atlantic, provide essential habitat to anadromous and catadromous ("diadromous") fishes. These habitats include freshwater streams, rivers, streamside wetlands, and the banks and associated vegetation that may be bordered by other freshwater habitats (NEFMC 1998). Depending upon the local water velocity and other physical characteristics, riverine systems may include a variety of benthic substrates ranging from exposed bedrock, cobble, and other hard bottom types to extremely unconsolidated, soft bottom material. These features have a great bearing on the fish and invertebrate species that may be present.

Riverine habitats serve multiple purposes including migration, feeding, spawning, nursery, and rearing functions. An important component of a river system also includes the riparian corridor. The term "riparian" refers to the land directly adjacent to a stream, lake, or estuary. A healthy riparian area has vegetation supporting prey items (e.g., insects); contributes necessary nutrients; provides large woody debris that creates channel structure and cover for fish; and provides shade, which controls stream temperatures (NEFMC 1998).

Estuarine/nearshore

Estuaries are the bays and inlets influenced by both the ocean and rivers that serve as the transition zone between fresh and salt water. In the northeastern United States, they also may include the substantial inland reaches of large river systems where salinities exceed 0.5 ppt. For instance, ocean tides influence the lower 153 miles of the Hudson River, and oligohaline salinities (0.5 pp – 5 ppt) can extend well inland under low flow conditions. Typically, the northernmost intrusion of brackish water does not extend past the city of Poughkeepsie, nearly 75 miles north of The Battery at the southern tip of Manhattan, NY.

Estuaries support a community of plants and animals that are adapted to the zone where fresh and salt waters mix. Estuarine habitats fulfill fish and wildlife needs for reproduction, feeding, refuge, and other physiological necessities (NEFMC 1998). Coastal and estuarine features such as salt marshes, mud flats, rocky intertidal zones, sand beaches, and submerged aquatic vegetation are critical to inshore and offshore habitats and fishery resources of the northeastern United States (Stevenson et al. 2004). For example, healthy estuaries include eelgrass beds that protect young fish from predators, provide habitat for fish and wildlife, improve water quality, and can help stabilize sediments. In addition, mud flats, high salt marshes, and saltmarsh creeks also provide productive shallow water habitat for epibenthic fishes and decapods. Inshore habitats are dynamic and heterogeneous environments that support the majority of marine and anadromous fishes at some stage of development (NEFMC 1998).

Marine/offshore

The Gulf of Maine is an enclosed coastal sea, characterized by relatively cold waters and deep basins with a patchwork of various sediment types. Georges Bank is a relatively shallow coastal plateau that slopes gently from north to south and has steep submarine canyons on its eastern and southeastern edge. It is characterized by highly productive, well-mixed waters and strong currents. The Mid-Atlantic Bight is composed of the sandy, relatively flat, gently sloping continental shelf from southern New England to Cape Hatteras, NC. The continental slope begins at the continental shelf break and continues eastward with increasing depth until it becomes the continental rise. It is fairly homogenous, with exceptions at the shelf break, some of the canyons, the Hudson Shelf Valley (offshore New York), and areas of glacially rafted hard bottom (Stevenson et al. 2004).

The offshore benthic habitat features include sand waves, shell aggregates, gravel beds, boulder reefs, and submerged canyons which provide nursery areas for many fish species (NEFMC 1998). Many marine organisms inhabit the stable offshore environment for multiple stages of their life history.

Essential Fish Habitat

In 1996, the US Congress declared that "one of the greatest long-term threats to the viability of the commercial and recreational fisheries is the continuing loss of marine, estuarine, and other aquatic habitats. Habitat considerations should receive increased attention for the conservation and management of fishery resources of the United States" (Magnuson-Stevens 1996, sec. 2.a.9.). Along with this declaration, Congress added new habitat conservation provisions to the Magnuson-Stevens Fishery Conservation and Management Act (MSA), the federal law that governs US marine fisheries management. The MSA requires that fishery management plans describe and identify essential fish habitat, minimize adverse effects on habitat caused by fishing, and identify other actions to encourage the conservation and enhancement of such habitat. Essential fish habitat has been defined as "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" (Magnuson-Stevens 1996, sec. 3.10.).

The MSA also requires federal agencies to consult with the Secretary of Commerce, acting through the NOAA's National Marine Fisheries Service, on all actions authorized, funded or undertaken, or proposed to be authorized or undertaken by the agency, that may adversely affect EFH. The process developed for conducting these EFH consultations is described in the EFH regulations (50 CFR §600.905 – 920). In summary, federal agencies initiate consultation by preparing and submitting an EFH assessment to the National Marine Fisheries Service that describes the action, analyzes the potential adverse effects of the action on EFH, and provides the agency's conclusions regarding the effects of the action on EFH. In response, the National Marine Fisheries Service provides the agencies with conservation recommendations to conserve EFH by avoiding, minimizing, mitigating, or otherwise offsetting the adverse effects to EFH. Adverse effect is defined as any impact which reduces the quality and/or quantity of EFH. Adverse effects may 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. Adverse effects may be site-specific or habitat-wide, including individual, cumulative, or synergistic consequences of actions [50 CFR §600.910(a)]. This broad definition of adverse effects has been employed in this report to describe the various activities and sources of nonfishing impacts that can degrade fisheries habitat.

Once the National Marine Fisheries Service provides conservation recommendations, the federal action agencies must provide a detailed response in writing to the Nationial Marine Fisheries Service. The response must include measures proposed for avoiding, mitigating, or offsetting the impact of a proposed activity on EFH. If the federal action agency chooses not to adopt National Marine Fisheries Service's conservation recommendations, it must explain its reasons for not following the recommendations.

Impacts to Habitat

Habitat alteration and disturbance occur from natural processes and human activities. Deegan and Buchsbaum (2005) placed human impacts to marine habitats into three categories: (1) permanent loss; (2) degradation; and (3) periodic disturbance. Permanent loss of habitat can result from activities such as wetland filling, coastal development, harbor dredging, and offshore mining operations (Robinson and Pederson 2005). Habitat degradation may be caused by physical changes, such as increased suspended sediment loading, overshadowing from new piers and wharves, as well as introduction of chemical contamination from land-based human activities (Robinson and Pederson 2005). Periodic disturbances are created by activities such as trawling and dredging for fish and shellfish and maintenance dredging of navigation channels.

The primary differences between these three categories are that permanent loss is irreversible, habitat degradation may or may not be reversible, and periodic disturbance is generally reversible once the source of disturbance is removed (Deegan and Buchsbaum 2005). These authors indicate that recovery times for degraded habitat depend on the nature of the agent causing the degradation and the physical characteristics of the habitat. Recovery times for periodic disturbances will vary depending on the intensity and periodicity of the disturbance and the nature of the habitat itself. Natural fluctuations in habitats, such as storms and long-term climatic changes, occur independently of anthropogenic impacts.

Deegan and Buchsbaum (2005) state that "habitat quantity is a measure of the total area available, while habitat quality is a measure of the carrying capacity of an existing habitat." Generally, activities that lead to a permanent loss of habitat reduce the quantity of habitat, whereas habitat degradation and periodic disturbances result in a loss of habitat quality. The reduced quality of habitat (e.g., siltation, eutrophication, and alteration of salinity and food webs) may be equally damaging to the biological community as a loss in habitat quantity. As Deegan and Buchsbaum (2005) have noted, "the physical structure of the habitat does not need to be directly altered for negative consequences to occur." For example, reductions in water quality can impair and limit the ability of aquatic organisms to grow, feed, and reproduce.

The end point of gradual declines in the quality of habitat can be the complete loss of habitat structure and function (Deegan and Buchsbaum 2005). Losses of habitat quantity and quality may reduce the ability of a region to support healthy and productive fish populations. From the population perspective, the loss of habitat quantity and quality creates stresses on a population. Populations that are stressed by one or more factors can be more susceptible to stresses caused by other factors (Robinson and Pederson 2005), resulting in cumulative effects. These authors call for a holistic approach to fishery management: one that considers the interactions among exploitation, contaminants, and habitat degradation on various fish stocks.

Lotze et al. (2006) show that severe depletion of marine resources (i.e., 50% reduction in abundance level) first began with the onset of European colonization. This study found that 45% of species depletions and 42% of extinctions involved multiple human impacts, mostly exploitation and habitat loss. Seventy eight percent of resource recoveries are attributed to both habitat protection and restricted exploitation, while only 22% of recoveries are attributed to reduced exploitation alone (Lotze et al. 2006). These authors also conclude that reduced exploitation, increased habitat protection, and improved water quality need to be considered together and that the cumulative effects of multiple human interventions must be included in both management and conservation strategies.

References for Introduction

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Herman P, Heip C. 1999. Biogeochemistry of the MAximum TURbidity zone of Estuaries (MATURE): some conclusions. Journal of Marine Systems 22:89-104.

Jury S, Field J, Stone S, Nelson D, Monaco M. 1994. Distribution and abundance of fishes and invertebrates in North Atlantic estuaries. Silver Spring (MD): NOAA/NOS Strategic Environmental Assessments Division. ELMR Report 13. 221 p.

Lotze HK, Lenihan HS, Bourque BJ, Bradbury RH, Cooke RG, Kay MC, Kidwell SM, Kirby MX, Peterson CH, Jackson JBC. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312:1806-9.

Magnuson-Stevens Fishery Conservation and Management Act. 1996. P.L. 94-265, 16 U.S.C. 1801 et seq. [Internet]. [cited 2008 Jul 31]. Available from: http://www.nmfs.noaa.gov/sfa/magact/.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol 1.

Robinson WE, Pederson J. 2005. Contamination, habitat degradation, overfishing - An "either-or" debate? In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 1-10.

Sherman, K, Jaworski NA, Smayda TJ, editors. 1996. The Northeast Shelf Ecosystem: assessment, sustainability, and management. Cambridge (MA): Blackwell Science. 564 p.

Stevenson D, Chiarella L, Stephan D, Reid R, Wilhelm K, McCarthy J, Pentony M. 2004. Characterization of the fishing practices and marine benthic ecosystems of the Northeast U.S. Shelf, and an evaluation of the potential effects of fishing on essential fish habitat. Woods Hole (MA): National Marine Fisheries Service, Northeast Fisheries Science Center, NOAA Technical Memorandum NMFS-NE-181. 179 p.

Tyrrell MC. 2005. Gulf of Maine marine habitat primer. [Internet]. Gulf of Maine Council on the Marine Environment. vi + 54 p. [cited 2008 Jul 2]. Available from: www.gulfofmaine.org/habitatprimer/.

CHAPTER ONE: TECHNICAL WORKSHOP ON IMPACTS TO COASTAL FISHERIES HABITAT FROM NONFISHING ACTIVITIES

Introduction

A technical workshop was hosted by the Northeast Region Essential Fish Habitat Steering Committee on January 10-12, 2005 in Mystic, CT, to seek the views and recommendations of approximately 40 scientists, resource managers, and other marine resource professionals on threats to fishery habitat from nonfishing activities in the northeast coastal region. The participants of the workshop, entitled *Technical Workshop on Impacts to Coastal Fishery Habitat from Nonfishing Activities*, were federal and state environmental managers and regulators, as well as individuals from academic institutions and other organizations that have expertise and knowledge of various human-induced impacts on coastal environmental resources. A list of workshop participants and their affiliations is provided in the appendix of this report. The workshop's primary purpose was to convene marine resource professionals to review and evaluate existing information on nonfishing impacts for the purpose of updating, as necessary, fishery management plans under the New England and Mid-Atlantic Fishery Management Councils. In addition, the National Marine Fisheries Service sought to develop a nonfishing impacts reference document for use by professionals engaged in marine habitat assessment, permitting agencies, and state and federal marine resource managers. The information gathered during the workshop was used by the Northeast Region's Habitat Conservation Division staff to prepare selected chapters in the report. In general, the activities and effects contained within the various chapters of this report reflect the categories of activities and effects evaluated and discussed during the workshop.

The specific goals/tasks of the workshop included:

1. Identify all known and potential adverse effects for each category of nonfishing activity by life history strategies or stages (i.e., benthic/demersal and pelagic) and ecosystem strata (i.e., riverine, estuarine, and marine). This list of activities may also include adverse impacts to identified prey species or other specific life history requirements for species.
2. Create a matrix of nonfishing impacts for life history strategies/stages and ecosystem strata and ask the participants of the workshop to score the severity of each impact by using a relative scoring method.
3. Develop a suite of conservation measures and best management practices (BMPs) intended to avoid and minimize the adverse effects on fishery habitat and resources.
4. Identify possible information and data limitations and research needs in assessing impacts on fishery habitat or measures necessary to avoid and minimize those impacts.

Conservation measures were, to the extent possible, based on methods and technologies that have been evaluated through a scientific, peer-reviewed process. The intent was to develop recommendations that provide resource managers and regulators with specific methods and technologies yet have flexibility in their applications for various locations or project types. Ideally, providing a suite of conservation measures appropriate for various activities would give the end user several options of recommendations to consider.

Based upon the results of the workshop and effects scoring, some recommended research needs were developed. Identified research needs included basic life history requirements for some species and habitat types, physiological and biochemical responses of organisms to various physical and chemical perturbations and stressors, and technological advances in understanding or solutions to impact assessment and mitigation. Refer to the Conclusions and Recommendations chapter at the end of this report for a discussion on recommended research.

The format of the two-day workshop consisted of a series of breakout sessions, attended by the workshop participants, which represented the primary categories of nonfishing activities believed to threaten fishery resources and habitats in the northeast coast. There were ten separate breakout sessions conducted during the workshop, which are reflected in the chapters of this report. For each of the breakout sessions, a matrix of activities and known or potential adverse effects to fishery habitat, prepared by the workshop organizers, was reviewed by the workshop participants. The participants were encouraged to openly discuss and evaluate the relevance and significance for each of the activities and effects and to provide any additional activities and effects not included in the matrix. A large number of nonfishing activities occur within the coastal region and have a wide range of effects and intensities on fishery habitat. Each activity type and effect identified was evaluated in the context of life history strategies or stages (i.e., benthic and demersal) and ecosystem type or strata (i.e., riverine, estuarine/nearshore, and marine/offshore), in order to identify the importance of those factors. Following an open discussion, the participants were asked to score, by life history strategies/stages and ecosystem strata, the various activities and adverse effects on the impact matrix. In addition, participants were asked to include specific and relevant "conservation recommendations" and BMPs to avoid and minimize adverse effects to fishery habitat and resources.

On the last day of the workshop, the participants engaged in an informal discussion on the significance of cumulative effects and how multiple and additive effects can influence impacts to fishery habitat and resources. While the discussions were general in nature and few specifics of cumulative effects were discussed, there was a general agreement that cumulative effects are important and should play a larger role in assessment of habitat impacts. We found that the scores provided by the participants in the impact matrices for most breakout sessions to be relatively consistent throughout. While the variability in scores for some impact categories was high, we believe that the mean and median values for most effects' scores provide an accurate reflection of professional judgment by the participants. The relatively high variability in the scores of some activity types and effects may be due to varying interpretations of ecosystem strata and life history strategies or stages by the participants.

Effects Scoring System

Because one workshop goal was to assess the severity or degree of threat for known and potential impacts to fishery habitats, the workshop organizers strived to develop a semiquantitative scoring system that could measure the relative impacts for each activity and effect based upon the professional judgment of the participants. Developing defined values for measuring the significance of adverse effects for an activity is difficult and can depend upon the type of habitat being affected; the characteristic, intensity, and duration of the activity and disturbance; and a number of natural physical, chemical, and biological processes that may be occurring in the area and at the time of the activity. For this reason, the workshop organizers chose a semiquantitative scoring system with a range from 0 to 5, with a 1 being the lowest impact and a 5 being the highest impact. A "0" was used if an impact is not expected to occur or is not applicable, and a "UN" (unknown) was used if the participant does not know the degree of impact for a particular activity.

We believe that a relative scoring method that allows for flexibility and professional judgment in assigning a value for an effect is better than an absolute scoring system that has discreet and predefined values. Using a relative scoring range of 0 through 5 provided the participants a choice from a continuum of impact values for each effect and avoids the difficulty in finding consensus for the definition of predefined values. We then calculated the mean and median values of each effect and assigned a qualitative value of the threat for each effect by using the following criteria:

If either the mean or median value was greater than or equal to 4.0, a "high" index score was assigned; if the mean value was between 2.1 and 3.9, a "medium" index score was assigned; and if the mean value was less than or equal to 2.0, a "low" index score was assigned.

Note: We defined the "high" index score to include either mean or median values in order to be risk averse in identifying activities that are known to be or may be a potentially high threat. Only mean values were used in assessing "medium" and "low" index scores.

Workshop Summary

The results of the workshop scoring in each session are listed in Table 1 through 10. "High," "medium," and "low" index scores are notated as H, M, and L, respectively. As might be expected, there were positive correlations between the highest scoring effects and the ecosystem types in which those activities generally occur. For example, the high scoring effects in the alteration of freshwater systems and agriculture and silviculture sessions were generally all in the riverine ecosystem. Except for the offshore dredging and disposal session, there were fewer effects that were scored high in the marine/offshore ecosystem compared to the riverine and estuarine/nearshore ecosystems. This suggests the participants viewed the intensity of effects from nonfishing impacts to decrease as the distance from the activity increases. As one might expect, many of the far field effects that scored high were those activities that affect the water column (e.g., ocean noise, impacts to water quality) or effects that are capable of being transported by currents (oil spills or drilling mud releases). In addition, the global effects and other impacts session had high scores more evenly distributed across all ecosystems because of the nature of the impacts discussed in this session (e.g., climate change, atmospheric deposition, ocean noise). The number of activities and threats identified in the coastal development session were greater than other sessions because of the cross cutting nature of activities associated with human coastal development. Because of this, some activity types and effects assessed in the coastal development session were discussed to some degree in other sessions.

Some sessions had index scores with relatively high variability. For example, the scores for all activity types of the offshore dredging and disposal session had relatively low mean values and high standard deviations for effects in the estuarine/nearshore ecosystem. About half of the participants in this session either did not provide a score for impacts in the riverine or estuarine/nearshore ecosystems, or they marked them as "not-applicable." Participants who provided a score for these two ecosystems generally scored them relatively high. This suggests a difference in participants' interpretation of where "offshore" activities are located. Specifically, some individuals may consider the "offshore" area to be within close enough proximity of the nearshore and estuarine environments to adversely affect these areas, while others may perceive the "offshore" area to be too far removed to have a noticeable effect. There were activities in other sessions, such as beach nourishment in coastal development, with scores with high standard deviations. The high variability in perceived threats may be a reflection of regional perspectives. While the majority of the participants involved in this workshop were from the New England region, about one-quarter of the participants were from the mid-Atlantic or southeast regions where beach nourishment projects are much more common. The associated impacts to benthic habitats from beach nourishment are also generally thought to be greater in the New England (where cobble or hard bottom habitats may be present) and south Atlantic (where live bottom habitats may be present) regions than in the mid-Atlantic. However, because the responses of the workshop participants were anonymous, it was not possible to test this hypothesis.

Many of the effects that were scored as high in the workshop sessions were those that are well documented in the literature as having adverse effects on coastal resources. For example, nutrient enrichment and siltation/sedimentation effects were scored as high in nearly all workshop sessions, demonstrating the widely accepted views that these impacts translate to general reductions in the quality and quantity of fishery resources and habitats. Some of the more unexpected results of the workshop session scores are those effects that had high mean and/or median values but may be a topic that does not have a wealth of research documenting those impacts. Some of these results may be based upon a collective judgment by the participants that these activities or effects require additional scientific investigations to resolve the perceived risks and concerns. In several of these effects or activities, the authors of the associated report chapters were unable to locate information in the scientific literature regarding those threats. For example, release of pharmaceuticals and endocrine disruptors were two effects that were scored high in the workshop session, and yet the potential scope and intensity of adverse effects that these chemicals have on fishery resources has not been thoroughly investigated.

Those activities and effects considered by the workshop participants to have “high” threats to fishery habitat warrant further investigations, including research in characterizing and quantifying these impacts on fishery resources, as well as investigating methods for avoiding and/or minimizing the impacts. Refer to the Conclusions and Recommendations chapter for further discussions regarding the workshop results.

Table 1. Habitat impact categories in coastal development workshop session (N=14)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Nonpoint Source Pollution and Urban Runoff	Nutrient loading/eutrophication	H	H	M	H	H	M
	Loss/alteration of aquatic vegetation	H	H	L	H	H	L
	Release of petroleum products	M	M	M	M	M	M
	Alteration of water alkalinity	M	M	L	M	M	L
	Release of metals	H	H	M	M	H	M
	Release of radioactive wastes	M	M	L	M	M	L
	Release of pesticides	H	H	M	H	H	M
	Release of pharmaceuticals	H	M	L	H	H	L
	Alteration of temperature regimes	H	M	L	H	M	L
	Sedimentation/turbidity	H	H	L	H	H	L
	Altered hydrological regimes	M	M	L	M	M	L
	Introduction of pathogens	M	M	L	M	M	L
Road Construction and Operation	Release of sediments in aquatic habitat	H	M	L	M	M	L
	Increased sedimentation/turbidity	H	H	L	H	H	L
	Impaired fish passage	H	M	L	H	H	L
	Altered hydrological regimes	H	H	L	H	H	L
	Altered temperature regimes	H	M	L	H	M	L
	Altered stream morphology	H	M	L	H	M	L
	Altered stream bed characteristics	H	M	L	H	M	L
	Reduced dissolved oxygen	H	H	L	H	H	L
	Introduction of exotic invasive species	M	M	L	M	M	L
	Loss/alteration of aquatic vegetation	H	H	L	H	H	L
	Altered tidal regimes	H	H	L	H	M	L
	Contaminant releases	M	M	L	M	M	L
	Fragmentation of habitat	H	M	L	H	H	L
	Altered salinity regimes	M	M	L	M	M	L
Flood Control/ Shoreline Protection	Altered hydrological regimes	H	H	L	H	M	L
	Altered temperature regimes	M	M	L	M	M	L
	Altered stream morphology	H	M	L	H	M	L
	Altered sediment transport	H	H	L	H	H	L
	Alteration/loss of benthic habitat	H	H	L	M	M	L
	Reduction of dissolved oxygen	M	M	L	M	M	L
	Impaired fish passage	H	M	L	H	M	L
	Alteration of natural communities	H	M	L	M	M	L
	Impacts to riparian habitat	H	M	L	H	M	L
	Loss of intertidal habitat	H	H	L	M	H	L
	Reduced ability to counter sea level rise	H	H	L	M	H	L
	Increased erosion/accretion	H	H	L	H	H	L
Beach Nourishment	Altered hydrological regimes	M	M	L	M	M	L
	Altered temperature regimes	L	L	L	L	L	L
	Altered sediment transport	M	M	L	M	M	L
	Alteration/loss of benthic habitat	M	M	L	L	M	L
	Alteration of natural communities	M	M	M	L	M	L
	Increased sedimentation/turbidity	M	M	L	M	M	L

Table 1 (continued). Habitat impact categories in coastal development workshop session (N=14)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshor	Riverine	Estuarine/ Nearshore	Marine/ Offshor
Wetland Dredging and Filling	Alteration/loss of habitat	H	H	L	H	H	L
	Loss of submerged aquatic vegetation	H	H	L	M	H	L
	Altered hydrological regimes	H	H	L	H	H	L
	Reduction of dissolved oxygen	M	M	L	M	M	L
	Release of nutrients/eutrophication	M	M	L	M	M	L
	Release of contaminants	M	M	L	M	M	L
	Altered tidal prism	M	M	L	M	M	L
	Altered current patterns	M	M	L	M	M	L
	Altered temperature regimes	M	M	L	M	M	L
	Loss of wetlands	H	H	L	H	H	L
	Loss of fishery productivity	H	H	L	H	H	L
	Introduction of invasive species	M	M	L	M	M	L
	Loss of flood storage capacity	H	H	L	H	H	L
	Increased sedimentation/turbidity	M	M	L	M	M	L
Overwater Structures	Shading impacts to vegetation	M	M	L	M	M	L
	Altered hydrological regimes	M	M	L	M	M	L
	Contaminant releases	M	M	L	M	M	L
	Benthic habitat impacts	M	M	L	M	M	L
	Increased erosion/accretion	M	M	L	M	M	L
	Eutrophication from bird roosting	M	M	L	M	M	L
	Shellfish closures because of bird roosting	H	M	L	M	M	L
	Changes in predator/prey interactions	H	H	L	H	H	L
Pile Driving and Removal	Energy impacts	M	M	L	M	M	L
	Benthic habitat impacts	M	M	L	M	M	L
	Increased sedimentation/turbidity	M	M	L	M	M	L
	Contaminant releases	M	M	L	M	M	L
	Shading impacts to vegetation	M	M	L	M	M	L
	Changes in hydrological regimes	M	M	L	M	M	L
	Changes in species composition	M	M	L	M	M	L
Marine Debris	Entanglement	M	M	L	M	M	L
	Ingestion	L	M	L	M	M	M
	Contaminant releases	L	M	L	L	M	M
	Introduction of invasive species	M	M	L	M	M	M
	Introduction of pathogens	L	M	L	L	M	M
	Conversion of habitat	L	M	L	L	M	L

Table 2. Habitat impact categories in energy-related activities workshop session (N=13)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Petroleum Exploration, Production, and Transportation	Underwater noise	M	M	M	M	M	M
	Habitat conversion	H	H	H	H	H	M
	Loss of benthic habitat	M	H	M	M	M	M
	Contaminant discharge	M	H	M	M	H	M
	Discharge of debris	M	M	M	M	M	L
	Oil spills	H	H	H	H	H	H
	Siltation/sedimentation/turbidity	M	M	M	M	M	M
	Resuspension of contaminants	M	H	M	M	M	L
	Impacts from clean-up activities	H	H	M	M	H	M
Liquified Natural Gas	Habitat conversion	H	H	M	M	M	M
	Loss of benthic habitat	H	H	M	M	M	L
	Discharge of contaminants	H	H	H	H	H	H
	Discharge of debris	M	M	M	M	M	L
	Siltation/sedimentation/turbidity	M	H	M	M	M	M
	Resuspension of contaminants	M	H	M	M	H	L
	Entrainment/impingement	M	M	M	M	H	M
	Alteration of temperature regimes	M	M	L	M	M	L
	Alteration of hydrological regimes	M	M	L	M	M	L
	Underwater noise	M	M	M	H	H	M
	Release of contaminants	H	H	M	H	H	M
	Exclusion zone impacts	M	M	L	M	M	L
	Physical barriers to habitat	M	M	M	M	M	L
	Introduction of invasive species	H	H	M	H	M	M
	Vessel impacts	H	H	L	M	M	L
	Benthic impacts from pipelines	H	H	M	M	M	M
Offshore Wind Energy Facilities	Loss of benthic habitat	M	H	H	L	M	M
	Habitat conversion	M	H	H	L	M	M
	Siltation/sedimentation/turbidity	L	M	M	L	M	M
	Resuspension of contaminants	L	M	L	L	M	L
	Alteration of hydrological regimes	L	M	M	L	M	M
	Altered current patterns	L	M	M	L	M	M
	Alteration of electromagnetic fields	L	L	L	L	L	L
	Underwater noise	L	L	M	L	M	H
	Alteration of community structure	M	H	M	L	H	M
	Erosion around structure	L	M	M	L	L	L
	Spills associated w/ service structure	M	H	M	L	M	M

Table 2 (continued). Habitat impact categories in energy-related activities workshop session (N=13)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Wave/Tidal Energy Facilities	Habitat conversion	H	H	M	M	M	M
	Loss of benthic habitat	H	H	M	M	M	L
	Siltation/sedimentation/turbidity	M	H	M	M	M	L
	Resuspension of contaminants	M	M	L	M	M	L
	Alteration of hydrological regimes	M	M	M	M	H	L
	Altered current patterns	M	M	M	M	H	M
	Entrainment/impingement	M	M	L	H	H	M
	Impacts to migration	M	M	L	H	M	L
	Electromagnetic fields	L	L	L	L	L	L
Cables and Pipelines	Loss of benthic habitat	H	H	M	L	M	L
	Habitat conversion	H	H	M	M	M	M
	Siltation/sedimentation/turbidity	M	H	M	M	M	M
	Resuspension of contaminants	H	H	M	M	M	M
	Altered current patterns	M	M	M	L	M	L
	Alteration of electromagnetic fields	L	L	L	L	L	L
	Underwater noise	L	L	L	L	M	M
	Alteration of community structure	M	M	M	M	M	M
	Erosion around structure	L	M	M	L	M	M
	Biocides from hydrostatic testing	M	M	M	M	M	M
	Spills associated w/ service structure	H	H	M	M	M	M
	Physical barriers to habitat	H	H	H	L	L	L
	Impacts to submerged aquatic vegetation	M	H	M	M	M	L
	Water withdrawal	M	M	L	H	H	L
	Impacts from construction activities	M	H	H	M	M	M
	Impact from maintenance activities	M	M	M	L	M	M
	Thermal impacts associated with cables	L	L	L	L	L	L
	Impacts associated with armoring of pipe	M	M	M	L	L	L
	Impacts to migration	H	H	H	L	L	L

Table 3. Habitat impact categories in alteration of freshwater systems workshop session (N=13)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Dam Construction /Operation	Impaired fish passage	H	H	L	H	H	L
	Altered hydrological regimes	H	H	L	H	M	L
	Altered temperature regimes	H	H	L	H	M	L
	Altered sediment/ large woody debris transport	H	M	L	H	M	L
	Altered stream morphology	H	M	L	H	M	L
	Altered stream bed characteristics	H	M	L	H	M	L
	Reduced dissolved oxygen	H	M	L	H	M	L
	Alteration of extent of tide	H	H	L	H	H	L
	Alteration of wetlands	H	H	L	H	H	L
	Change in species communities	H	M	L	H	M	L
	Bank erosion because of drawdown	M	L	L	M	L	L
	Riparian zone development	H	M	L	H	M	L
	Acute temperature shock	H	M	L	H	M	L
Dam Removal	Release of contaminated sediments	H	H	L	H	M	L
	Alteration of wetlands	H	M	L	H	M	L
Stream Crossings	Impacts to fish passage	H	M	L	H	M	L
	Alteration of hydrological regimes	H	M	L	H	M	L
	Bank erosion	H	L	L	M	L	L
	Habitat conversion	H	M	L	H	M	L
Water Withdrawal/ Diversion	Entrainment and impingement	M	M	L	H	M	L
	Impaired fish passage	H	H	L	H	H	L
	Altered hydrological regimes	H	M	L	H	M	L
	Reduced dissolved oxygen	H	M	L	H	M	L
	Altered temperature regimes	H	H	L	H	M	L
	Release of nutrients/eutrophication	H	M	L	H	M	L
	Release of contaminants	H	M	L	H	M	L
	Altered stream morphology	H	L	L	H	M	L
	Altered stream bed characteristics	H	M	L	H	M	L
	Siltation/sedimentation/turbidity	H	M	L	H	M	L
	Change in species communities	H	M	L	H	H	L
	Alteration in groundwater levels	H	L	L	H	L	L
	Loss of forested/palustrine wetlands	H	L	L	H	L	L
	Impacts to water quality	H	M	L	H	M	L
	Loss of flood storage	M	L	L	M	L	L

Table 3 (continued). Habitat impact categories in alteration of freshwater systems workshop session (N=13)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Dredging and Filling, Mining	Reduced flood water retention	H	M	L	H	M	L
	Reduced nutrient uptake and release	M	M	L	M	M	L
	Reduced detrital food source	H	M	L	M	M	L
	Altered hydrological regimes	H	M	L	H	M	L
	Increased storm water runoff	H	M	L	H	M	L
	Loss of riparian and riverine habitat	H	M	L	H	M	L
	Altered stream morphology	H	M	L	H	L	L
	Altered stream bed characteristics	H	M	L	H	M	L
	Siltation/sedimentation/turbidity	H	M	L	H	M	L
	Reduced dissolved oxygen	H	M	L	H	M	L
	Altered temperature regimes	H	M	L	H	M	L
	Release of nutrients/eutrophication	H	M	L	H	H	L
	Release of contaminants	H	M	L	H	M	L
	Loss of submerged aquatic vegetation	H	H	L	H	H	L
	Change in species communities	H	H	L	H	M	L

Table 4. Habitat impact categories in marine transportation workshop session (N=18)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Construction and Expansion of Ports and Marinas	Loss of benthic habitat	H	H	H	M	M	M
	Siltation/sedimentation/turbidity	H	H	M	M	M	M
	Contaminant releases	H	H	M	M	H	M
	Altered hydrological regimes	H	H	L	H	H	L
	Altered tidal prism	M	H	L	M	H	L
	Altered current patterns	M	M	L	M	M	L
	Altered temperature regimes	H	M	L	H	M	L
	Loss of wetlands	H	H	L	H	H	L
	Underwater blasting/noise	M	M	L	M	M	M
	Loss of submerged aquatic vegetation	H	H	M	H	H	M
	Conversion of substrate/habitat	H	H	M	M	M	M
	Loss of intertidal flats	H	H	L	L	M	L
	Loss of water column	M	M	L	H	H	L
	Altered light regime	M	M	L	M	M	L
	Derelict structures	M	M	L	M	M	L
Operations and Maintenance of Ports and Marinas	Contaminant releases	H	H	M	M	M	M
	Storm water runoff	H	H	M	M	M	L
	Underwater noise	M	M	L	M	M	L
	Alteration of light regimes	M	M	L	M	M	L
	Derelict structures	M	M	L	L	L	L
	Mooring impacts	M	M	L	L	L	L
	Release of debris	M	M	L	M	L	L
Operation and Maintenance of Vessels	Impacts to benthic habitat	H	H	L	M	M	L
	Resuspension of bottom sediments	M	M	L	M	M	L
	Erosion of shorelines	M	M	L	M	M	L
	Contaminant spills and discharges	M	H	M	M	H	M
	Underwater noise	M	M	M	M	M	M
	Derelict structures	M	M	L	L	L	L
	Increased air emissions	L	L	L	L	L	L
	Release of debris	M	M	L	L	L	L
Navigation Dredging	Conversion of substrate/habitat	H	H	M	M	M	L
	Loss of submerged aquatic vegetation	H	H	M	H	H	L
	Siltation/sedimentation/turbidity	H	H	M	H	M	L
	Contaminant releases	H	H	M	M	M	M
	Release of nutrients/eutrophication	M	M	M	M	M	L
	Entrainment and impingement	M	M	M	M	M	L
	Underwater blasting/noise	M	M	L	M	M	L
	Altered hydrological regimes	H	H	L	H	M	L
	Altered tidal prism	M	M	L	M	M	L
	Altered current patterns	M	M	L	M	M	L
	Altered temperature regimes	H	H	L	M	M	L
	Loss of intertidal flats	H	H	L	H	H	L
	Loss of wetlands	H	H	L	H	H	L
	Contaminant source exposure	M	M	M	M	M	L

Table 5. Habitat impact categories in offshore dredging and disposal workshop session (N=22)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Offshore Mineral Mining	Loss of benthic habitat types	L	L	H	L	L	M
	Conversion of substrate/habitat	L	L	H	L	L	L
	Siltation/sedimentation/turbidity	L	L	M	L	L	M
	Changes in bottom topography	L	L	M	L	L	L
	Changes in sediment composition	L	L	H	L	L	L
	Sediment transport from site (erosion)	L	L	M	L	L	L
	Impacts to water quality	L	L	M	L	L	M
	Release of contaminants	L	L	M	L	L	M
	Change in community structure	L	L	H	L	L	M
	Changes in water flow	L	L	M	L	L	M
	Noise impacts	L	L	L	L	L	M
Petroleum Extraction	Contaminant releases	L	L	H	L	L	H
	Drilling mud impacts	L	L	H	L	L	H
	Siltation/sedimentation/turbidity	L	L	M	L	L	M
	Release of debris	L	L	M	L	L	L
	Noise impacts	L	L	M	L	L	M
	Changes in light regimes	L	L	M	L	L	M
	Habitat conversion	L	L	M	L	L	M
	Pipeline installation	L	L	M	L	L	L
Offshore Dredge Material Disposal	Burial/disturbance of benthic habitat	L	M	H	L	L	M
	Conversion of substrate/habitat	L	L	H	L	L	M
	Siltation/sedimentation/turbidity	L	L	M	L	L	M
	Release of contaminants	L	L	M	L	L	M
	Release of nutrients/eutrophication	L	L	M	L	L	M
	Altered hydrological regimes	L	L	M	L	L	M
	Altered current patterns	L	L	M	L	L	M
	Changes in bottom topography	L	L	M	L	L	L
	Changes in sediment composition	L	L	H	L	L	L
	Changes in water bathymetry	L	L	M	L	L	L
Fish Waste Disposal	Introduction of pathogens	L	L	H	L	L	H
	Release of nutrients/eutrophication	L	L	H	L	L	H
	Release of biosolids	L	L	H	L	L	M
	Loss of benthic habitat types	L	L	H	L	L	L
	Behavioral affects	L	L	M	L	L	M
Vessel Disposal	Release of contaminants	L	L	M	L	L	M
	Conversion of substrate/habitat	L	L	H	L	L	M
	Changes in bathymetry	L	L	M	L	L	L
	Changes in hydrodynamics	L	L	M	L	L	M
	Changes in community structure	L	L	H	L	L	M
	Impacts during deployment	L	L	M	L	L	M
	Release of debris	L	L	M	L	L	L

Table 6. Habitat impact categories in chemical effects: water discharge facilities workshop session (N=19)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Sewage Discharge Facilities	Release of nutrients/eutrophication	H	H	H	H	H	H
	Release of contaminants	H	H	H	H	H	H
	Impacts to submerged aquatic vegetation	H	H	M	H	H	M
	Reduced dissolved oxygen	H	H	M	H	H	M
	Siltation/sedimentation/turbidity	H	H	M	H	H	M
	Impacts to benthic habitat	H	H	M	M	M	M
	Changes in species composition	H	H	M	H	H	M
	Trophic level alterations	H	H	M	H	H	M
	Introduction of pathogens	H	H	M	M	H	M
	Introduction of harmful algal blooms	H	H	H	H	H	M
	Bioaccumulation/biomagnification	H	H	H	H	H	M
	Behavioral avoidance	M	H	M	M	H	M
	Release of pharmaceuticals	M	M	M	M	M	M
Industrial Discharge Facilities	Alteration of water alkalinity	H	M	M	M	M	L
	Release of metals	H	H	M	M	M	M
	Release of chlorine compounds	H	H	M	H	H	M
	Release of pesticides	H	H	M	H	H	M
	Release of organic compounds	H	H	H	M	H	M
	Release of petroleum products	H	H	M	M	H	M
	Release of inorganic compounds	H	H	M	H	H	M
	Release of organic wastes	M	M	M	M	M	M
	Introduction of pathogens	M	M	M	M	M	M
Combined Sewer Overflows	Potential for all of the above effects	H	H	H	H	H	H

Table 7. Habitat impact categories in physical effects: water intake and discharge facilities workshop session

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Discharge Facilities	Scouring of substrate	M	M	L	L	L	L
	Turbidity/sedimentation	H	H	M	M	M	L
	Alteration of sediment composition	H	H	M	L	L	L
	Reduced dissolved oxygen	H	H	M	H	H	L
	Alteration of salinity regimes	H	H	L	H	H	M
	Alteration of temperature regimes	H	H	M	H	H	M
	Conversion/loss of habitat	M	M	M	M	M	M
	Habitat exclusion/avoidance	H	H	L	H	H	L
	Restrictions to migration	H	H	L	H	H	L
	Acute toxicity	M	H	M	H	H	M
	Behavioral changes	M	M	L	M	M	L
	Cold shock	M	M	M	H	M	L
	Stunting of growth in fishes	M	M	L	M	M	L
	Attraction to flow	H	H	M	H	H	M
	Alteration of community structure	H	H	M	H	H	M
	Changes in local current patterns	M	M	L	M	M	L
	Physical/chemical synergies	M	H	M	M	M	M
	Increased need for dredging	H	H	L	H	H	L
	Ballast water discharge	H	H	M	M	M	M
	Gas-bubble disease/mortality	M	M	L	M	H	L
	Release of radioactive wastes	H	H	M	H	H	M
Intake Facilities	Entrainment/impingement	H	H	H	H	H	H
	Alteration of hydrological regimes	H	H	M	H	H	L
	Flow restrictions	H	H	L	H	H	L
	Construction related impacts	H	M	M	M	M	M
	Conversion/loss of habitat	H	H	M	H	H	M
	Seasonal loss of habitat	M	M	L	M	M	M
	Backwash (cleaning of system)	M	M	L	M	M	L
	Alteration of community structure	H	H	L	H	H	L
	Increased need for dredging	H	H	M	H	H	L
	Ballast water intake	H	H	M	H	H	M

Table 8. Habitat impact categories in agriculture and silviculture workshop session (N=11)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Cropland, Rangelands, Livestock, and Nursery Operations	Release of nutrients/eutrophication	H	H	L	H	H	L
	Bank/soil erosion	H	H	L	M	M	L
	Altered temperature regimes	M	M	L	M	M	L
	Siltation/sedimentation/turbidity	H	H	L	H	H	L
	Altered hydrological regimes	M	M	L	M	M	L
	Entrainment and impingement	M	L	L	H	L	L
	Impaired fish passage	M	L	L	H	M	L
	Reduced soil infiltration	M	L	L	M	L	L
	Release of pesticides	H	H	L	H	M	L
	Reduced dissolved oxygen	H	M	L	H	M	L
	Soil compaction	M	M	L	M	L	L
	Loss/alteration of wetlands	H	H	L	M	M	L
	Land-use change (post agriculture)	H	M	L	H	M	L
	Introduction of invasive species	M	M	L	M	L	L
	Introduction of pathogens	H	M	L	M	M	L
	Endocrine disruptors	H	H	L	H	H	L
	Change of community structure	M	M	L	M	M	L
	Change in species composition	H	M	L	M	M	L
Silviculture and Timber Harvest Activities	Reduced soil infiltration	M	M	L	M	L	L
	Siltation/sedimentation/turbidity	H	M	L	H	M	L
	Altered hydrological regimes	M	M	L	M	M	L
	Impaired fish passage	M	L	L	H	M	L
	Bank/soil erosion	H	M	L	H	M	L
	Altered temperature regimes	H	M	L	H	M	L
	Release of pesticides	H	H	L	H	H	L
	Release of nutrients/eutrophication	H	H	L	H	H	L
	Reduced dissolved oxygen	H	M	L	H	M	L
	Loss/alteration of wetlands	H	M	L	H	M	L
	Soil compaction	M	L	L	M	L	L
Timber and Paper Mill Processing Activities	Chemical contaminant releases	H	H	L	H	H	L
	Entrainment and impingement	M	L	L	H	M	L
	Thermal discharge	H	L	L	M	L	L
	Reduced dissolved oxygen	H	M	L	H	M	L
	Conversion of benthic substrate	H	M	L	M	L	L
	Loss/alteration of wetlands	M	M	L	M	M	L
	Alteration of light regimes	M	L	L	M	L	L

Table 9. Habitat impact categories in introduced/nuisance species and aquaculture workshop session (N=14)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Introduced/ Nuisance Species	Habitat alterations	H	H	M	M	M	M
	Trophic alterations	M	H	M	M	M	M
	Gene pool alterations	H	H	M	H	H	M
	Alterations of communities	H	H	M	M	H	M
	Introduced diseases	M	H	M	M	H	M
	Changes in species diversity	H	H	H	H	H	M
	Alteration in health of native species	M	M	M	M	M	M
	Impacts to water quality	M	M	M	M	M	M
Aquaculture	Discharge of organic waste	M	H	M	M	M	M
	Seafloor impacts	M	H	M	M	M	M
	Introduction of exotic invasive species	H	H	M	M	H	M
	Food web impacts	H	H	M	H	H	M
	Gene pool alterations	H	H	M	H	M	M
	Impacts to water column	M	M	M	M	H	M
	Impacts to water quality	M	H	L	M	H	M
	Changes in species diversity	M	H	M	M	H	M
	Sediment deposition	H	H	M	L	L	L
	Introduction of diseases	M	H	M	M	M	M
	Habitat replacement/exclusion	H	H	M	M	M	L
	Habitat conversion	H	H	M	M	H	M

Table 10. Habitat impact categories in global effects and other impacts workshop session (N=17)

Activity Type	Potential Effects	Habitat Impact Categories					
		Life History/Ecosystem Type					
		Benthic/Demersal Stages			Pelagic Stages		
		Riverine	Estuarine/ Nearshore	Marine/ Offshore	Riverine	Estuarine/ Nearshore	Marine/ Offshore
Climate Change	Alteration of hydrological regimes	H	H	M	H	H	H
	Alteration of temperature regimes	H	H	H	H	H	H
	Changes in dissolved oxygen	H	H	M	H	H	M
	Nutrient loading/eutrophication	M	H	M	M	M	M
	Release of contaminants	H	H	M	M	M	M
	Bank/soil erosion	H	M	L	M	M	L
	Alteration in salinity	M	H	M	M	H	M
	Alteration of weather patterns	H	H	M	H	H	H
	Alteration of alkalinity	M	M	M	M	M	M
	Changes in community structure	H	H	H	H	H	H
	Changes in ocean/coastal use	M	M	M	M	M	M
	Changes in ecosystem structure	M	H	L	M	H	L
	Loss of wetlands	H	H	L	H	H	L
Ocean Noise	Mechanical injury to organisms	M	M	H	M	M	H
	Impacts to feeding behavior	M	M	M	M	M	M
	Impacts to spawning behavior	M	M	M	M	M	M
	Impacts to migration	M	M	M	M	M	M
	Exclusion of organisms to habitat	M	M	M	M	M	M
	Changes in community structure	M	M	M	M	M	M
Atmospheric Deposition	Nutrient loading/eutrophication	H	H	M	H	H	M
	Mercury loading/bioaccumulation	H	H	M	H	H	H
	Polychlorinated biphenyls and other contaminants	H	H	M	H	H	M
	Alteration of ocean alkalinity	M	M	M	M	M	M
	Alteration of climatic cycle	M	M	M	M	M	M
Military/ Security Activities	Exclusion of organisms to habitat	L	L	M	L	M	M
	Noise impacts	M	M	M	M	M	H
	Chemical releases	M	H	M	M	M	M
	Impacts to tidal/intertidal habitats	M	M	L	L	M	L
	Blasting injuries from ordinances	M	M	M	M	M	M
Natural Disasters and Events	Loss/alteration of habitat	H	H	M	H	H	M
	Impacts to habitat from debris	M	M	M	M	M	L
	Impacts to water quality	M	H	M	H	H	M
	Impacts from emergency response	M	M	L	M	M	L
	Alteration of hydrological regimes	M	M	M	M	M	L
	Changes in community composition	M	H	M	M	M	M
	Underwater landslides	L	L	M	L	L	M
Electromagnetic Fields	Changes to migration of organisms	M	M	M	M	M	M
	Behavioral changes	M	M	M	M	M	M
	Changes in predator/prey relationships	L	M	M	M	M	M

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.

References for Coastal Development

Able KW, Manderson JP, Studholme AL. 1998. The distribution of shallow water juvenile fishes in an urban estuary: The effects of manmade structures in the lower Hudson River. Estuaries 21(4B):731-44.

Amaral M, Lee V, Rhodes J. 2005. Environmental guide for marinas: controlling nonpoint source and storm water pollution in Rhode Island: best management practices for marinas. [Internet]. Narragansett (RI): University of Rhode Island, Coastal Resources Center, Rhode Island Sea Grant. [cited 2007 Dec 28]. Available from: http://seagrant.gso.uri.edu/BMP/BMP.html.

Arkoosh MR, Clemons E, Huffman P, Kagley AN, Casillas E, Adams N, Sanborn HR, Collier TK, Stein JE. 2001. Increased susceptibility of juvenile chinook salmon to vibriosis after exposure to chlorinated and aromatic compounds found in contaminated urban estuaries. Journal of Aquatic Animal Health 13(3):257-68.

[ASMFC] Atlantic States Marine Fisheries Commission. 1992. Fishery management plan for inshore stocks of winter flounder. Washington (DC): ASMFC. FMR No. 21. 138 p.

[ASMFC] Atlantic States Marine Fisheries Commission. 2004. NRC report outlines the impacts of oil on marine fish habitat. Habitat Hotline Atlantic 11(3):1-5.

Bahgat FJ, King PE, Shackley SE. 1989. Ultrastructural changes in the muscle tissue of *Clupea harengus* L. larvae induced by acid pH. Journal of Fish Biology 34:25-30.

Balcom B, Howell P. 2006. Responding to a resource disaster: American lobsters in Long Island Sound 1999-2004. Groton (CT): University of Connecticut, Connecticut Sea Grant. CTSG-06-02. 22 p.

Barr BW. 1993. Environmental impacts of small boat navigation: vessel/sediment interactions and management implications. In: Magoon OT, editor. Coastal Zone '93: proceedings of the eighth symposium on coastal and ocean management; 1993 Jul 19-23; New Orleans, LA. American Shore and Beach Preservation Association. p 1756-70.

Bejda AJ, Phelan BA, Studholme AL. 1992. The effects of dissolved oxygen on growth of young-of-the-year winter flounder. Environmental Biology of Fishes 34:321-7.

Belford DA, Gould WR. 1989. An evaluation of trout passage through six highway culverts in Montana. North American Journal of Fisheries Management 9(4):437-45.

Bigford TE. 1991. Sea-level rise, nearshore fisheries, and the fishing industry. Coastal Management 19:417-37.

Bilby RE, Sullivan K, Duncan SH. 1989. The generation and fate of road-surface sediment in forested watersheds in southwestern Washington. Forest Science 35(2):453-68.

Blaxter JHS. 1969. Development: eggs and larvae. In: Hoar WS, Randall DJ, editors. Fish physiology: Vol. 3, reproduction and growth, bioluminescence, pigments, and poisons. New

York (NY): Academic Press. p 177-252.

Blaxter JHS. 1977. The effect of copper on the eggs and larvae of plaice and herring. Journal of the Marine Biological Association of the United Kingdom 57:849-59.

Bodammer JE. 1981. The cytopathological effects of copper on the olfactory organs of larval fish (*Pseudopleuronectes americanus* and *Melanogrammus aeglefinus*). Copenhagen (Denmark): ICES CM-1981/E: 46.

Bowling A, Mackin K. 2003. The state of the Ipswich River. Ipswich (MA): Ipswich River Watershed Association. 9 p.

Bozek CM, Burdick DM. 2005. Impacts of seawalls on saltmarsh plant communities in the Great Bay Estuary, New Hampshire USA. Wetlands Ecology and Management 13:553-68.

Breitburg DL. 1988. Effects of turbidity on prey consumption by striped bass larvae. Transactions of the American Fisheries Society 117(1):72-7.

Bricker SB, Clement CG, Pirhalla DE, Orlando SP, Farrow DRG. 1999. National estuarine eutrophication assessment: effects of nutrient enrichment in the Nation's estuaries. Silver Spring (MD): NOAA, NOS, Special Projects Office and the National Centers for Coastal Ocean Science. 71 p.

Brodeur JC, Sherwood G, Rasmussen JB, Hontela A. 1997. Impaired cortisol secretion in yellow perch (*Perca flavescens*) from lakes contaminated by heavy metals: *in vivo* and *in vitro* assessment. Canadian Journal of Fisheries and Aquatic Sciences 54(12):2752-8.

Buchholtz ten Brink MR, Manheim FT, Bothner MH. 1996. Contaminants in the Gulf of Maine: what's there and should we worry? In: Dow D, Braasch E, editors. The health of the Gulf of Maine ecosystem: cumulative impacts of multiple stressors. Hanover (NH): Dartmouth College, Regional Association for Research on the Gulf of Maine (RARGOM) Report 96-1. p 91-5.

Buck EH, Copeland C, Zinn JA, Vogt DU. 1997. *Pfiesteria* and related harmful blooms: natural resource and human health concerns. [Internet]. Washington (DC): National Council for Science and the Environment. Congressional Research Service Report for Congress 97-1047 ENR. [cited 2008 Jul 9]. Available from: http://www.cnie.org/NLE/CRSreports/marine/mar-23.cfm.

Burdick DM, Buchsbaum R, Holt E. 2001. Variation in soil salinity associated with expansion of *Phragmites australis* in salt marshes. Environmental and Experimental Botany 46(3):247-61.

Burdick DM, Short FT. 1999. The effects of boat docks on eelgrass beds in coastal waters of Massachusetts. Environmental Management 23(2):231-40.

Caltrans. 2001. San Francisco-Oakland Bay Bridge East Span seismic safety project, pile installation demonstration project, fisheries impact assessment. Sacramento (CA): California Department of Transportation. 59 p.

Carlton JT. 2001. Introduced species in U.S. coastal waters: environmental impacts and management priorities. Arlington (VA): Pew Oceans Commission. 28 p.

Chapman MG. 2003. Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. Marine Ecology Progress Series 264:21-9.

Chiasson AG. 1993. The effects of suspended sediment on rainbow smelt (*Osmerus mordax*): a laboratory investigation. Canadian Journal of Zoology 71:2419-24.

Clancy CG, Reichmuth DR. 1990. A detachable fishway for steep culverts. North American Journal of Fisheries Management 10(2):244-6.

Collette BB, Klein-MacPhee G, editors. 2002. Bigelow and Schroeder's fishes of the Gulf of Maine. 3rd ed. Washington (DC): Smithsonian Institution Press. 748 p.

Copello S, Quintana F. 2003. Marine debris ingestion by southern giant petrels and its potential relationship with fisheries in the Southern Atlantic Ocean. Marine Pollution Bulletin 46(11):1504-15.

Cottingham D. 1988. Persistent marine debris: challenge and response: the federal perspective. Fairbanks (AK): Alaska Sea Grant. Report No. 88-1.

Deegan LA, RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

De Guise S, Maratea J, Perkins C. 2004. Malathion immunotoxicity in the American lobster (*Homarus americanus*) upon experimental exposure. Aquatic Toxicology 66(4):419-25.

DeLorenzo ME, Scott GI, Ross PE. 2001. Toxicity of pesticides to aquatic microorganisms: a review. Environmental Toxicology and Chemistry 20(1):84-98.

De Robertis A, Ryer CH, Veloza A, Brodeur RD. 2003. Differential effects of turbidity on prey consumption of piscivorous and planktivorous fish. Canadian Journal of Fisheries and Aquatic Sciences 60:1517-1526.

Duffy-Anderson JT, Able KW. 1999. Effects of municipal piers on the growth of juvenile fishes in the Hudson River estuary: a study across a pier edge. Marine Biology 133(3):409-18.

Evans WA, Johnston B. 1980. Fish migration and fish passage: a practical guide to solving fish passage problems. Washington (DC): US Department of Agriculture, Forest Service. 163 p.

Fulton MH, Scott GI, Fortner A, Bidleman TF, Ngabe B. 1993. The effects of urbanization on small high salinity estuaries of the southeastern United States. Archives of Environmental Contamination and Toxicology 25(4):476-84.

Furniss MJ, Roelofs TD, Yee CS. 1991. Road construction and maintenance. In: Meehan WR, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19th ed. Bethesda (MD): American Fisheries Society. p 297-323.

Gilbert SF. 2000. Developmental biology. [Internet]. Sunderland (MA): Sinauer Associates. 6th ed. [updated 2007 Oct 17; cited 2006 May 17]. Available from: http://www.ncbi.nlm.nih.gov/

Glasby TM. 1999. Effects of shading on subtidal epibiotic assemblages. Journal of Experimental Marine Biology and Ecology 234:275-90.

Goldsborough WJ. 1997. Human impacts on SAV-a Chesapeake Bay case study. In: Stephan CD, Bigford TE, editors. Atlantic coastal submerged aquatic vegetation: a review of its ecological role, anthropogenic impacts, state regulation, and value to Atlantic coastal fisheries. Washington (DC): ASMFC Habitat Management Series #1. 68 p + appendices.

Gould E, Clark PE, Thurberg FP. 1994. Pollutant effects on dermersal fishes. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 30-41.

Greenberg CH, Crownover SH, Gordon DR. 1997. Roadside soils: a corridor for invasion of xeric scrub by nonindigenous plants. Natural Areas Journal 17(2):99-109.

Greene K. 2002. Beach nourishment: a review of the biological and physical impacts. Washington (DC): ASMFC. Habitat Management Series #7. 179 p.

Haas MA, Simenstad CA Jr, Cordell JR, Beauchamp DA, Miller BS. 2002. Effects of large overwater structures on epibenthic juvenile salmon prey assemblages in Puget Sound, WA. [Internet]. Seattle (WA): Washington State Transportation Center (TRAC), University of Washington, and Washington State Department of Transportation. Final Research Report WA-RD 550.1. [cited 2008 Jul 9]. 114 p. Available from: http://depts.washington.edu/trac/bulkdisk/pdf/550.1.pdf

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Hoagland KD, Carder JP, Spawn RL. 1996. Chapter 15: Effects of organic toxic substances. In: Stevenson RJ, Bothwell ML, Love RL, editors. Algal ecology: freshwater benthic ecosystems. San Diego (CA): Academic Press. p 469-96.

Hoagland P, Kite-Powell HL. 1997. Characterization and mitigation of marine debris in the Gulf of Maine. A report prepared for the U.S. Gulf of Maine Association under contract no. GM97-13. Gulf of Maine Council on the Marine Environment. 41 p.

Holland AF, Sanger DM, Gawle CP, Lerberg SB, Santiago MS, Riekerk GHM, Zimmerman LE, Scott GI. 2004. Linkages between tidal creek ecosystems and the landscape and demographic attributes of their watersheds. Journal of Experimental Marine Biology and Ecology 298(2):151-78.

[IPCC] Intergovernmental Panel on Climate Change. 2007. Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. [Internet]. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL, editors. New York (NY): Cambridge University Press. [cited 2008 Jul 9]. 996 p. Available from: http://www.ipcc.ch.

[ICES] International Council for the Exploration of the Sea. 1991. Report of the study group on ecosystem effects of fishing activities. 11-15 March 1991; Lowestoft, England. Copenhagen (Denmark): ICES C.M. 1991/G:7 Session Y.

Jackson SD. 2003. Design and construction of aquatic organism passage at road-stream crossings: ecological considerations in the design of river and stream crossings. In: Irwin CL, Garrett P, McDermott KP, editors. 2003 Proceedings of the International Conference on Ecology and Transportation; 2003 Aug 24-29; Lake Placid, NY. Raleigh (NC): Center for Transportation and the Environment, North Carolina State University. p 20-9.

Kearney VF, Segal Y, Lefor MW. 1983. The effects of docks on salt marsh vegetation. Hartford (CT): Connecticut Department of Environmental Protection, Water Resources Unit. 22 p.

Kelley JT. 1992. Sea-level change and coastal erosion in the western Gulf of Maine. In: The Gulf of Maine. Silver Spring (MD): NOAA Coastal Ocean Program Regional Synthesis Series No. 1. p 27-44.

Kennedy VS, Twilley RR, Kleypas JA, Cowan JH, Hare SR. 2002. Coastal and marine ecosystems and global climate change: potential effects on U.S. resources. Arlington (VA): Pew Center on Global Climate Change. 51 p.

Kennish MJ. 1998. Pollution impacts on marine biotic communities. Boca Raton (FL): CRC Press.

Kennish MJ. 2002. Impacts of motorized watercraft on shallow estuarine and coastal marine environments. Journal of Coastal Research Special Issue 37:1-202.

Kenworthy JW, Hauners DE. editors. 1991. The light requirements of seagrasses: proceedings of a workshop to examine the capability of water quality criteria, standards and monitoring programs to protect seagrasses. Beaufort (NC): NMFS Beaufort Laboratory. NOAA Technical Memorandum NMFS-SEFC-287. 179 p.

Klein-MacPhee G, Cardin JA, Berry WJ. 1984. Effects of silver on eggs and larvae of the winter flounder. Transactions of the American Fisheries Society 113(2):247-51.

Koski KV. 1981. The survival and quality of two stocks of chum salmon (*Oncorhynchus keta*) from egg deposition to emergence. Rapport Et Proces-Verbaux Des Reunions Du Conseil International Pour L'Exploration De La Mer (ICES) 178:330-3.

Laist DW. 1997. Impacts of marine debris: entanglement of marine life in marine debris including a comprehensive list of species with entanglement and ingestion records. In: Coe JM, Rogers

DB, editors. Marine debris: sources, impacts, and solutions. New York (NY): Springer. p 99-139.

Lang T, Dethlefsen V. 1987. Cadmium in skeletally deformed and normally developed Baltic cod (*Gadus morhua* L.). Copenhagen (Denmark): ICES CM-1987/ E:30. 18 p.

Larsen PF. 1992. An overview of the environmental quality of the Gulf of Maine. In: The Gulf of Maine. Silver Spring (MD): NOAA Coastal Ocean Program Synthesis Series No. 1. p 71-95.

Lin Q, Mendelssohn IA. 1996. A comparative investigation of the effects of south Louisiana crude oil on the vegetation of fresh, brackish, and salt marshes. Marine Pollution Bulletin 32(2):202-9.

Lohrer AM, Wetz JJ. 2003. Dredging-induced nutrient release from sediments to the water column in a southeastern saltmarsh tidal creek. Marine Pollution Bulletin 46(9):1156-63.

Longmuir C, Lively T. 2001. Bubble curtain systems for use during marine pile driving. New Westminister (BC), Canada: Fraser River Pile and Dredge LTD. 9 p.

Lonsdale WN, Lane AM. 1994. Tourist vehicles as vectors of weed seeds in Dadoed National Park, northern Australia. Biological Conservation 69(3):277-83.

MacDonald LH, Sampson RW, Anderson DM. 2001. Runoff and road erosion at the plot and road segment scales, St John, US Virgin Islands. Earth Surface Processes and Landforms 26:251-72.

MacKenzie CL Jr. 2007. Causes underlying the historical decline in eastern oyster (*Crassostrea virginica* Gmelin, 1791) landings. Journal of Shellfish Research 26(4):927-38.

Messieh SN, Rowell TW, Peer DL, Cranford PJ. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. Continental Shelf Research 11(8-10):1237-63.

Meyers TR, Hendricks JD. 1982. A summary of tissue lesions in aquatic animals induced by controlled exposures to environmental contaminants, chemotherapeutic agents, and potential carcinogens. Marine Fisheries Review 44(12):1-17.

Milliken AS, Lee V. 1990. Pollution impacts from recreational boating: a bibliography and summary review. Narragansett (RI): Rhode Island Sea Grant. National Sea Grant Depository Publication # RIU-G-002.

Mitsch WJ, Gosselink JG. 1993. Wetlands. 2nd ed. New York (NY): Van Nostrand Reinhold. 722 p.

Moore A, Waring CP. 2001. The effects of a synthetic pyrethroid pesticide on some aspects of reproduction in Atlantic salmon (*Salmo salar* L.). Aquatic Toxicology 52:1-12.

Moring J. 2005. Recent trends in anadromous fishes. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant

College Program; Publication No. MITSG 05-5. p 25-42.

[NOAA] National Oceanic and Atmospheric Administration. 1991. Status and trends in concentrations of selected contaminants in Boston Harbor sediments and biota. Seattle (WA): National Ocean Service, Office of Oceanography and Marine Assessment. NOAA Technical Memorandum NOS OMA 56.

Nedeau E. 2004. Effects of climate change on the Gulf of Maine region. [Internet]. Gulf of Maine Council on the Marine Environment. [cited 2008 Jul 9]. 12 p. Available from: http://www.gulfofmaine.org/summit/ClimateChangeKbase.pdf.

Newcombe CP, Jensen JO. 1996. Channel suspended sediment and fisheries: A synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4):693-727.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol 1.

Niering WA. 1988. The Audubon Society nature guides. Wetlands. New York (NY): Alfred A. Knopf. 638 p.

Nightingale BJ, Simenstad CA Jr. 2001. Overwater structures: marine issues. [Internet]. Olympia (WA): Washington Department of Fish and Wildlife. White Paper. [cited 2008 Jul 9]. 133 p + appendices. Available from: http://wdfw.wa.gov/hab/ahg/finalmar.pdf.

Niimi AJ. 2004. Role of container vessels in the introduction of exotic species. Marine Pollution Bulletin 49(9-10):778-82.

O'Reilly JE. 1994. Nutrient loading and eutrophication. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 25-30.

Orth RJ, Carruthers TJB, Dennison WC, Duarte CM, Fourqurean JW, Heck KL Jr, Hughes AR, Kendrick GA, Kenworthy WJ, Olyarnik S, Short FT, Waycott M, William SL. 2006. A global crisis for seagrass ecosystems. Bioscience 56(12):987-96.

Pacheco A. 1984. Final report, Seasonal occurrence of finfish and larger invertebrates at three sites in Lower New York Harbor, 1981-1982. Sandy Hook (NJ): NOAA/NMFS. Special report for USACE, New York District. 53 p.

Penttila D, Doty D. 1990. Results of 1989 eelgrass shading studies in Puget Sound. Olympia (WA): Washington Department of Fisheries. Draft Progress Report. 41 p.

Peterson CH, Hickerson DHM, Johnson GG. 2000. Short-term consequences of nourishment and bulldozing on the dominant large invertebrates of a sandy beach. Journal of Coastal Research 16(2):368-78.

Pilkey OH, Dixon KL. 1996. The Corps and the shore. Washington (DC): Island Press. 272 p.

Posey MH, Alphin TD, Meyer DL, Johnson JM. 2003. Benthic communities of common reed *Phragmites australis* and marsh cordgrass *Spartina alterniflora* marshes in Chesapeake Bay. Marine Ecology Progress Series 261:51-61.

Poston T. 2001. Treated wood issues associated with overwater structures in freshwater and marine environments. [Internet]. Olympia (WA): Washington State Departments of Transportation, Fish and Wildlife, and Ecology. White Paper. [cited 2008 Jul 9]. 85 p. Available from: http://wdfw.wa.gov/hab/ahg/finaltw.pdf.

Preston BL. 2002. Indirect effects in aquatic ecotoxicology: implications for ecological risk assessment. Environmental Management 29(3):311-23.

[PSDS] Program for the Study of Developed Shorelines. 2005. The U.S. beach nourishment experience: including New England, east coast barrier islands, Pacific coast and Gulf of Mexico shorelines. [Internet]. Cullowhee (NC): Western Carolina University, c2006. [Cited 2008 Jul 9]. Available from: http://psds.wcu.edu/1038.asp.

Readman JW, Kwong LLW, Grondin D, Bartocci J, Villeneuve JP, Mee LD. 1993. Coastal water contamination from triazine herbicide used in antifouling paints. Environmental Science and Technology 27(9):1940-2.

Rice SD, Short JW, Heintz RA, Carls MG, Moles A. 2000. Life-history consequences of oil pollution in fish natal habitat. In: Catania P, editor. Energy 2000: the beginning of a new millennium. Lancaster (PA): Technomic Publishers. p 1210-5.

Safavi HR. 1996. Quality control of urban runoff and sound management. Hydrobiologia: Diapause in the Crustacea:131-41.

Scott GI, Fulton MH, Moore DW, Wirth EF, Chandler GT, Key PB, Daugomah JW, Strozier ED, Devane J, Clark JR, and others. 1999. Assessment of risk reduction strategies for the management of agricultural nonpoint source pesticide runoff in estuarine ecosystems. Toxicology and Industrial Health 15:200-13.

Shafer DJ. 1999. The effects of dock shading on the seagrass *Halodule wrightii* in Perdido Bay, Alabama. Estuaries 22(4):936-43.

Short FT, Burdick DM. 1996. Quantifying, eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. Estuaries 19(3):730-9.

Shumway SE, Kraeuter JN, editors. 2000. Molluscan shellfish research and management: charting a course for the future. Final Proceedings from the Workshop; 2000 Jan; Charleston, SC. Washington (DC): Department of Commerce. 156 p.

Smith KN, Mezich RR. 1999. Blending management and research for protection of manatee habitat in Florida: the seagrass dock effect. In: Poster presentation at the 15th Biennial International Estuarine Research Federation Conference; 1999 Sep 25-30; New Orleans, LA. Port Republic (MD): Coastal and Estuarine Research Federation.

Spence BC, Lomnicky GA, Hughes RM, Novitzki RP. 1996. An ecosystem approach to salmonid conservation. [Internet]. Corvallis (OR): ManTech Environmental Research Services Corp. TR-4501-96-6057. [cited 2008 Jul 9]. Available from: http://www.nwr.noaa.gov/Publications/Reference-Documents/ManTech-Report.cfm.

Staurnes M, Hansen LP, Fugelli K, Haraldstad O. 1996. Short-term exposure to acid water impairs osmoregulation, seawater tolerance, and subsequent marine survival of smolts of Atlantic salmon (*Salmo salar* L.). Canadian Journal of Fisheries and Aquatic Sciences 53(8):1695-704.

Thurberg FP, Gould E. 2005. Pollutant effects upon cod, haddock, pollock, and flounder of the inshore fisheries of Massachusetts and Cape Cod Bays. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 43-66.

Tyrrell MC. 2005. Gulf of Maine marine habitat primer. [Internet]. Gulf of Maine Council on the Marine Environment. vi + 54 p. [cited 2008 Jul 2]. Available from: www.gulfofmaine.org/habitatprimer/.

[USEPA] US Environmental Protection Agency. 1993. Guidance specifying management measures for sources of nonpoint pollution in coastal waters. [Internet]. Washington (DC): US EPA Office of Water. EPA-840-B-92-002. [cited 2008 Jul 9]. Available from: http://www.epa.gov/OWOW/NPS/MMGI.

[USEPA] US Environmental Protection Agency. 1995. National water quality inventory: 1994 Report to Congress. Washington (DC): US EPA Office of Water. EPA-841-R-95-005.

[USEPA] US Environmental Protection Agency. 2001. National management measures guidance to control nonpoint source pollution from marinas and recreational boating. Washington (DC): EPA Office of Water. EPA-841-B-01-005.

[USEPA] US Environmental Protection Agency. 2002. National water quality inventory: 2000 Report to Congress. [Internet]. Washington (DC): US EPA Office of Water. EPA-841-R-02-001. [cited 2008 Jul 9]. Available from: http://www.epa.gov/305b/2000report/.

[USEPA] US Environmental Protection Agency. 2003a. Guide for industrial waste management. Washington (DC): US EPA Office of Solid Waste. EPA-530-R-03-001.

[USEPA] US Environmental Protection Agency. 2003b. National management measures for the control of non-point pollution from agriculture. [Internet]. Washington (DC): US EPA Office of Water. EPA-841-B-03-004. [cited 2008 Jul 9]. Available from: http://www.epa.gov/owow/nps/agmm/index.html.

[USEPA] US Environmental Protection Agency. 2004. National coastal condition report II.

[Internet]. Washington (DC): US EPA Office of Research and Development/ Office of Water. EPA-620/R-03/002. [cited 2008 Jul 9]. Available from: http://www.epa.gov/owow/oceans/nccr2.

[USEPA] US Environmental Protection Agency. 2005. National management measures to control nonpoint source pollution from urban areas. Washington (DC): US EPA Office of Water. EPA-841-B-05-004. 518 p.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1999. Status review of anadromous Atlantic salmon in the United States. Hadley (MA): USFWS. p 1-131.

Valiela I. 1995. Marine ecological processes, 2nd ed. New York (NY): Springer-Verlag.

Valiela I, Rutecki D, Fox S. 2004. Salt marshes: biological controls of food webs in a diminishing environment. Journal of Experimental Marine Biology and Ecology 300(1-2):131-59.

Valverde HR, Trembanis AC, Pilkey OH. 1999. Summary of beach nourishment episodes on the U.S. East Coast Barrier Islands. Journal of Coastal Research 15(4):1100-18.

Vandermeulen JH, Mossman D. 1996. Sources of variability in seasonal hepatic microsomal oxygenase activity in winter flounder (*Pleuronectes americanus*) from a coal tar contaminated estuary. Canadian Journal of Fisheries and Aquatic Sciences 53:1741-53.

Weis JS, Weis P. 2002. Contamination of saltmarsh sediments and biota by CCA treated wood walkways. Marine Pollution Bulletin 44:504-10.

Wilber DH, Clarke DG. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21(4):855-75.

Williams GD, Thom RM. 2001. Marine and estuarine shoreline modification issues. [Internet]. White Paper submitted to Washington Department of Fish and Wildlife, Washington Dept. of Ecology, Washington Dept. of Transportation. Sequim (WA): Battelle Memorial Institute. 99 p. [cited 2008 Jul 9]. Available from: http://wdfw.wa.gov/hab/ahg/marnrsrc.htm.

Ziegler AD, Sutherland RA, Gaimbelluca TW. 2001. Interstorm surface preparation and sediment detachment by vehicle traffic on unpaved mountain roads. Earth Surface Processes and Landforms 26(3):235-50.

CHAPTER THREE: ENERGY-RELATED ACTIVITIES

Petroleum Exploration, Production, and Transportation

Introduction

The exploration, production, and transportation of petroleum have the potential to impact riverine, estuarine, and marine environments on the northeastern US coast. Petroleum exploration, production, and transportation are a particular concern in areas such as the Gulf of Maine and Georges Bank, which support important fishery resources and represent significant value to the US economy. Although petroleum exploration and production do not currently occur within the northeast coastal and offshore region, the transportation of oil and gas (i.e., pipelines and tankers) and the associated infrastructure are widespread. It is expected that issues relating to petroleum development will continue to gain importance as world energy costs and demands rise. The Energy Policy Act of 2005 (Pub. L. 109-58, § 357, 42 U.S.C. §15912) authorizes the Minerals Management Service (MMS) to perform surveys (exploration) for petroleum reserves on the Outer Continental Shelf (OCS) of the United States. The OCS is the submerged lands, subsoil, and seabed lying between the United States' seaward jurisdiction and the seaward extent of federal jurisdiction.

Petroleum exploration involves seismic testing, drilling sediment cores, and test wells in order to locate potential oil and gas deposits. Petroleum production includes the drilling and extraction of oil and gas from known reserves. Oil and gas rigs are placed on the seabed and as oil is extracted from the reservoirs, it is transported directly into pipelines. While rare, in cases where the distance to shore is too great for transport via pipelines, oil is transferred to underwater storage tanks. From these storage tanks, oil is transported to shore via tanker (CEQ 1977). According to the MMS, there are 21,000 miles of pipeline on the United States OCS. According to the National Research Council (NRC), pipeline spills account for approximately 1,900 tonnes per year of petroleum into US OCS waters, primarily in the central and western Gulf of Mexico (NRC 2003).

The major sources of oil releases as a result of petroleum extraction include accidental spills and daily operational discharges. The NRC estimates the largest anthropogenic source of petroleum hydrocarbon releases into the marine environment is from petroleum extraction-related activities. Approximately 2,700 tonnes per year in North America and 36,000 tonnes per year worldwide are introduced to the marine environment as a result of "produced waters" (NRC 2003). "Produced waters" are waters that are pumped to the surface from oil reservoirs which cannot be separated from the oil. Produced waters are either injected back into reservoirs or discharged into the marine environment (NRC 2003). Over 90% of the oil released from extraction activities is from produced water discharges which contain dissolved compounds (i.e., polycyclic aromatic hydrocarbons, PAH) and dispersed crude oil (NRC 2003*)*. These compounds stay suspended in the water column and undergo microbial degradation or are sorbed onto suspended sediments and are deposited on the seabed. Elevated levels of PAH in sediments are typically found up to 300 m from the discharge point (NRC 2003).

While petroleum extraction and transportation can result in impacts to the marine environment, it is important to note that natural seeps contribute to approximately 60% of all petroleum hydrocarbons that are released into the marine environment **(**NRC 2003**)**. In addition, land-based runoff and discharges by two–stroke recreational boating engines account for nearly 22% of the total petroleum released into the marine environment in North America (NRC 2003).

Underwater noise

Oil and gas activities generate noise from drilling activities, construction, production facility operations, seismic exploration, and supply vessel and barge operations that can disrupt or damage living marine resources. The effects of oil exploration-related seismic energy may cause fish to disperse from the acoustic pulse with possible disruption to their feeding patterns (Marten et al. 2001). Larvae and young fish are particularly sensitive to noise generated from underwater seismic equipment. Noise in the marine environment may adversely affect marine mammals by causing them to change behavior (e.g., movement and feeding), interfering with echolocation and communication, or injuring hearing organs (Richardson et al. 1995). Noise issues related to petroleum tanker traffic can adversely affect fishery resources within the marine environment, particularly within estuarine areas which host much of the nation's petroleum land-based port activities. Refer to the chapters on Marine Transportation and Global Effects and Other Impacts for information regarding impacts to fishery resources from underwater noise.

Habitat conversion and loss

Petroleum extraction and transportation can lead to a conversion and loss of habitat in a number of ways. Activities such as vessel anchoring, platform or artificial island construction, pipeline laying, dredging, and pipeline burial can alter bottom habitat by altering substrates used for feeding or shelter. Disturbances to the associated epifaunal communities, which may provide feeding or shelter habitat, can also result. The installation of pipelines associated with petroleum transportation can have direct and indirect impacts on offshore, nearshore, estuarine, wetland, beach, and rocky shore coastal zone habitats. The destruction of benthic organisms and habitat can occur through the installation of pipelines on the sea floor (Gowen 1978). Benthic organisms, especially prey species, may recolonize disturbed areas, but this may not occur if the composition of the substrate is drastically changed or if facilities are left in place after production ends.

The discharge of drilling cuttings (i.e., crushed sedimentary rock) during petroleum extraction operations can result in varying degrees of change to the sea floor and affect feeding, nursery, and shelter habitat for various life stages of marine organisms. Cuttings may adversely affect bottom-dwelling organisms at the site by burial of immobile forms or forcing mobile forms to migrate. The accumulation of drill cuttings on the ocean floor can alter the benthic sedimentary environment (NRC 2003).

Physical damage to coastal wetlands and other fragile areas can be caused by onshore infrastructure and pipelines associated with petroleum production and transportation. Physical alterations to habitat can occur from the construction, presence, and eventual decommissioning and removal of facilities such as islands or platforms, storage and production facilities, and pipelines to onshore common carrier pipelines, storage facilities, or refineries. For additional information regarding impacts of pipelines associated with petroleum production, refer to the section on Cables and Pipelines in this chapter of the report.

Contaminant discharge

A variety of contaminants can be discharged into the marine environment as a result of petroleum extraction operations. Waste discharges associated with a petroleum facility include drilling well fluids, produced waters, surface runoff and deck drainage, and solid-waste from wells (i.e., drilling mud and cuttings) (NPFMC 1999). In addition to crude oil spills, chemical, diesel, and other contaminant spills can occur with petroleum-related activities (NPFMC 1999).

Produced waters contain finely dispersed oil droplets that can stay suspended in the water column or can settle out into sediments. Produced waters are generally more saline than seawater and contain elevated concentrations of radionuclides, metals, and other contaminants. Elevated levels of contaminated sediments typically extend up to 300 m from the discharge point (NRC 2003). In estuarine waters, higher saline produced waters can affect the salt wedge and form dense saltwater plumes.

The discharge of oil drilling mud can change the chemical and physical characteristics of benthic sediments at the disposal site by introducing toxic chemical constituents. The addition of contaminants can reduce or eliminate the suitability of the water column and substrate as habitat for fish species and their prey. The discharge of oil-based drill cuttings are currently not permitted in US waters; however, where oil-based drill cuttings have been discharged, there is evidence that sediment contamination and benthic impacts can occur up to 2 km from the production platform (NRC 2003).

The petroleum refining process converts crude oil into gasoline, home heating oil, and other refined products. The process of refining crude oil into various petroleum products produces effluents, which can degrade coastal water quality. Oil refinery effluents contain many different chemicals at different concentrations including ammonia, sulphides, phenol, and hydrocarbons. Toxicity tests have shown that most refinery effluents are toxic, but to varying extents. Some species are more sensitive and the toxicity may vary throughout the life cycle. Experiments have shown that not only can the effluents be lethal, but they can often have sublethal effects on growth and reproduction (Wake 2005). Field studies have shown that oil refinery effluents often have an adverse impact on aquatic organisms (i.e., an absence of all or most species), which is more pronounced in the area closest to the outfall (Wake 2005).

The operation of oil tankers can discharge contaminants into the water column and result in impacts to pelagic and benthic organisms. Older tankers that do not have segregated ballast tanks (i.e., completely separated from the oil cargo and fuel systems) can discharge ballast water containing contaminants (NRC 2003).

Discharge of debris

Petroleum extraction and transportation can result in the discharge of various types of debris, including domestic wastewater generated from offshore facilities, solid-waste from wells (i.e., drilling mud and cuttings), and other trash and debris from human activities associated with the facility (NPFMC 1999). Debris, either floating on the surface, suspended in the water column, covering the benthos, or along the shoreline can have deleterious impacts on fish and shellfish within riverine habitat, as well as in benthic and pelagic habitats in the marine environment (NEFMC 1998). Debris from petroleum extraction and transportation activities can be ingested by fish (Hoagland and Kite-Powell 1997). Reduction and degradation of habitat by debris can alter community structure and affect the sustainability of fisheries.

Oil spills

In even moderate quantities, oil discharged into the environment can affect habitats and living marine resources. Accidental discharge of oil can occur during almost any stage of exploration, development, or production on the OCS and in nearshore coastal areas and can occur from a number of sources, including equipment malfunction, ship collisions, pipeline breaks, other human error, or severe storms (Hanson et al. 2003). Oil spills can also be attributed to support activities associated with product recovery and transportation and can also involve various contaminants including hazardous chemicals and diesel fuel (NPFMC 1999).

Oil, characterized as petroleum and any derivatives, can be a major stressor to inshore fish habitats. Oil can kill marine organisms, reduce their fitness through sublethal effects, and disrupt the structure and function of the marine ecosystem (NRC 2003). Short-term impacts include interference with the reproduction, development, growth and behavior (e.g., spawning and feeding) of fishes, especially at early life-history stages (Gould et al. 1994). Petroleum compounds are known to have carcinogenic and mutagenic properties (Larsen 1992). Various levels of toxicity have been observed in Atlantic herring (*Clupea harengus*) eggs and larvae exposed to crude oil in concentrations of 1-20 ml/L (Blaxter and Hunter 1982). 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. These impacts may eventually lead to disruption of community organization and dynamics in affected regions. Oil can persist in sediments for years after the initial contamination (NRC 2003), interfering with physiological and metabolic processes of demersal fishes (Vandermeulen and Mossman 1996).

Oil spills can have adverse effects to both subtidal and intertidal vegetation. Direct exposure to petroleum can lead to die off of submerged aquatic vegetation (SAV) in the first year of exposure. Certain species which propagate by lateral root growth rather than seed germination may be less susceptible to oil in the sediment (NRC 2003). Oil has been demonstrated to disrupt the growth of vegetation in estuarine habitats (Lin and Mendelssohn 1996). Kelp located in low energy environments can retain oil in their holdfasts for extended periods of time. Oil spills are known to cause severe and long-term damage to salt marshes through the covering of plants and contamination of sediments. Lighter and more refined oils such as No. 2 fuel oil are extremely toxic to smooth cordgrass (*Spartina alterniflora*) (NRC 2003). Impacts to salt marsh habitats from oil spills depend on type, coverage, and amount of oil. Oil spills within salt marshes will likely have a greater impact in the spring growing season, compared to the dormant periods in the fall and winter.

Habitats that are susceptible to damage from oil spills include the low-energy coastal bays and estuaries where heavy deposits of oil may accumulate and essentially smother intertidal and salt marsh wetland communities. High-energy cobble environments are also susceptible to oil spills, as oil is driven into sediments through wave action. For example, many of the beaches in Prince William Sound, AK, with the highest persistence of oil following the *Exxon Valdez* oil spill were high-energy environments containing large cobbles overlain with boulders. These beaches were pounded by storm waves following the spill, which drove the oil into and well below the surface (Michel and Hayes 1999). Oil contamination in sediments may persist for years. For example, subsurface oil was detected in beach sediments of Prince William Sound twelve years after the *Exxon Valdez* oil spill, much of it unweathered and more prevalent in the lower intertidal biotic zone than at higher tidal elevations (Short et al. 2002).

Oil can have severe detrimental impacts on offshore habitats, although the effects may not be as acute as in inshore, sheltered areas. Offshore spills or wellhead blowouts can produce an oil slick on surface waters which can disrupt entire pelagic communities (i.e., phytoplankton and zooplankton). The disruption of plankton communities can interfere with the reproduction, development, growth, and behavior of fishes by altering an important prey base.

Physical and biological forces act to reduce oil concentrations (Hanson et al. 2003). Generally, the lighter fraction aromatic hydrocarbons evaporate rapidly, particularly during periods of high wind and wave activity. Heavier oil fractions typically pass through the water column and settle to the bottom. Suspended sediments can adsorb and carry oil to the seabed. Hydrocarbons may be solubilized by wave action which may enhance adsorption to sediments, which then sink to the seabed and contaminate benthic sediments (Hanson et al. 2003). Tides and hydraulic gradients allow movement of soluble and slightly soluble contaminants (e.g., oil) from beaches to surrounding streams in the hyporheic zone (i.e., the saturated zone under a river or stream, comprising substrate with the interstices filled with water) where pink salmon (*Oncorynchus* *gorbuscha*) eggs incubate (Carls et al. 2003). Oil can reach nearshore areas and affect productive nursery grounds, such as estuaries that support high densities of fish eggs and larvae. An oil spill near a particularly important hydrological zone, such as a gyre where fish or invertebrate larvae are concentrated, could also result in a disproportionately high loss of a population of marine organisms (Hanson et al. 2003). Epipelagic biota, such as eggs, larvae and other planktonic organisms, would be at risk from an oil spill. Planktonic organisms cannot actively avoid exposure, and their small size means contaminants may be absorbed quickly. In addition, their proximity to the sea surface can increase the toxicity of hydrocarbons several-fold and make them more vulnerable to photo-enhanced toxicity effects (Hanson et al. 2003).

Many factors determine the degree of damage from a spill, including the composition of the petroleum compound, the size and duration of the spill, the geographic location of the spill, and the weathering process present (NRC 2003). Although oil is toxic to all marine organisms at high concentrations, certain species and life history stages of organisms appear to be more sensitive than others. In general, the early life stages (i.e., eggs and larvae) are most sensitive, juveniles are less sensitive, and adults least so (Rice et al. 2000). Some marine species may be particularly susceptible to hydrocarbon spills if they require specific habitat types in localized areas and utilize enclosed water bodies, like estuaries or bays (Stewart and Arnold 1994).

Small but chronic oil spills may be a particular problem to the coastal ecosystem because residual oil can build up in sediments. Low-levels of petroleum components from such chronic pollution have been shown to accumulate in fish tissues and cause lethal and sublethal effects, particularly at embryonic stages. Effects on Atlantic salmon (*Salmo salar*) from low-level chronic exposure to petroleum components and byproducts (i.e., polycyclic aromatic hydrocarbons [PAH]) have been shown to increase embryo mortality, reduce growth (Heintz et al. 2000), and lower the return rates of adults returning to natal streams (Wertheimer et al. 2000).

As spilled petroleum products become weathered, the aromatic fraction of oil is dominated by PAH as the lighter aromatic components evaporate into the atmosphere or are degraded. Because of its low solubility in water, PAH concentrations probably contribute little to acute toxicity (Hanson et al. 2003). However, lipophilic PAH (those likely to be bonded to fat compounds) may cause physiological injury if they accumulate in tissues after exposure (Carls et al. 2003; Heintz et al. 2000). Even concentrations of oil that are diluted sufficiently to not cause acute impacts in marine organisms may alter certain behavior or physiological patterns. For example, "fatty change," a degenerative disease of the liver, can occur from chronic exposure to organic contaminants such as oil (Freeman et al. 1981).

Sublethal effects that may occur with exposure to PAH include impairment of feeding mechanisms for benthic fish and shellfish, growth and development rates, energetics, reproductive output, juvenile recruitment rates, increased susceptibility to disease and other histopathic disorders (Capuzzo 1987), and physical abnormalities in fish larvae (Urho and Hudd 1989). Effects of exposure to PAH in benthic species of fish include liver lesions, inhibited gonadal growth, inhibited spawning, reduced egg viability and reduced growth (Johnson et al. 2002). Gould et al. (1994) summarized various toxicity responses to winter flounder (*Pseudopleuronectes americanus*) exposed to PAH and other petroleum-derived contaminants, including liver and spleen diseases, immunosuppression responses, tissue necrosis, altered blood chemistry, gill tissue clubbing, mucus hypersecretion, altered sex hormone levels, and altered reproductive impairments. For Atlantic cod (*Gadus morhua*) exposed to various petroleum products, responses included reduced growth rates, gill hyperplasia, increased skin pigmentation, hypertrophy of gall bladder, liver disease, delayed spermatogenesis, retarded gonadal development and other reproductive impairments, skin lesions, and higher parasitic infections (Gould et al. 1994).

Oil spill clean-up activities

There are a number of oil spill response and cleanup methods available. Chemical dispersants are used primarily in open water environments. Dispersants contain surfactant chemical that under proper mixing conditions and concentrations attach to oil molecules and reduce the interfacial tension between oil molecules (NOAA 1992). This allows oil molecules to break apart and thus break down the oil slick. Depending on the environmental conditions and biological resource present, dispersants can result in acute toxicity. Exposure to high concentrations of oil dispersants has been shown to block the fertilization of eggs and induce rapid cytolysis of developing eggs and larvae in Atlantic cod (Lonning and Falk-Petersen 1978). Other methods of cleanup for open water spills include in-situ burning and nutrient and microbial remediation. In each case, impacts are dependent on the resources present in the particular location. Other forms of shoreline cleanup include the use of sorbents, trenching, sediment removal, and water flooding/pressure washing. Sediment removal and pressure washing will result in direct impact to the benthos. Trampling and cutting of salt marsh vegetation during cleanup activities can be severe, causing damage to plants and forcing oil into the sediments. However, impacts associated with the cleanup activities need to be weighed against the impacts created by the the spill itself.

Siltation, sedimentation, and turbidity

Exploratory and construction activities may result in resuspension of fine-grained mineral particles, usually smaller than silt, in the water column. Fish and invertebrate habitat may be adversely affected by elevated levels of suspended particles (Arruda et al. 1983), which can result in both lethal and sublethal impacts to marine organisms (Newcombe and MacDonald 1991; Newcombe and Jensen 1996). Short-term impacts from increases in suspended particles may include high turbidity, reduced light, and sedimentation which may lead to the loss or complexity of benthic habitat (USFWS and NMFS 1999). Suspended particles can reduce light penetration and lower the rate of photosynthesis and the primary productivity of the aquatic area, especially if the turbidity is persistent (Gowen 1978). Groundfish and other fish species can suffer reduced feeding ability and limited growth if high levels of suspended particles persist in the water column. Other problems associated with suspended solids include disrupted respiration and water transport rates in marine organisms, reduced filtering efficiencies in invertebrates, reduced egg buoyancy, disrupted ichthyoplankton development, reduced growth and survival of filter feeders, and decreased foraging efficiency of sight-feeders (Gowen 1978; Messieh et al. 1991; Barr 1993). Demersal eggs of fish and invertebrates can be adversely impacted by sediment deposition and suffocation. For example, hatching is delayed for striped bass (*Morone saxatilis*) and white perch (*Morone americana*) exposed to sediment concentrations as low as 100 mg/L for 1 day (Wilber and Clarke 2001). Berry et al. (2004) reported a decreased hatching success for winter flounder eggs with increasing depth of burial by sediment. No hatching occurred at burial depths of approximately 2 mm. Breitburg (1988) found the predation rates of striped bass larvae on copepods to decrease by 40% when exposed to high turbidity conditions in the laboratory. Anadromous fish passage in estuarine and riverine environments can also be adversely impacted by increased turbidity. For example in laboratory experiments, 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 sediment concentrations with an “alarm reaction” (Chiasson 1993).

Shallow water environments, rocky reefs, nearshore and offshore rises, salt and freshwater marshes (wetlands), and estuaries are more likely to be adversely impacted than are open-water habitats. This is due, in part, to their higher sustained biomass and lower water volumes, which decrease their ability to dilute and disperse suspended sediments (Gowen 1978).

Conservation recommendations and best management practices for petroleum exploration, production, and transportation (adapted from Hanson et al. 2003)

1. Conduct preconstruction biological surveys in consultation with resource agencies to determine the extent and composition of biological populations or habitat in the proposed impact area. Construction should be sited to minimize impacts to fishery resources.
2. Avoid the discharge of produced waters into marine and estuarine environments. Reinject produced waters into the oil formation whenever possible.
3. Avoid discharge of drilling mud and cuttings into the marine, estuarine, and riverine environment.
4. Avoid placing roads and bridges and structures associated with petroleum exploration and production in the nearshore marine environment. Particular care should be made to avoid SAV, intertidal flats, and salt marsh habitat.
5. Use methods to transport oil and gas that limit the need for handling in sensitive fishery habitats.
6. Use horizontal directional drilling for installation of pipelines in areas containing sensitive habitats, whenever possible.
7. Provide for monitoring and leak detection systems at oil extraction, production, and transportation facilities that preclude oil from entering the environment.
8. Evaluate impacts to habitat during the decommissioning phase, including impacts during the demolition phase.
9. Schedule dredging and excavation activities when the fewest species and least vulnerable life stages are present. Appropriate work windows can be established based on the multiple season biological sampling. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
10. Ensure that oil extraction, production, and transportation facilities have developed and implemented adequate oil spill response plans. Assist government agencies responsible for oil spills (e.g., US Coast Guard, state and local resource agencies) in developing response plans and protocols, including identification of sensitive marine habitats and development and implementation of appropriate oil spill-response measures.
11. Potential adverse impacts to marine resources from oil spill clean-up operations should be weighed against the anticipated adverse affects of the oil spill itself. The use of chemical dispersants in nearshore areas where sensitive habitats are present should be avoided.
12. Address the cumulative impacts of past, present, and foreseeable future development projects on aquatic habitats by considering them in the review process for petroleum exploration, production, and transportation projects.

Liquefied Natural Gas (LNG)

Introduction

Liquefied Natural Gas (LNG) is expected to provide a large proportion of the future energy needs in the northeastern United States. In recent years there has been an increase in proposals for new LNG facilities, including both onshore and offshore facilities from Maine to Delaware. In the northeastern United States, there are currently onshore LNG facilities operating in Everett, MA, and Cove Point, MD, and two offshore LNG facilities have been approved to operate in Massachusetts Bay.

The LNG process cools natural gas to its liquid form at approximately -260 degrees Fahrenheit (F). This reduces the volume of natural gas to approximately 1/600th of its gaseous state volume, making it possible for economical transportation with tankers. Upon arrival at the destination, the LNG is either regasified onshore or offshore and sent out into an existing pipeline infrastructure, or transported onshore for storage and future regasification. The process of regasification occurs when LNG is heated and converted back to its gaseous state. LNG facilities can utilize either "open loop," "closed loop," or "combined loop" systems for regasification. Open loop systems utilize warm seawater for regasification, and closed loop systems generally utilize a recirculating mixture of ethylene glycol for regasification. Combined loop systems utilize a combination of the two systems.

Onshore LNG facilities generally include a deepwater access channel, land-based facilities for regasification and distribution, and storage facilities. Offshore facilities generally include some type of a deepwater port with a regasification facility and pipelines to transport natural gas into existing gas distribution pipelines or onshore storage facilities. Deepwater ports require specific water depths and generally include some form of exclusion zone for LNG vessel and/or port facility security.

Habitat conversion and loss

The conversion of habitat and/or the loss of benthic habitats can occur from the construction and operation of LNG facilities. The placement of pipelines and associated structures on the seafloor can impact benthic habitats from physical occupation and conversion of the seafloor. The installation of pipelines can impact shellfish beds, hard-bottomed habitats, and SAV (Gowen 1978). Plowing or trenching for pipeline installation and side-casting of material can lead to a conversion of substrate and habitat. Placement of anchors for the construction of the deepwater port facilities can have direct impact to the substrate and benthos.

Because of the large size of LNG tankers, dredging may need to occur in order to access onshore terminals. The deepening of channel areas and turning basins can result in permanent and temporary dredging impacts to fishery habitat, including the loss of spawning and juvenile development habitat caused by changes in bathymetry, suitable substrate type, and sedimentation. Disruption of the areas from dredging and sedimentation may cause spawning fish to leave the area for more suitable spawning conditions. Dredging, as well as the equipment used in the process such as pipelines, may damage or destroy other sensitive habitats such as emergent marshes and SAV, including eelgrass beds (Mills and Fonseca 2003) and macroalgae beds. The stabilization and hardening of shorelines for the development of upland facilities can lead to a direct loss of SAV, intertidal mudflats, and salt marshes that serve as important habitat for a variety of living marine resources. See the Marine Transportation, Offshore Dredging and Disposal, and Coastal Development chapters for more detailed information on impacts from dredging.

Discharge of contaminants

Discharge of contaminants can occur as a result of spills during offloading procedures associated with either onshore or offshore facilities. There is limited information and experience regarding the aquatic impacts resulting from an LNG spill; however, because of the toxic nature of natural gas, acute impacts to nearby resources and habitats can be expected.

Biocides (e.g., copper and aluminum compounds) are often utilized in the hydrostatic testing of pipelines. LNG tankers utilize large amounts of seawater for regasification purposes (i.e., open-loop system), for engine cooling, and for ship ballast water. Biocides are commonly utilized to prevent pipeline and engine fouling from marine organisms and are subsequently discharged into surrounding waters. 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). The release of contaminants can reduce or eliminate the suitability of water bodies as habitat for fish species and their prey. In addition, contaminants, such as copper and aluminum, can accumulate in sediments and become toxic to organisms contacting or feeding on the bottom.

Discharge of debris

LNG facilities can result in the discharge of debris, including domestic waste waters generated from the offshore facility, and other trash and debris from human activities associated with the facility (NPFMC 1999). Impacts from the discharge of debris from LNG are similar to those described in the Petroleum Exploration, Production, and Transportation section of this chapter.

Siltation, sedimentation, and turbidity

LNG construction activities may result in increased suspended sediment in the water column caused by dredging, the installation of pipelines, anchors and chains, and the movement of vessels through confined areas, and upland site development. Impacts from siltation and sedimentation from LNG are similar to those described in the Petroleum Exploration, Production, and Transportation section of this chapter.

Entrainment and impingement

Intake structures for traditional power plants can result in impingement and entrainment of marine organisms through the use of seawater for cooling purposes (Enright 1977; Helvey 1985; Callaghan 2004). Likewise, intake structures utilized for the LNG regasification process can result in impingement and entrainment of living marine resources. “Open-loop” LNG regasification systems utilize seawater for warming into a gaseous state and are typically utilized when ambient water temperatures are greater than about 45°F. In addition, “combined loop” systems can utilize seawater for partial regasification. Depending on the geographic location and the water depth of the intake pipe, phytoplankton, zooplankton, and fish eggs and larvae can be entrained into the system. Juvenile fish can also be impinged on screens of water intake structures (Hanson et al. 1977; Hanson et al. 2003). Normal ship operations utilize intake structures for ballast water and engine cooling and can result in additional impingement and entrainment of resources, as well.

The entrainment and impingement impacts on aquatic organisms from LNG facilities have the potential to be substantial. For example, an assessment of impacts of a proposed LNG facility in the Gulf of Mexico determined that an open-loop regasification system could utilize 176 million gallons of water per day, which may entrain 1.6 billion fish and 60 million shrimp larvae per year, 3.3 billion fish eggs per year, and 500 billion zooplankton per year (R. Ruebsamen, pers. comm.). Additional entrainment and impingement impacts were expected for vessel ballast and cooling water uses. In the northeastern United States, an offshore LNG regasification facility approved in Massachusetts Bay with a closed-loop system has estimated annual mortality rates caused by vessel ballast and cooling water for the eggs and larvae for Atlantic mackerel (*Scomber scombrus*), pollock (*Pollachius virens*), yellowtail flounder (*Limanda ferruginea*), and Atlantic cod of 8.5 million, 7.8 million, 411,000, and 569,000, respectively (USCG 2006).

Alteration of temperature regimes

The operation of LNG facilities can result in the alteration of temperature regimes. Discharge of water from engine cooling operations can be at temperatures up to 10°F higher than surrounding waters. Water utilized for the purposes of regasification could be discharged at temperatures colder than the surrounding water by about 10-15°F. Changes in water temperatures can alter physiological functions of marine organisms, including respiration, metabolism, reproduction, and growth. In riverine and estuarine environments, changes to water temperatures can impact the egg and juvenile life stages of Atlantic salmon (USFWS and NMFS 1999). Thermal effluent in inshore habitat can cause severe problems by directly altering the benthic community or adversely affecting marine organisms, especially egg and larval life stages (Pilati 1976; Rogers 1976). For example, the seaward migration of juvenile American shad (*Alosa sapidissima*) are cued to water temperatures (Richkus 1974; MacKenzie et al. 1985), and temperature influences biochemical processes of the environment and the behavior (e.g., migration) and physiology (e.g., metabolism) of marine organisms (Blaxter 1969; Stanley and Colby 1971).

Alteration of hydrological regimes

The operation of LNG facilities can affect the hydrology of confined waterbodies, waterbodies with limited flows such as streams and rivers, and estuaries fed by streams and rivers. Depending upon the characteristics of the waterbody and the nature of the water intake and discharge, altered stream flow can result in reductions in stream flow and subsequent degradation of ecosystem functions (Reiser et al. 2004).

Alteration of salinity regimes

The operation of LNG tankers can result in the alteration of hydrological regimes caused by the discharge of brine from onboard desalination operations. For example, the operation of LNG tankers within riverine and estuarine environments can impact anadromous fish by altering salinity regimes (Dodson et al. 1972; Leggett and O'Boyle 1976) and affecting the ability of fish to access migration corridors.

Underwater noise

Underwater noise sources generate sound pressure that can disrupt or damage marine life. LNG activities generate noise from construction, production facility operations, and tanker traffic. Larvae and young fish are particularly sensitive to noise generated from underwater seismic equipment. It is also known that noise in the marine environment may adversely affect marine mammals by causing them to change behavior (e.g., movement, feeding), interfering with echolocation and communication or injuring hearing organs (Richardson et al. 1995). Noise issues related to LNG tanker traffic may adversely affect fishery resources in the marine environment, particularly in estuarine areas where some LNG port activities are located or proposed. A more thorough review of underwater noise can be found in the chapter on Global Effects and Other Impacts.

Exclusion zones

Because of security concerns, LNG tankers and terminals include safety and exclusion areas. Different types of restrictions are put in place based on the distance from the facility. However, restrictions on commercial and recreational fishing activities around the LNG facilities can lead to a displacement of fishing effort to other/adjacent areas. This in turn, may increase fishing effort and habitat impacts to more ecologically sensitive areas.

Introduction of invasive species

Introductions of nonnative invasive species into marine and estuarine waters are a significant threat to living marine resources in the United States (Carlton 2001). Nonnative species can be released unintentionally when ships release ballast water (Hanson et al. 2003; Niimi 2004). Hundreds of species have been introduced into United States waters from overseas and from other regions around North America, including finfish, shellfish, phytoplankton, bacteria, viruses, and pathogens (Drake et al. 2005). LNG tankers entering US waters are generally loaded with cargo and do not need to release large amounts of ballast water. However, even small amounts of released ballast water have the potential to contain invasive exotic species. In addition, as vessels are unloaded and ballast is taken on in US waters, the water may contain species that are potentially invasive to other locations. The transportation of nonindigenous organisms to new environments can have severe impacts on habitat (Omori et al. 1994), change the natural community structure and dynamics, lower the overall fitness and genetic diversity of natural stocks, and pass and/or introduce exotic lethal disease. Refer to the chapters on Marine Transportation and Introduced/Nuisance Species and Aquaculture for more information on invasive species and shipping.

Conservation recommendations and best management practices for LNG facilities

1. Conduct preconstruction biological surveys in consultation with resource agencies to determine the extent and composition of biological populations or habitat in the proposed impact area.
2. Recommend the use of "closed loop" systems, which minimize the volume of water utilized for regasification, over "open loop" systems. This will serve to minimize the level of impingement and entrainment of living marine resources.
3. Locate facilities that use surface waters for regassification and engine cooling purposes away from areas of high biological productivity, such as estuaries.
4. Design intake structures to minimize entrainment or impingement.
5. Regulate discharge temperatures (both heated and cooled effluent) such that they do not appreciably alter the temperature regimes of the receiving waters, which could cause a change in species assemblages and ecosystem function. Strategies should be implemented to diffuse the heated effluent.
6. Avoid the use of biocides (e.g., aluminum, copper, chlorine compounds) to prevent fouling where possible. The least damaging antifouling alternatives should be implemented.
7. Implement operational monitoring plans to analyze impacts resulting from intake and discharge structures and link them to a plan for adaptive management.
8. Provide for monitoring and leak detection systems at natural gas production and transportation facilities that preclude gas from entering the environment.
9. Schedule dredging and excavation activities when the fewest species and least vulnerable life stages are present. Appropriate work windows can be established based on the multiple season biological sampling. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
10. Address cumulative impacts of past, present, and foreseeable future development projects on aquatic habitats by considering them in the project review process of LNG facilities construction and operations. Based on evaluation of the foreseeable impacts to fishery habitats, a determination can be made regarding the most suitable location and operational procedures for LNG facilities. Ideally, such an analysis would be done at the regional or national level based on natural gas usage and need.

11. Ensure that gas production and transportation facilities have developed and implemented adequate gas spill response plans. Assist government agencies responsible for gas spills (e.g., US Coast Guard, state and local resource agencies) in developing response plans and protocols, including identification of sensitive marine habitats and development and implementation of appropriate gas spill-response measures.

Offshore Wind Energy Facilities

Introduction

Offshore wind energy facilities (windmills) convert wind energy into electricity through the use of turbines. An offshore facility generally consists of a series of wind turbine generators, an inner-array of submarine electric cables that connect each of the turbines, and a single electric service platform (ESP). Electricity is transmitted from the ESP to an onshore facility through one or a series of submarine cables.

While there are no operating offshore wind facilities in the United States at the writing of this report, there is an increasing number of proposals to develop offshore wind facilities within the northeast region. The construction and operation of offshore wind facilities has the potential to adversely affect fishery habitats.

Habitat conversion and loss

The construction of offshore wind turbines and support structures can result in benthic habitat conversion and loss because of the physical occupation of the natural substrate. Scour protection around the structures, consisting of rock or concrete mattresses, can also lead to a conversion and loss of habitat. For example, the total seafloor area occupied by 130 wind turbines, ESP and associated scour mats for an offshore wind farm proposed in Nantucket Sound, MA, is expected to be approximately 3.21 acres (USACE 2004). Should scour around cables and the base of structures occur, subsequent substrate stabilization activity would lead to additional impact on benthic habitat. Likewise, the burial and installation of submarine cable arrays can impact the benthic habitat through temporary disturbance from plowing and from barge anchor damage. In some cases, plowing or trenching for cable installation can permanently convert benthic habitats when top layers of sediments are replaced with new material. The installation of cables and associated barge anchor damage can adversely affect SAV, if those resources are present in the project area. Cable maintenance, repairs, and decommissioning can also result in impacts to benthic resources and substrate.

Siltation, sedimentation, and turbidity

The construction of wind turbine and support structures can cause increased turbidity in the water column and sedimentation impacts on adjacent benthic habitats. Likewise, the subsurface installation of underwater cables can result in similar impacts. Most of these impacts are relatively short-term and should subside after construction is completed. Maintenance and repairs of wind turbines and submarine electric cables can be expected to persist during the operation of the wind generator facilities. Increased sedimentation and turbidity during the decommissioning of wind energy facilities could be greater than the construction impacts if all submarine structures were to be removed. Siltation, sedimentation, and turbidity impacts related to the construction and maintenance activities from offshore wind energy projects are similar to those described in the Petroleum Exploration, Production, and Transportation section of this chapter.

Alteration of hydrological regimes

The placement of wind energy facilities, especially large arrays or "farms," in marine and estuarine habitats may affect hydrological regimes by altering tidal and current patterns. Altered current patterns could affect the distribution of eggs and larvae and the distribution of species within estuaries and bays, as well as the migration patterns of anadromous fishes.

Alteration of electromagnetic fields

Background direct current electric fields originate from the metallic core of the Earth and the electric currents flowing in the upper layer of the Earth's crust. The strength of this geomagnetic field is highest at the magnetic poles and the lowest at the equator. Marine fishes, such as elasmobranches and anadromous fishes, utilize natural electromagnetic fields (EMFs) for navigation and migratory behavior (Gill et al. 2005). Studies have shown sharks and rays are capable of detecting artificial EMFs (Meyer et al. 2005), and some species have a remarkable sensitivity to electric fields in seawater (Kalmijn 1982). Some species of fish have shown sensitivity to underwater EMFs, including several species of sharks (i.e., *Scyliorhinus canicula, Mustelus canis,* and *Prionace glauca)* and thornback skate (*Raja clavata*) (Kalmijn 1982); and sea lamprey (*Petromyzon marinus*), eels (*Anguilla sp.*), Atlantic cod, plaice (*Pleuronectes platessa*), yellowfin tuna (*Thunnus albacares*) and Atlantic salmon (Gill et al. 2005). Electrical cables associated with offshore wind energy facilities produce EMFs (and induced electric fields) which could interfere with fish behavior. However, at the present time there is no conclusive evidence that EMFs have an adverse effect on marine species (Gill et al. 2005).

Underwater noise

Underwater noise during construction of turbines may have impacts to hearing in fish and may cause fish to disperse with possible disruption to their feeding and spawning patterns. Underwater noise from the operation of wind turbines may decrease the effective range for sound communication in fish and mask orientation signals (Wahlberg and Westerberg 2005). Atlantic salmon and cod have been shown to detect offshore windmills at a maximum distance of about .04 km to 25 km at high wind speeds (i.e., >13 m/s), and noise from turbines can lead to permanent avoidance by fish within ranges of about 4 m (Wahlberg and Westerberg 2005). Noise from construction of wind farms (e.g., pile driving) could have significant effects on fish (Hoffmann et al. 2000). It is also known that noise in the marine environment may adversely affect marine mammals by causing them to change behavior (e.g., movement, feeding), interfering with echolocation and communication or injuring hearing organs (Richardson et al. 1995). A more thorough review of underwater noise can be found in the chapter on Global Effects and Other Impacts.

Alteration of community structure

Offshore wind energy facilities have the potential to alter the local community structure of the marine ecosystem. There is significant debate as to whether the presence of underwater vertical structures (e.g., oil platforms) contribute to new fish production by providing additional spawning and settlement habitat or simply attract and concentrate existing fishes (Bohnsack et al. 1994; Pickering and Whitmarsh 1997; Bortone 1998). The aggregation of fish in the vicinity of the wind turbine structures may subject certain species to increased fishing. Additive and synergistic effects of multiple stressors, such as the presence of electric cables on the seafloor and underwater sound generated by the turbines, could have cumulative effects on marine ecosystem and community dynamics (e.g., predator-prey population densities, migration corridors).

Discharge of contaminants

An ESP serves as a connection point for the inner-array of cables as well as a staging area for maintenance activities. Hazardous materials that may be stored at the ESP include fluids from transformers, diesel fuel, oils, greases and coolants for pumps, fans and air compressors. Discharge of these contaminants into the water column can affect the water quality in the vicinity of the offshore wind facility. Further information regarding the impacts of oil spills and contaminants can be found in the Petroleum Exploration, Production, and Transportation section of this chapter, and the chapters on Coastal Development and Chemical Affects: Water Discharge Facilities of the report.

Conservation recommendations and best management practices for offshore wind energy facilities

1. Conduct preconstruction biological surveys in consultation with resource agencies to determine the extent and composition of biological populations or habitat in the proposed impact area.
2. Avoid placing cables associated with offshore wind facilities near sensitive benthic habitats, such as SAV.
3. Use horizontal directional drilling to avoid impacts to sensitive habitats, such as salt marshes and intertidal mudflats.
4. Make contingency plans and response equipment available to respond to spills associated with service platforms.
5. Use scour protection for turbines and associated structures and cables to the minimum practicable in order to avoid alteration and conversion of benthic habitat.
6. Bury cables to an adequate depth in order to minimize the need for maintenance activities and to reduce conflicts with other ocean uses.
7. Time construction of facilities to avoid impacts to sensitive life stages and species. 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 activities on aquatic habitats in the review process for offshore wind energy facilities construction and operations.

Wave and Tidal Energy Facilities

Introduction

Wave power facilities involve the construction of stationary or floating devices that are attached to the ocean floor, the shoreline, or a marine structure like a breakwater with exposure to adequate "wave climate." Ocean wave power systems can be utilized in the offshore or nearshore environments. Offshore systems can be situated in deep water, typically in depths greater than 40 m (131 ft). Some examples of offshore systems include the Salter Duck, which uses the bobbing motion of the waves to power a pump that creates electricity. Other offshore devices use hoses connected to floats that move with the waves. The rise and fall of the float stretches and relaxes the hoses, which pressurizes the water, which in turn rotates a turbine. In addition, some seagoing vessels can be built to capture the energy of offshore waves. These floating platforms create electricity by funneling waves through internal turbines.

Wave energy can be utilized to generate power from the nearshore area in three ways:

1. Floats or pitching devices generate electricity from the bobbing or pitching action of a floating object. The object can be mounted to a floating raft or to a device fixed on the ocean floor. A

similar device, the pendulor, is a wave-powered device consisting of a rectangular box, which is open to the sea at one end. A flap is hinged over the opening and the action of the waves causes the flap to swing back and forth. The motion powers a hydraulic pump and a generator.
2. Oscillating water columns generate electricity from the wave-driven rise and fall of water in a cylindrical shaft. The rising and falling water column drives air into and out of the top of the shaft, powering an air-driven turbine.
3. Wave surge or focusing devices, also called "tapered channel" or "tapchan" systems, rely on a shore-mounted structure to channel and concentrate the waves, driving them into an elevated reservoir. Water flow out of this reservoir is used to generate electricity by using standard hydropower technologies (USDOE 2003).

Tidal energy facilities are designed to generate power in tidal estuaries through the use of turbines. A barrage, or dam, can be placed across a tidal river or estuary. This design utilizes a build-up of water within a headpond to create a differential on either side (depending on the tide), and then the water is released to turn the turbines. While less efficient, tidal power facilities can also utilize water currents to turn turbines. Turbines can be designed in a number of ways and include the "helical-type" turbines, as well as the "propeller-type" turbines. Turbines are generally placed within areas of fast moving water with strong currents to take advantage of both ebb and flow tides. For impacts associated with conventional hydropower facilities, refer to the chapter on Alteration of Freshwater Systems.

Habitat conversion and loss

The construction of tidal and wave energy facilities includes the placement of structures within the water column, thus converting open water habitat to anthropogenic structure. The placement of support structures, transmission lines, and anchors on the substrate will result in a direct impact to benthic habitats which serve as feeding or spawning habitats for various species. Large-scale tidal power projects which utilize a barrage can cause major changes in the tidal elevations of the headpond which can affect intertidal habitat. Alterations in the range and duration of tide flow can adversely affect intertidal communities that rely on specific hydrological regimes. Mud and sand flats may be converted to subtidal habitat, while high saltmarsh areas that may be normally flooded only on the highest spring tides can become colonized by terrestrial vegetation and invasive species (Gordon 1994).

Siltation, sedimentation, and turbidity

Construction of tidal facilities in riverine and estuarine areas can result in increased sedimentation. Structures placed within riverine and estuarine habitats can reduce the natural transport of sediments and cause an accretion of silt and sediments within impoundments. Deposition of sediments can adversely impact benthic spawning habitats of various anadromous fish species, including riffle and pool complexes. Clean gravel substrates, which are preferred by rainbow smelt and Atlantic salmon, can be subjected to increased siltation from alterations in the sediment transport. Shallow water environments, rocky reefs, nearshore and offshore rises, salt, and freshwater marshes (wetlands), and estuaries are more likely to be adversely impacted than open-water habitats. This is due, in part, to their higher sustained biomass and lower water volumes, which decrease their ability to dilute and disperse suspended sediments (Gowen 1978). Impacts from siltation and sedimentation from wave and tidal power facilities are similar to those described in the Petroleum Exploration, Production, and Transportation section of this chapter.

Alteration of hydrological regimes

Water circulation patterns and the tidal regimes can be altered during the operation of a barrage-type tidal facility. This can result in poor tidal flushing of the headwaters of estuaries and rivers and can lead to decreased water quality and increases in water temperature (Rulifson and Dadswell 1987). Altered current patterns could affect the distribution of eggs and larvae and the distribution of species within estuaries and bays as well as the migration patterns of anadromous fishes. Hydrological regimes may also be impacted by flows passing through and around tidal turbines and support structures.

Entrainment, impingement, and other impacts to migration

Water control structures, such as dams, alter the flow, volume, and depth of water within impoundments and below the structures. Water impoundments tend to stratify the water column, increasing water temperatures and decreasing dissolved oxygen levels. Projects operating as “store and release” facilities can drastically affect downstream water flow and depth, resulting in dramatic fluctuations in habitat accessibility, acute temperature changes and an overall decline in water quality (NEFMC 1998). The construction of dams, with either inefficient or nonexistent fish bypass structures, has been a major cause of the population decline of US Atlantic salmon (USFWS and NMFS 1999). Tidal energy facilities located within estuaries or riverine environments have the potential to directly impact migrating fish (Dadswell et al. 1986). Dadswell and Rulifson (1994) reported various physical impacts to fish traversing low-head, tidal turbines in the Bay of Fundy, Canada, including mechanical strikes with turbine blades, shear damage, and pressure- and cavitation-related injuries/mortality. They found between 21-46% mortality rates for tagged American shad passing through the turbine. The physical presence of tidal power facilities can impact the return of diadromous fishes to natal rivers (Semple 1984). Refer to the chapter on Alteration of Freshwater Systems for further information on impacts from water control structures.

Alteration of electromagnetic fields

Electrical distribution cables associated with ocean wave-power facilities produce EMFs similar to offshore wind energy facilities and may interfere with fish behavior (Gill et al. 2005). Refer to the discussion under the Offshore Wind Energy Facilities in this chapter for information on the affects of EMFs.

Conservation recommendations and best management practices for wave and tidal energy facilities

1. Do not permit the construction of barrage-type tidal energy facilities because of the potential for large impacts to the ecosystem and migratory fishery resources.
2. Require preconstruction assessments for analysis of potential impacts to fishery resources for all projects. Assessments should include comprehensive monitoring of the timing, duration, and utilization of the area by diadromous and resident species, potential impacts from the project, and contingency planning using adaptive management.
3. Do not site projects in areas that may result in adverse effects to sensitive marine and estuarine resources and habitats.
4. Avoid project siting of any wave or tidal energy facility within riverine, estuarine, and marine ecosystems utilized by diadromous species.
5. Time construction of facilities to avoid impacts to sensitive life stages and species. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.

6. Include impacts associated with the decommissioning and/or dismanteling of wave or tidal energy facility as part of the environmental analyses. Contingency for removal of structures should be required as part of any permits or licenses.
7. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats in the review process for wave and tidal facilities construction and operations.

Cables and Pipelines

Introduction

With the continued development of coastal regions comes greater demand for the installation of cables, utility lines for power and other services, and pipelines for oil and gas. The installation of pipelines, utility lines, and cables can have direct and indirect impacts on the offshore, nearshore, estuarine, wetland, beach, and rocky shore coastal zone habitats.

Habitat conversion and loss

The installation of cables and pipelines can result in the loss of benthic habitat from dredging and plowing through the seafloor. This can result in a direct loss of benthic organisms, including shellfish. Construction impacts can result in long-term or permanent damage, depending on the degree and type of habitat disturbance and best management practices employed for a project. The installation of pipelines can impact shellfish beds, hard-bottomed habitats, and SAV (Gowen 1978). Cables can damage complex habitats containing epifaunal growth during installation, if allowed to "sweep" along the bottom while being positioned into the correct location. Shallow water environments, rocky reefs, nearshore and offshore rises, salt and freshwater marshes (wetlands), and estuaries are more likely to be adversely impacted than are open-water habitats. This is due to their higher sustained biomass and lower water volumes, which decrease their ability to dilute and disperse suspended sediments (Gowen 1978). Benthic organisms, especially prey species, may recolonize disturbed areas, but this may not occur if the composition of the substrate is drastically changed or if pipelines are left in place after production ends.

Pipelines installed on the seafloor or over coastal wetlands can alter the environment by causing erosion and scour around the pipes, resulting in escarpments on coastal dune and salt marshes, and on the seafloor. Alterations to the geomorphology of coastal habitats from pipelines can exacerbate shoreline erosion and fragment wetlands. Because vegetated coastal wetlands provide forage and protection to commercially important invertebrates and fish, marsh degradation caused by plant mortality, soil erosion, or submergence will eventually decrease productivity.

Pipelines are generally buried below ground by digging trenches or canals. Digging trenches may change the coastal hydrology by: (1) facilitating rapid drainage of interior marshes during low tides or low precipitation; (2) reducing or interrupting freshwater inflow and associated littoral sediments; and (3) allowing saltwater to move farther inland during periods of high tides (Chabreck 1972). Saltwater intrusion into freshwater marsh often causes a loss of salt-intolerant emergent plants and SAV (Chabreck 1972; Pezeshki et al. 1987). Soil erosion and a net loss of organic matter may also occur (Craig et al. 1979).

Conversion of benthic habitat can occur if cables and pipelines are not buried sufficiently within the substrate. Conversion of habitats can also occur in areas where a layer of fine sediment is underlain with coarser materials. Once these materials are plowed for pipeline/cable installation, they can be mixed with underlying coarse sediment, and thus, alter the substrate composition. This can adversely affect the habitat of benthic organisms which rely on soft sand or mud habitats. The armoring of pipeline with either rock or concrete can result in permanent habitat alterations if placed within soft substrate. The placement of cables and pipelines often necessitates removal of hard bottom or rocky habitats in the pipeline corridor. These habitats are removed by using explosives or mechanical fracturing and can result in a reduction of available hard bottom substrate and habitat complexity.

Subsea pipelines that are placed on the substrate have the potential to create physical barriers to benthic invertebrates during migration and movement. In particular, the migration of American lobster (*Homarus americanus*) between inshore and offshore habitats can be adversely affected if pipelines are not buried to sufficient depths (Fuller 2003). Furthermore, erosion around buried pipelines and cables can lead to uncovering of the structure and the formation of escarpements. This, in turn, can interfere with the migratory patterns of benthic species.

Siltation, sedimentation, and turbidity

The installation of cables and pipelines can lead to increased turbidity and subsequent sedimentation, caused by either the plowing or jetting method of installation. Elevated siltation and turbidity during cable and pipeline installation is typically short-term and restricted to the area surrounding the cable and pipeline corridor. However, pipelines that are left unburied and exposed can cause erosion of the substrate and cause persistent siltation and turbidity in the surrounding area. Maintenance activities related to cables and pipelines, as well as removal for decommissioned cables and pipelines, can release suspended sediments into the water column. Long-term effects of suspended sediment include reduced light penetration and lowered photosynthesis rates and the primary productivity of the area (Gowen 1978). Impacts from siltation, sedimentation, and turbidity from cables and pipelines are similar to those described in the Petroleum Exploration, Production, and Transportation section of this chapter.

Release of contaminants

Petroleum products can be released into the environment if pipelines are broken or ruptured by unintentional activities, such as shipping accidents or deterioration of pipelines. A review of impacts from petroleum spills can be found in the Petroleum Exploration, Production, and Transportation section of this chapter. In addition, resuspension of contaminants in sediments, such as metals and pesticides, during pipeline installation can have lethal and sublethal effects to fishery resources (Gowen 1978). Contaminants may have accumulated in coastal sediments from past industrial activities, particularly in heavily urbanized areas. Metals may initially inhibit reproduction and development of marine organisms, but at high concentrations they can directly or indirectly contaminate or kill fish and invertebrates. The early life-history stages of fish are the most susceptible to the toxic impacts associated with metals (Gould et al. 1994). The release of contaminants can reduce or eliminate the suitability of water bodies as habitat for fish species and their prey. In addition, contaminants, such as copper and aluminum, can accumulate in sediments and become toxic to organisms contacting or feeding on the bottom.

Impacts to sensitive wetland and subtidal habitats can be avoided during pipeline and cable installation using horizontal directional drilling techniques, which allow the pipe or cable to be installed in a horizontal drill hole below the substrate. "Frac-outs" (i.e., releases of drilling mud or other lubricants, such as bentonite mud) can occur during the drilling process, and material can escape through fractures in the underlying rock. This typically happens when the drill hole encounters a natural fracture in the rock or when insufficient precautions are taken to prevent new fractures from occurring. Fishery habitats can be adversely affected if a "frac-out" occurs during the installation process and discharges drilling mud or other contaminants into the surrounding area.

Cranford et al. (1999) found that chronic intermittent exposure to sea scallops (*Placopecten magellanicus*) of dilute concentrations of operational drilling wastes, characterized by acute lethal tests as practically nontoxic, can affect growth, reproductive success, and survival.

Maintenance of cables and pipelines can also result in subsequent impacts to the aquatic environment. The maintenance of pipelines includes the "pigging" of pipelines to clean out residual materials from time-to-time. The release of these materials into the surrounding environment can lead to water quality impacts and contamination of adjacent benthic habitats. For example, biocides (e.g., copper and aluminum compounds) are often utilized in the hydrostatic testing of pipelines and are subsequently discharged into surrounding waters. 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).

Alteration of electromagnetic fields

Underwater electrical distribution cables produce EMFs that may interfere with fish behavior (Gill et al. 2005). However, at the present time there is no conclusive evidence that EMFs have an adverse effect on marine species (Gill et al. 2005). See also the discussion of underwater EMFs in the Offshore Wind Energy Facilities section of this chapter and the Global Effects and Other Impacts chapter of the report.

Underwater noise

The installation of cables and pipelines can produce underwater noise that may disrupt or damage fishery resources. Noise from construction activities (e.g., pile driving) can have significant effects on fish (Hoffmann et al. 2000). Larvae and young fish are particularly sensitive to noise generated from underwater explosives during blasting. It is also known that noise in the marine environment may adversely affect marine mammals by causing them to change behavior (movement, feeding), interfering with echolocation and communication, or injuring hearing organs (Richardson et al. 1995).

Alteration of community structure

The construction of pipelines and other underwater structures has the potential to alter the local community structure of the marine ecosystem. There is significant debate as to whether the presence of underwater vertical structures (e.g., oil platforms) contribute to new fish production by providing additional spawning and settlement habitat or simply attract and concentrate existing fish within an area (Bohnsack et al. 1994; Pickering and Whitmarsh 1997; Bortone 1998). Underwater pipelines are anthropogenic structures that could have similar attraction and production issues relating to fishery management. As with wind turbines and offshore LNG facilities, aggregation of fishes in the vicinity of pipeline structures may subject certain species to increased fishing pressure. By altering the age and species composition in the area around pipelines, predator/prey interactions and reproduction can be altered, and these changes may have community-level affects on fisheries.

Conservation recommendations and best management practices for cables and pipelines (adapted from Hanson et al. 2003)

1. Align crossings along the least environmentally damaging route. Sensitive habitats such as hard-bottom (e.g., rocky reefs), SAV, oyster reefs, emergent marsh, and mud flats should be avoided.

2. Use horizontal directional drilling where cables or pipelines would cross sensitive habitats, such as intertidal mudflats and vegetated intertidal zones, to avoid surface disturbances. Measures should be employed to avoid/minimize impacts to sensitive fishery habitats from potential frac-outs, including:
    a. The use of nonpolluting, water-based lubricants should be required.
    b. Drill stem pressures should be monitored closely so that potential frac-outs can be identified.
    c. Drilling should be halted, if frac-outs are suspected.
    d. Above ground monitoring should be employed to identify potential frac-outs.
    e. Spill clean-up plan and protocols should be developed, and clean-up equipment should be on-site to quickly respond to frac-outs.
3. Avoid construction of permanent access channels since they disrupt natural drainage patterns and destroy wetlands through excavation, filling, and bank erosion.
4. Backfill excavated wetlands with either the same or comparable material capable of supporting similar wetland vegetation. Original marsh elevations should be restored.
5. Use existing rights-of-way whenever possible to lessen overall encroachment and disturbance of wetlands.
6. Bury pipelines and submerged cables where possible. Unburied pipelines or pipelines buried in areas where scouring or wave activity eventually exposes them can result in impacts to invertebrate migratory patterns.
7. Use silt curtains or other types of sediment control in order to protect sensitive habitats and resources.
8. Limit access for equipment to the immediate project area avoid access through sensitive resources.
9. Avoid the use of open trenching for installation. Methods in which the trench is immediately backfilled reduce the impact duration and should therefore be employed when possible.
10. Conduct construction during the time of year that will have the least impact on sensitive habitats and species. Appropriate work windows can be established based on the multiple season biological sampling. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
11. Evaluate impacts to habitat during the decommissioning phase, including impacts during the demolition phase and impacts resulting from permanent habitat losses.
12. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats in the review process for cable and pipeline construction and operations.
13. Ensure that oil and gas pipeline systems include leak detection capabilities to minimize potential impacts from spills.

References for Energy-related Activities

Arruda JA, Marzolf GR, Faulk RT. 1983. The role of suspended sediments in the nutrition of zooplankton in turbid reservoirs. Ecology 64(5):1225-35.

Barr BW. 1993. Environmental impacts of small boat navigation: vessel/sediment interactions and management implications. In: Magoon OT, editor. Coastal Zone '93: proceedings of the eighth Symposium on Coastal and Ocean Management; 1993 Jul 19-23; New Orleans, LA. American Shore and Beach Preservation Association. p 1756-70.

Berry WJ, Hinchey EK, Rubinstein NI, Klein-MacPhee G. 2004. Winter flounder, *Pseudopleuronectes americanus*, hatching success as a function of burial depth in the laboratory. In: Ninth flatfish biology conference- poster presentation; 2004 Dec 1-2; Westbrook, CT. Woods Hole (MA): Northeast Fisheries Science Center Reference Document 04-13.

Blaxter JHS. 1969. Development: eggs and larvae. In: Hoar WS, Randall DJ, editors. Fish physiology: Vol. 3, reproduction and growth, bioluminescence, pigments, and poisons. New York (NY): Academic Press. p 177-252.

Blaxter JHS. 1977. The effect of copper on the eggs and larvae of plaice and herring. Journal of the Marine Biological Association of the United Kingdom 57:849-59.

Blaxter JHS, Hunter JR. 1982. The biology of the clupeoid fishes. In: Blaxter JHS, Russel FS, Yonge M, editors. Advances in marine biology – Vol. 20. New York (NY): Academic Press. p 187-91.

Bohnsack JA, Harper DE, McClellan DB, Hulsbeck M. 1994. Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, U.S.A. Bulletin of Marine Science 55(2-3):796-823.

Bortone SA. 1998. Resolving the attraction-production dilemma in artificial reef research: Some yeas and nays. Fisheries 23(3):6-10.

Breitburg DL. 1988. Effects of turbidity on prey consumption by striped bass larvae. Transactions of the American Fisheries Society 117(1):72-7.

Callaghan TP. 2004. Entrainment and impingement of organisms in power plant cooling. In: Cleveland CJ, editor. Encyclopedia of Energy, Vol. 2. Amsterdam (The Netherlands): Elsevier. p 447-57.

Capuzzo JM. 1987. Biological effects of petroleum hydrocarbons: assessment of experimental results. In: Boesch DF, Rabalais NN, editors. Long-term environmental effects of offshore oil and gas development. London (England): Elsevier Applied Science. p 343-410.

Carls MG, Thomas RE, Lilly MR, Rice SD. 2003. Mechanism for transport of oil-contaminated groundwater into pink salmon redds. Marine Ecology Progress Series 248:245-55.

Carlton JT. 2001. Introduced species in U.S. coastal waters: environmental impacts and management priorities. Arlington (VA): Pew Oceans Commission. 28 p.

Chabreck RH. 1972. Vegetation, water, and soil characteristics of the Louisiana coastal region. Baton Rouge (LA): LSU Agricultural Experimental Station.

Chiasson AG. 1993. The effects of suspended sediment on rainbow smelt (*Osmerus mordax*): a laboratory investigation. Canadian Journal of Zoology 71:2419-24.

[CEQ] Council on Environmental Quality. 1977. Oil and gas in coastal lands and waters. Washington (DC): US Government Printing Office. 153 p.

Craig NJ, Turner RE, Day JW Jr. 1979. Land Loss in coastal Louisiana (U.S.A.). Environmental Management 3(2):133-44.

Cranford PJ, Gordon DC Jr, Lee K, Armsworthy SL, Tremblay GH. 1999. Chronic toxicity and physical disturbance effects of water- and oil-based drilling fluids and some major constituents on adult sea scallops (*Placopecten magellanicus*). Marine Environmental Research 48(3):225-56.

Dadswell MJ, Rulifson RA. 1994. Macrotidal estuaries: A region of collision between marine animals and tidal power development. Biological Journal of the Linnean Society 51(1-2):93-113.

Dadswell MJ, Rulifson RA, Daborn GR. 1986. Potential impact of large-scale tidal power developments in the upper Bay of Fundy on fisheries resources of the northwest Atlantic. Fisheries 11(4):26-35.

Dodson JJ, Leggett WC, Jones RA. 1972. The behavior of adult American shad (*Alosa sapidissima*) during migration from salt to fresh water as observed by ultrasonic tracking techniques. Journal of the Fisheries Research Board of Canada 29(10):1445-9.

Drake LA, Jenkins PT, Dobbs FC. 2005. Domestic and international arrivals of NOBOB (no ballast on board) vessels to lower Chesapeake Bay. Marine Pollution Bulletin 50(5):560-5.

Enright JT. 1977. Power plants and plankton. Marine Pollution Bulletin 8(7):158-61.

Freeman HC, Uthe JF, Sangalang GB, Garside ET. 1981. Contaminant/liver histopathology in Canadian Atlantic cod (*Gadus morhua*). Copenhagen (Denmark): ICES CM-1981/ E: 50.

Fuller C. editor. 2003. A workshop on the potential effects of the construction and operation of subsea pipelines on lobster movement and behavior. Blue Atlantic Lobster Workshop Proceedings; 2003 Jun 24; Providence, RI. Willington (CT): ENSR International. 152 p.

Gill AB, Gloyne-Phillips I, Neal KJ, Kimber JA. 2005. The potential effects of electromagnetic fields generated by sub-sea power cables associated with offshore wind farm developments on electrically and magnetically sensitive marine organisms-a review. Final Report: Collaborative Offshore Wind Energy Research into the Environment. London (England): COWRIE 1.5 Electromagnetic Fields Review. 128 p.

Gordon DC Jr. 1994. Intertidal ecology and potential power impacts, Bay of Fundy, Canada. Biological Journal of the Linnean Society 51(1-2):17-23.

Gould E, Clark PE, Thurberg FP. 1994. Pollutant effects on dermersal fishes. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 30-41.

Gowen AW. 1978. The environmental effects of outer continental shelf (OCS) pipelines. Initial findings. Boston (MA): New England River Basins Commission 4:24-43.

Hanson CH, White JR, Li HW. 1977. Entrapment and impingement of fishes by power plant cooling water intakes: an overview. Marine Fisheries Review 39:7-17.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Heintz RA, Rice SD, Wertheimer AC, Bradshaw RF, Thrower FP, Joyce JE, Short JW. 2000. Delayed effects on growth and marine survival of pink salmon *Oncorhynchus gorbuscha* after exposure to crude oil during embryonic development. Marine Ecology Progress Series 208:205-16.

Helvey M. 1985. Behavioral factors influencing fish entrapment at offshore cooling-water intake structures in southern California. Marine Fisheries Review 47:18-26.

Helvey M. 2002. Are southern California oil and gas platforms essential fish habitat? ICES Journal of Marine Science 59(Suppl.):S266-S271.

Hoagland P, Kite-Powell HL. 1997. Characterization and mitigation of marine debris in the Gulf of Maine. A report prepared for the U.S. Gulf of Maine Association under contract no. GM97-13. Gulf of Maine Council on the Marine Environment. 41 p.

Hoffman E, Astrup J, Larsen F, Munch-Petersen S. 2000. Effects of marine windfarms on the distribution of fish, shellfish, and marine mammals in the Horns Rev Area. Lyngby (Denmark): Danish Institute for Fisheries Research. Report to ELSAMPROJEKT A/S. Report No. Baggrundsrapp 24. 42 p.

Johnson LL, Collier TK, Stein JE. 2002. An analysis in support of sediment quality thresholds for polycyclic aromatic hydrocarbons (PAH) to protect estuarine fish. Aquatic Conservation: Marine and Freshwater Ecosystems 12(5):517-38.

Kalmijn AJ. 1982. The electric and magnetic field detection in Elasmobranch fishes. Science 218(4575):916-18.

Larsen PF. 1992. An overview of the environmental quality of the Gulf of Maine. In: The Gulf of Maine. Silver Spring (MD): NOAA Coastal Ocean Program Synthesis Series No. 1. p 71-95.

Leggett WC, O'Boyle RN. 1976. Osmotic stress and mortality in adult American shad during transfer from saltwater to freshwater. Journal of Fish Biology 8(6):459-69.

Lin Q, Mendelssohn IA. 1996. A comparative investigation of the effects of south Louisiana crude oil on the vegetation of fresh, brackish, and salt marshes. Marine Pollution Bulletin 32(2):202-9.

Lonning S, Falk-Petersen IB. 1978. The effects of oil dispersants on marine eggs and larvae. Astarte 11:135-8.

MacKenzie C, Weiss-Glanz LS, Moring JR. 1985. Species profiles: life histories and environmental requirements of coastal fishes and invertebrates. Washington (DC): US Fish and Wildlife Service. Biological Report 82(11.37). 18 p.

Marten K, Herzing D, Poole M, Newman-Allman K. 2001. The acoustic predation hypothesis: Linking underwater observations and recordings during Odontocete predation and observing the effects of loud impulsive sounds on fish. Aquatic Mammals 27(1):56-66.

Messieh SN, Rowell TW, Peer DL, Cranford PJ. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. Continental Shelf Research 11(8-10):1237-63.

Meyer CG, Holland KN, Papastamatiou YP. 2005. Sharks can detect changes in the geomagnetic field. Journal of the Royal Society Interface 2(2):129-30.

Michel J, Hayes MO. 1999. Weathering patterns of oil residues eight years after the *Exxon Valdez* oil spill. Marine Pollution Bulletin 38(10):855-63.

Mills KE, Fonseca MS. 2003. Mortality and productivity of eelgrass *Zostera marina* under conditions of experimental burial with two sediment types. Marine Ecology Progress Series 255:127-34.

[NOAA] National Oceanic and Atmospheric Administration. 1992. An introduction to coastal habitats and biological resources for oil spill response. Seattle (WA): Hazardous Materials Response and Assessment Division. Report No. HMRAD 92-4.

[NRC] National Research Council. 2003. Oil in the sea III: inputs, fates and effects. Washington (DC): National Academy Press.

Newcombe CP, Jensen JO. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4):693-727.

Newcombe CP, MacDonald DD. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11(1):72-82.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol. 1.

Niimi AJ. 2004. Role of container vessels in the introduction of exotic species. Marine Pollution Bulletin 49(9-10):778-82.

[NPFMC] North Pacific Fisheries Management Council. 1999. Environmental Assessment for Amendment 55 to the Fishery Management Plan for the Groundfish Fishery of the Bering Sea and Aleutian Islands Area; Amendment 55 to the Fishery Management Plan for Groundfish of the Gulf of Alaska; Amendment 8 to the Fishery Management Plan for the King and Tanner Crab Fisheries in the Bering Sea and Aleutian Islands; Amendment 5 to the Fishery Management Plan for Scallop Fisheries off Alaska; Amendment 5 to the Fishery Management Plan for the Salmon Fisheries in the EEZ off the Coast of Alaska, Essential Fish Habitat. Anchorage (AK): NPFMC. 367 p.

Omori M, VanderSpoel S, Norman CP. 1994. Impact of human activities on pelagic biogeography. Progress in Oceanography 34(2-3):211-9.

Pezeshki SR, Delaune RD, Patrick WH Jr. 1987. Response of the freshwater marsh species, *Panicum hemitomon* Schult., to increased salinity. Freshwater Biology 17(2):195-200.

Pickering H, Whitmarsh D. 1997. Artificial reefs and fisheries exploitation: a review of the 'attraction versus production' debate, the influence of design and its significance for policy. Fisheries Research 31(1-2):39-59.

Pilati DA. 1976. Cold shock: biological implications and a method for approximating transient environmental temperatures in the near-field region of a thermal discharge. Science of the Total Environment 6(3):227-37.

Reiser DW, Chapin D, DeVries P, Ramey MP. 2004. Flow regime and ecosystem interactions in spring dominated streams: Implications for selecting instream flow methods. Hydroecologie Appliquee 14(1):93-104.

Rice SD, Short JW, Heintz RA, Carls MG, Moles A. 2000. Life-history consequences of oil pollution in fish natal habitat. In: Catania P, editor. Energy 2000: the beginning of a new millennium. Lancaster (PA): Technomic Publishers. p 1210-5.

Richardson WJ, Greene CR, Malme CI, Thomas DH. 1995. Marine mammals and noise. San Diego (CA): Academic Press. 576 p.

Richkus WA. 1974. Factors influencing the seasonal and daily patterns of alewife (*Alosa pseudoharengus*) migration in a Rhode Island river. Journal of the Fisheries Research Board of Canada 31:1485-97.

Rogers CA. 1976. Effects of temperature and salinity on the survival of winter flounder embryos. Fisheries Bulletin 74(1):52-8.

Rulifson RA, Dadswell MJ. 1987. Mortality of fish passing through tidal, low-head hydropower turbines and possible mitigation strategies. In: Proceedings of Ocean 87: the ocean-an international workplace; 1987 Sep 28-Oct 1; Halifax, Nova Scotia. Columbia (MD): Marine Technology Society and IEEE Ocean Engineering Society. p 944-9.

Semple JR. 1984. Possible effects of tidal power development in the upper Bay of Fundy on anadromous fish passage. In: Update on the environmental consequences of tidal power in the upper reaches of the Bay of Fundy. Dartmouth (Nova Scotia): Canadian Technical Report of the Fisheries and Aquatic Sciences No. 1256:565-75.

Short JW, Lindeberg MR, Harris PM, Maselko J, Rice SD. 2002. Vertical oil distribution within the intertidal zone 12 years after the *Exxon Valdez* oil spill in Prince William Sound, Alaska. Proceedings of the 25th Arctic and Marine Oilspill Program Technical Seminar. Ottawa (Ontario): Minister of Supply and Services Canada. p 57-72.

Stanley JG, Colby PJ. 1971. Effects of temperature on electrolyte balance and osmoregulation in the alewife (*Alosa pseudoharengus*) in fresh and sea water. Transactions of the American Fisheries Society 100(4):624-38.

Stewart PL, Arnold SH. 1994. Environmental requirements Atlantic herring (*Clupea harengus harengus*) in eastern Canada and its response to human impacts. Dartmouth (Nova Scotia): Canadian Technical Report of Fisheries and Aquatic Sciences 2003: ix + 37p.

Urho L, Hudd R. 1989. Sublethal effects of an oil spill on fish (herring) larvae in the northern Quark, in the Baltic. In: Blaxter JHS, Gamble JC, Westernhagen HV, editors. The early life history of fish; 1988 Oct 3-5; Bergen, Norway. Third ICES Symposium: Rapports et Proces-Verbaux Des Reunions 191. Copenhagen (Denmark): International Council for the Exploration of the Sea. p 494.

[USACE] US Army Corps of Engineers. 2004. Cape Wind Energy Project Draft Environmental Impact Statement (Cape Wind DEIS). Vol. 1. Concord (MA): US Army Corp of Engineers, New England District.

[USCG] US Coast Guard. 2006. Environmental impact statement/environmental impact report for the Neptune LNG deepwater port license application. Washington (DC): USCG-2005-22611.

[USDOE] US Department of Energy. 2003. A consumer's guide to energy efficiency and renewable energy: Ocean wave power. [Internet]. [updated 2003 Feb 7; cited 2007 Dec 19]. Available from: http://www.eere.energy.gov/consumer/renewable_energy/ocean/index.cfm/mytopic=50009.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1999. Status review of anadromous Atlantic salmon in the United States. Hadley (MA): USFWS. p 1-131.

Vandermeulen JH, Mossman D. 1996. Sources of variability in seasonal hepatic microsomal oxygenase activity in winter flounder (*Pleuronectes americanus*) from a coal tar contaminated estuary. Canadian Journal of Fisheries and Aquatic Sciences 53:1741-53.

Wahlberg M, Westerberg H. 2005. Hearing in fish and their reactions to sounds from offshore wind farms. Marine Ecology Progress Series 288:295-309.

Wake H. 2005. Oil refineries: a review of their ecological impacts on the aquatic environment. Estuarine, Coastal and Shelf Science 62:131-40.

Wertheimer AC, Heintz RA, Thedinga JF, Maselko JM, Rice SD. 2000. Straying of adult pink salmon from their natal stream following embryonic exposure to weathered *Exxon Valdez* crude oil. Transactions of the American Fisheries Society 129(4):989-1004.

Wilber DH, Clarke DG. 2001. Biological effects of suspended sediments: A review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21(4):855-75.

CHAPTER FOUR: ALTERATON OF FRESHWATER SYSTEMS

Introduction

Freshwater riverine and riparian habitats located in the northeastern coastal United States provide important habitat for the growth, survival, and reproduction of diadromous fishes and are critical to maintaining healthy estuarine ecosystems. Some of the diadromous fish (species that migrate between freshwater and saltwater for specific life history functions) inhabiting the Northeast include Atlantic salmon (*Salmo salar*), striped bass (*Morone saxatilis*), alewife (*Alosa pseudoharengus*), blueback herring (*Alosa aestivalis*), American shad (*Alosa sapidissima*), rainbow smelt (*Osmerus mordax*), Atlantic sturgeon (*Acipenser oxyrinchus*), shortnose sturgeon (*Acipenser brevirostrum*), and American eel (*Anguilla rostrata*). Not only are diadromous fishes subject to environmental impacts in the marine environment, but they also encounter dams, pollution, effects of urbanization, and habitat changes in freshwater (Moring 2005). In addition, some forage species that are important prey for marine fisheries depend upon freshwater habitats for portions of their life cycle. The health and availability of freshwater systems and the preservation and maintenance of associated functions and values are vital to the diversity, health, and survival of marine fisheries.

Free flowing rivers, ponds, and lakes act as migratory corridors, spawning, nursery, and rearing areas and provide forage and refuge for life stages of these species. Riverine and riparian corridors, and palustrine and lacustrine wetlands provide important functions and values for resident and migratory fish, freshwater mussels, reptiles, amphibians, and insects (Chabreck 1988). Riparian corridors provide shade, nutrients, and habitat enhancing debris in riverine systems (Bilby and Ward 1991), which are essential elements necessary for these aquatic resources to thrive. In addition to supporting aquatic resources, freshwater wetlands perform important and broad ecological functions by reducing erosion, attenuating floodwater velocity and volume, improving water quality by the uptake of nutrients, and reducing sediment loads (Howard-Williams 1985; De Laney 1995; Fletcher 2003). Freshwater habitats are intricately connected to terrestrial and coastal ecosystems, making them vulnerable to a wide array of anthropogenic disturbances that can alter the functions, values, quantity, and accessibility of freshwater wetlands used by migratory fish (Beschta et al. 1987; Naiman 1992).

Biological, chemical, and physical threats to freshwater environments from terrestrial and aquatic sources have led to habitat fragmentation and degradation (Bodi and Erdheim 1986; Wilbur and Pentony 1999; USEPA 2000; Kerry et al. 2004). In particular, nonfishing activities, such as mining, dredging, fill placement, dam construction and alterations of hydrologic regimes, thermal discharges, and nonpoint source pollution have degraded and eliminated freshwater habitats (Zwick 1992; Wilbur and Pentony 1999; Hanson et al. 2003). Examples of nonpoint source pollution include urban stormwater and agricultural runoff (e.g., petroleum products, metals, pesticides, fertilizers, and animal wastes). Refer to the Coastal Development and Agriculture and Silviculture chapters for more detailed discussion on nonpoint source pollution. The federal Clean Water Act (CWA) has eliminated certain types of disposal activities, limited fill activities, and otherwise resulted in improved protection of the nation's wetlands and waterways. Despite these and other regulations to protect aquatic habitat, anthropogenic impacts continue, dramatically affecting fish habitat, including prey species and fisheries (Wilson and Gallaway 1997; Bodi and Erdheim 1986; Hanson et al. 2003; Ormerod 2003; Kerry et al. 2004).

Dam Construction and Operation

The history and effects of dam construction on passage and habitat is well documented (Larinier 2001; Heinz Center 2002). Among the major identified causative factors of the population demise of Atlantic salmon, dam construction and operation may be the most dramatic (NEFMC 1998; Parrish et al. 1998; USFWS and NMFS 1999). In the United States, 76,000 dams have been identified in the National Inventory of Dams by the US Army Corps of Engineers and the Federal Emergency Management Agency (Heinz Center 2002). This number may be as high as 2 million when small-scale dams are included (Graf 1993). Dam construction and operation in the northeastern United States have occurred for centuries to provide power generation, navigation, fire and farm ponds, reservoir formation, recreation, irrigation, and flood control. Important for the local economy when originally constructed, today many of these structures are obsolete, unused, abandoned, or decaying. Fish passages in any given river system may not be consistent or effective throughout, limiting the ability for Atlantic salmon and many other migratory and resident species to reach necessary habitat. Sections 18 and 10j of the Federal Power Act require fish passage and protection and mitigation for damages to fish and wildlife, respectively, at hydroelectric facilities.

The effects of dam construction and operation on fisheries and aquatic habitat include: (1) complete or partial upstream and downstream migratory impediment; (2) water quality and flow patterns alteration; (3) thermal impacts; (4) alterations to the floodplain, including riparian and coastal wetland systems and associated functions and values; (5) habitat fragmentation; (6) alteration to sediment and nutrient budgets; and (7) limitations on gene flow within populations.

Impaired fish passage

The construction of dams with either no fish passage or ineffective passage was the primary agent of the population decline of US Atlantic salmon (USFWS and NMFS 1999; NEFMC 1998). By 1950, less than 2% of the original habitat for Atlantic salmon in New England was accessible because of dams (Buchsbaum 2005). Dams physically obstruct passage and alter a broad range of habitat characteristics essential for passage and survival. Without any mechanism to get around a dam, there is no upstream passage to spawning and nursery habitat. Fish that gather at the base of the dam will either spawn in inadequate habitat, die, or return downstream without spawning. The presence of a fish passage structure does not necessarily ensure access to upstream habitat. Even with a structure in place, passage is contingent on many factors, including water-level fluctuations, altered seasonal and daily flow regimes, elevated temperatures, reduced water velocities, and discharge volumes (Haro et al. 2004).

Safe, timely, and effective downstream passage by fish is also hindered by dams. The time required for downstream migration is greatly increased because of reduced water flows within impoundments (Raymond 1979; Spence et al. 1996; PFMC 1999). This delay results in greater mortality associated with predation and the physiological stress associated with migration. Downstream passage for fish is hindered or prevented while passing over spillways and through turbines (Ruggles 1980; NEFMC 1998) and by entrainment or impingement on structures associated with a hydroelectric facility. Dadswell and Rulifson (1994) reported on the physical impacts observed in fish traversing low-head, tidal turbines in the Bay of Fundy, Canada, which included mechanical strikes with turbine blades, shear damage, and pressure- and cavitation-related injuries/mortality. They found 21-46% mortality rates for experimentally tagged American shad passing through the turbine.

Fragmentation of aquatic habitat caused by dams can result in a loss of genetic diversity and spawning potential that may make populations of fish more vulnerable to local extirpation and extinctions, particularly for species functioning as a metapopulation (Morita and Yamamoto 2002).

Altered hydrologic, salinity, and temperature regimes

Dams and dam operations alter flow patterns, volume, and depth of water within impoundments and below the dam. These hydrological alterations tend to increase water temperatures, stratify the water column, and decrease dissolved oxygen concentrations in the water impoundments. Projects operating as "store and release" facilities can drastically affect downstream water flow and depth, resulting in dramatic fluctuations in habitat accessibility, acute temperature changes, and overall water quality. Although large, impounding dams have the ability to alter the hydrology of large segments or entire rivers, smaller, run-of-the river dams that do not contain impoundments generally have little or no ability to alter downstream hydrology (Heinz Center 2002).

Reductions in river water temperatures are common below dams if the intake of the water is from lower levels of the reservoir. Stratification of reservoir water not only affects temperature but can create oxygen-poor conditions in deeper areas and, if these waters are released, can degrade the water quality of the downstream areas (Heinz Center 2002).

By design, dams often reduce peak flows as flood control measures. However, reductions of peak flows can decrease the physical integrity of the downstream river because the floodplains (including side channels, islands, bars, and beaches) are not as extensively connected to the river (Heinz Center 2002). In addition, dams can also reduce low flows during periods of drought and when dam operators reduce water releases in order to maintain water levels in the impoundments (Heinz Center 2002).

Dams with deep reservoirs have high hydrostatic pressures at the bottom and can force atmospheric gases into solution. If these waters are released below the dam, either by water spilling over dams or through turbines, it can cause dissolved gas supersaturation, resulting in injury or death to fish traversing the dam (NEFMC 1998; Heinz Center 2002).

Tidal fresh habitat is limited to a narrow zone in river systems where the water is tidally influenced, yet characteristically fresh (i.e., < 0.5 ppt salinity). This narrow habitat type may be altered or lost because of dam construction and operations.

Alteration of stream bed and stream morphology

The construction of a dam fragments habitat, altering both upstream and downstream biogeochemical processes and resulting in a wide array of direct and indirect cumulative impacts (Poff et al. 1997; Heinz Center 2002). Multiple habitat variables are affected by dams, principally streambed properties (Spence et al. 1996), the transport of sediments and large woody debris (Spence et al. 1996; PFMC 1999), and overall stream morphology.

Dams typically reduce peak flows as a flood control measure and can reduce low flows when water releases are reduced to save water during drought. As the range of flows in the river are decreased, the width of the active portion of the watershed is reduced and the river channel shrinks (Heinz Center 2002).

Altered sediment/large woody debris transport

Dams affect the physical integrity of watersheds by fragmenting the lengths of rivers, changing their hydrologic characteristics, and altering their sediment regimes by trapping most of the sediment entering the reservoirs and disrupting the sediment budget of the downstream landscape (Heinz Center 2002). Because water released from dams is relatively free of sediment, downstream reaches of rivers may be altered by increased particle size, erosion, channel shrinkage, and deactivation of floodplains (Heinz Center 2000).

Large woody debris (LWD) and other organic matter are often removed from rivers containing dams, as well as for other reasons, such as aesthetics, road and bridge maintenance, and commercial and recreational uses. Organic debris provides habitat for a variety of aquatic organisms, such as Atlantic salmon, by promoting habitat complexity, including the formation of pool and riffle complexes and undercut banks (Montgomery et al. 1995; Abbe and Montgomery 1996; Spence et al. 1996). Removing organic debris may change the structure, function, and value of the river system. From a broader perspective, removal of LWD from a river system disrupts a link between the forest and the sea (Maser and Sedell 1994; NRC 1996; Collins et al. 2002; Collins et al. 2003).

Riparian zone development and alteration of wetlands

Riparian wetlands may be lost to water level increases upstream and flow alterations downstream of the dam. Generally, the greater the storage capacity of a dam, the more extensive are the downstream geomorphological and biological impacts (Heinz Center 2002). Lost wetlands result in a loss of floodplain and flood storage capacity, and thus a reduced ability to provide flood control during storm events. A healthy riparian corridor is well vegetated, harbors prey items, contributes necessary nutrients, provides LWD that creates channel structure and cover for fish, and provides shade, which controls stream temperatures (Bilby and Ward 1991; Hanson et al. 2003). When vegetation is removed from riparian areas, water temperatures tend to increase and LWD is less common. The result is less refuge for fish, fundamental changes in channel structure (e.g., loss of pool habitats), instability of stream banks, and alteration of nutrient and prey sources within the river system (Hanson et al. 2003). Riparian zone development can be considered a secondary effect of dam construction. Residential, recreational, and commercial development may result from the associated impoundment.

Changes to native aquatic communities

Impoundments can concentrate predators and disease carrying organisms and disrupt fish development, thereby altering the community structure at various trophic levels and potentially changing the natural habitat and fishery dynamics of the aquatic habitat. In addition, the loss of wetlands by the increased impoundment level and reduction of freshwater input and sediments below the dam can have potentially serious impacts on both fish and invertebrate populations (NEFMC 1998).

Impoundments also create an opportunity for nonnative species to become established. Common carp (*Cyprinus carpio*), northern pike (*Esox lucius*), and walleye (*Sander vitreus*) are a few examples. These species have the ability to dramatically alter local habitats and aquatic communities. In some instances, introduced species such as smallmouth bass (*Micropterus dolomieu*) become managed as a sport fish to the exclusion of native species. Over time, these introduced species become accepted as part of the "natural" condition. Like the changes associated with creating an impoundment, these introduced species can change the community dynamics of the riverine system.

Conservation measures and best management practices for dam construction and operation (adapted from Hanson et al. 2003 and PFMC 1999)

1. Avoid the construction of new dam facilities, where possible.
2. Retrofit existing dams with efficient and functional upstream and downstream fish passage structures.
3. Construct and design facilities with efficient and functional upstream and downstream adult and juvenile fish passage which ensures safe, effective, and timely passage.
4. Construct dam facilities with the lowest hydraulic head practicable for the project purpose. Site the project at a location where dam height can be reduced.
5. Consider all upstream passage types, including natural-like bypass channels, denil-type and vertical slot fishways, Alaskan steeppass, fishlifts, etc. Volitional passage is preferable to trap and truck methods.
6. Downstream passage should prevent adults and juveniles from passing through the turbines and provide sufficient water downstream for safe passage.
7. Operate facilities to create flow conditions that provide for passage, water quality, proper timing of life history stages, and properly functioning channel conditions, and to avoid strandings and redd (i.e., spawning nest) dewatering. Run-of-river, such that the volume of water entering an impoundment exits the impoundment with minimal fluctuation of the headpond, is the preferred mode of operation for fishery and aquatic resource interests. Water flow monitoring equipment should be installed upstream and downstream of the facility. Generally, fluctuations in headpond water levels should be kept between 6 and 12 inches.
8. Coordinate maintenance and operations which require drawdown of the impoundment with state and federal resource agencies to minimize impacts to aquatic resources.
9. Use seasonal restrictions for construction, maintenance, and operations of dams 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. Develop water and energy conservation guidelines for integration into dam operation plans and into regional and watershed-based water resource plans.
11. Encourage the preservation of LWD, whenever possible. If possible, relocate debris as opposed to removing it completely. Remove LWD only to prevent damage to property or threats to human health and safety.
12. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in the review process for dam construction and operation.
13. Consider the removal of a dam when it is feasible (see the following section on dam removal).

Dam Removal

A number of factors may be considered in determining the efficacy of removing a dam, including habitat restoration, safety, and economics (Babbitt 2002; Heinz Center 2002). Dam removal provides overall environmental benefits to freshwater habitats and aquatic resources. The recovery of some anadromous species, such as Atlantic salmon and rainbow smelt, may be dependent on targeted dam removals, principally those dams blocking passage to high quality spawning and rearing habitat. Dam removal reconnects previously fragmented habitat, allowing the natural flow of water, sediment, nutrients, and the genetic diversity of fish populations and reestablishes floodplains and riparian corridors (Morita and Yokota 2002; Nislow et al. 2002).

The Heinz Center (2002) provides a thorough overview of environmental, economic, and social issues to consider when evaluating dam removal. Because there are a number of concerns and interests surrounding dams and their use, the overall benefits of dam removal must be weighed against all potential adverse impacts. It is important to bear in mind that although the removal of a dam may reverse most of the undesirable changes, it is unlikely to restore completely the natural conditions because of other dams on the river and the other anthropogenic effects on streams, such as channel control and land use management (Heinz Center 2002).

For many local residents, the impoundments created by these dams define a way of life for the community. Changing the existing conditions may not necessarily be perceived as good for all parties. For example, an impoundment may contain stocked game fish which provide recreational opportunities for the community. Dam removal may eliminate these species or bring about interactions with formerly excluded diadromous species. However, because dams alter sediment and nutrient transport processes and raise water levels upstream of the structure, dam removal can result in short and long-term impacts upstream and downstream.

The effects of dam removal on fisheries and aquatic habitat include: (1) release of contaminants; (2) short-term water quality degradation; (3) flow pattern alteration; (4) loss of benthic and sessile invertebrates; and (5) alterations of the riparian landscape and associated functions and values.

Release of contaminated sediments and short-term water quality degradation

Dam removal typically results in an increased transfer of sediments downstream of the dam, while the spatial and temporal extent of sediment transfer depends on the size of the dam and total sediment load. Sediments accumulated behind dams can bind and adsorb contaminants that when remobilized after the removal of a dam have the potential to adversely affect aquatic organisms including the eggs, larvae, and juvenile stages of finfish, filter feeders, and other sedentary aquatic organisms (Heinz Center 2002). For example, a reduction in macroinvertebrate abundance, diatom richness, and algal biomass has been attributed to the downstream transport of fine sediments previously stored within a dam impoundment (Thomson et al. 2005). However, as fine sediment loads are reduced and replaced by coarser materials in the streambed, macroinvertebrate and finfish assemblages should recover from the disturbance (Thomson et al. 2005). Dam removal can impact overall water quality during and after the demolition phase, although these are typically temporary effects that generally do not result in chronic water quality degradation (Nechvatal and Granata 2004; Thomson et al. 2005).

Flow pattern alteration

Dam removal generally changes downstream conditions by increasing the water and sediment discharges which tend to decrease channel gradients and increase stream depths and widths (Heinz Center 2002). In addition, flood events may increase; reactivate the floodplain; and reconnect side channels, islands, bars, and beaches. Reconnecting and increasing the active floodplain may help reduce low flow conditions in a river. Removal of a dam restores the natural timing of peak and low flows, which have important consequences for the biological components of the ecosystem. For example, seed production among native trees and spawning migrations of anadromous fish species often coincides with peak flows in the spring (Heinz Center 2002).

Loss of benthic and sessile invertebrates

As discussed above, remobilized sediments after the removal of a dam have the potential to adversely affect aquatic organisms including benthic and sessile invertebrates. However, although water quality often is degraded immediately following removal, the abundance and diversity of aquatic invertebrates should increase as the sediment budget and hydrology of the river approaches a natural equilibrium (Heinz Center 2002).

Alteration of wetlands

Lowering the water level will alter the wetland structure upstream of the old dam site and the associated wildlife assemblage. Lowering of impoundments can result in the alteration of existing wetlands (Nislow et al. 2002). As water levels recede, fringing wetlands may be lost while new wetlands are formed along the new riparian border. Newly exposed stream banks may need armoring or other erosion control methods to protect them. The history of the project, geomorphology of the watershed, and location in the river system, among other factors, will dictate the types of environmental issues dam removal will present. Geomorphic effects of downstream sediment transport may have long-term implications (Pizzuto 2002). However, many of these impacts are short-term, dissipating with time as the river system comes to a natural equilibrium (Bushaw-Newton et al. 2002; Thomson et al. 2005).

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

1. Conduct a comprehensive evaluation of the historic and existing hydrology, hydraulics, and sediment transport prior to the decision to remove a dam to assess possible adverse and cumulative effects of the removal of the structure on the watershed. Dam removal assessments should adopt a watershed scale of analysis.
2. Conduct an assessment of the biotic component of the effected area, particularly if anadromous fish restoration is one of the objectives of the dam removal. For example, the assessment may include characterization of the historic distribution and abundance of fish species, their various life history habitat requirements, and their limiting environmental factors. The assessment should also evaluate the predicted physical and chemical conditions following dam removal to determine if additional restoration may be necessary.
3. Conduct sufficient testing to evaluate the type, extent, and level of contamination upstream of the dam prior to the decision to remove a dam. Contaminated sediments, if extensively present, may require mechanical or hydraulic removal prior to the removal of the dam.
4. Conduct sufficient evaluation of the streambed within the impoundment to plan for any necessary streambed modifications.
5. Consider the possible necessity for removal of the dam in stages to control the release of sediments, if sediments are expected to be released downstream.
6. Schedule dam removal during the less sensitive time of year for aquatic resources, particularly outside the expected migratory period. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
7. Plan for revegetating the newly exposed stream bank with native vegetation.
8. Establish a contingency plan in the event that the stream channel needs modification (addition of riffle and pool complex, added features to create habitat complexity, meanders, etc.) to facilitate fish passage and habitat functions.
9. Establish a monitoring protocol to evaluate success of the restoration for fish passage and utilization.
10. Conduct outreach to the public to provide an understanding of the benefits of dam removal.

Stream Crossings

Stream crossings are characterized as any structure providing access over a stream, river, or other water body for transportation purposes (e.g., roads, utilities). The feasibility of effective fish passage at stream crossings may be complex. Land ownership, utility crossing, flood protection for low-lying properties, and safety along the transportation corridor must be considered. Unfortunately, many transportation corridors interact and interfere with fisheries corridors (i.e., streams and rivers). These transportation corridors require structures for crossing rivers, streams, and other water bodies. If improperly designed, stream crossings can alter, degrade, fragment or eliminate aquatic habitat and potentially impede, or eliminate, passage for resident and migratory species (Evans and Johnston 1980; Belford and Gould 1989; Clancy and Reichmuth 1990; Furniss et al. 1991; USGAO 2001; Jackson 2003). Until recently, the primary concerns related to designing these structures were cost, designed load capacity, and hydraulics. Furthermore, common practice for repairing deficient structures often resulted in maintaining inadequate stream crossing conditions (e.g., "slip-lining" with smaller diameter pipe, lining of culvert with concrete, or replacing the structure in-kind).

Some American states and Canadian provinces have recognized the concerns relating to fish passage and stream crossings. For example, the Maine Department of Transportation and Commonwealth of Massachusetts Riverways Program, among others, have independently published guidelines for addressing fish passage at stream crossings (MEDOT 2004; MRP 2005). These and similar documents provide extensive information regarding fish and aquatic organism passage, habitat continuity, and wildlife passage requirements for environmentally-sound and safe transportation across streams, rivers, and other waterbodies.

The construction, maintainance, and operation of roadways at stream crossings can also affect aquatic habitats by increasing rates of erosion, debris slides or landslides and sedimentation, introduction of exotic species, and degradation of water quality (Furniss et al. 1991; Hanson et al. 2003). However, the focus of this chapter is the design and operation of the fish passage structure. Refer to the Coastal Development chapter in this report for information pertaining to impacts associated with roadways and vehicular traffic at stream crossings.

Impacts to fish passage

Improperly designed stream crossings can block fish and aquatic organism passage in a variety of ways, including: (1) perched culverts constructed with the bottom of the structure above the level of the stream effectively act as a dam and physically block 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. Conversely, oversized culverts with large, flat bottom surfaces reduce water depth. Insufficient water depths may also be another hydraulic impediment to passage (Haro et al. 2004). In situations where water velocities are not physically limiting and water depths are sufficient, the impediments to passage may be a lack of resting pools. Many stream crossings, particularly longer culverts, are placed over wide stretches of river. Fish may not be capable of burst speeds and sustained swimming throughout the length of the crossing. Under such conditions, migrating fish are unable to reach spawning habitat.

Alteration of hydrologic regimes

Undersized and/or improperly placed stream crossings can also affect water quality. Undersized structures can act as dams, impounding water and increasing water temperature. In extreme cases, if flows are sufficiently reduced and the impounded area deep enough, increased surface temperatures can create thermal stratification and reduce dissolved oxygen. In addition, as water flows through the structure the temperature of the water can rise, affecting aquatic organisms downstream. Undersized culverts can also cause flooding upstream of the crossing, affecting upland and riparian habitat.

Conservation measures and best management practices for stream crossings

1. Design stream crossings for the target finfish species and various age classes. Other aquatic species, such as amphibians, reptiles, and mammals, should also be considered in the designs, as they play a role in healthy ecosystems.
2. Design structures to provide safe and timely passage to minimize injury and limit excessive predation.
3. Design and install new structures in a manner not to interfere with fish and aquatic organism passage and that complies with all applicable regulations.
4. Design structures to provide sufficient water depth and maintain suitable water velocities for target species during the migration season. Consider seasonal headwater and tailwater levels and how variations in them could affect passage of all aquatic life stages. Design considerations may include constructing a low flow channel, weir structure, energy dissipation pools, and designing structures for bank full width.
5. Consider the presence of nonnative, invasive aquatic species in fish passage design for stream crossings, particularly where the crossing may present an existing barrier to passage.
6. Design the structure to maintain or replicate natural stream channel and flow conditions to the greatest extent practicable. An open bottom arch or bridge is preferred. The structure should be able to pass peak flows in accordance with state and federal regulations. Ensure sufficient hydrologic data have been collected.
7. Bury culverts and pipes sufficiently to replicate a natural streambed. Doing so will also provide habitat functions, such as resting pools and reduced water velocities for longer structures.
8. Match the gradient of the stream crossing with the natural stream channel grade. Perched culverts should be removed, wherever practicable.
9. Maintain or stabilize upstream and downstream channel and bank conditions if the stream crossing structure causes erosion or accretion problems. Use of native vegetation should be required for erosion control and sediment stabilization.
10. Ensure the location and overall design of the fish passage structure and the stream crossing are compatible with local stream conditions and stream geomorphology.
11. Ensure that materials for the fish passage structure are nontoxic to fish and other aquatic organisms. Pressure treated lumber should be avoided.
12. Develop construction design and methods for repairing and replacing stream crossings that take into account fish passage requirements.
13. Conduct in-water construction activities during a time of year that would have the least environmental impacts to aquatic species (e.g., low flow seasons). Temporary diversions and coffer dams may be suitable alternatives with proper planning. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.

14. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in the review process for stream crossing projects.

Water Withdrawal and Diversion

Freshwater is becoming limited because of natural events (e.g., droughts), increasing commercial and residential demand of potable water, and inefficient use. Freshwater is diverted for human use from groundwater, lakes, and riverine environments or is stored in impoundments. The withdrawal or impoundment of water can alter natural current and sedimentation patterns, water quality, water temperature, and associated biotic communities (NEFMC 1998). Natural freshwater flows are subject to alteration through water diversion and use and modifications to the watershed such as deforestation, dams, tidal restrictions, and stream channelization (Boesch et al. 1997). Water withdrawal for freshwater drinking supply, power plant cooling systems, and irrigation occurs along urban and agricultural areas and may have potentially detrimental effects on aquatic habitats. Increased water diversion is associated with human population growth and development (Gregory and Bisson 1997). Water diversion is not only associated with water withdrawal and impoundment, but it also represents water discharges, which alter the flow and velocity and have associated water quality issues (Hanson et al. 2003). Water withdrawal in freshwater systems can also affect the health of estuarine systems. Refer to the Physical Effects: Water Intake and Discharge Facilities and Coastal Development chapters for additional information on the affects of water withdrawal on estuarine systems.

The effects of water withdrawal and diversion on freshwater fishery habitat can include: (1) entrainment and impingement; (2) impaired fish passage; (3) alteration of flow and flow rates, and processes associated with proper flows; (4) degradation of water quality (e.g., water temperature, dissolved oxygen) associated with proper water depth, drainage, and sedimentation patterns; (5) loss and/or degradation of riparian habitat; and (6) loss of prey and forage.

Entrainment and impingement

The diversion of water for power plant cooling and other reservoirs results in entrainment and impingement of invertebrates and fishes (especially early life-history stages of fish) (NEFMC 1998). Fish and invertebrate populations may be adversely affected by adding this source of mortality to the early life stage which often determines recruitment and strength of the year-class. Important habitat for aquatic organisms around water intakes may become unavailable for recruitment and settlement (Travnichek et al. 1993).

Impaired fish passage and altered hydrologic regimes

Water diversion and the withdrawal or discharge of water can result in a physical barrier to fish passage (Spence et al. 1996). Excessive water withdrawal can greatly reduce the usable river channel. Rapid reductions or increases in water flow, associated with dam operations for example, can greatly affect fish migratory patterns. Depending on the timing of reduced flows, fish can become stranded within the stream channel, in pools, or just below the river in an estuary system.

Water quality degradation

The release of water with poor quality (e.g., altered temperatures, low dissolved oxygen, and the presence of toxins) affects migration and migrating behavior. The discharge of irrigation water into a freshwater system can degrade aquatic habitat (NRC 1996) by altering currents, water quality, water temperature, depth, and drainage and sedimentation patterns. Both water quantity and quality can greatly affect the usable zone of passage within a channel (Haro et al. 2004). 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 2003). 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 extirpation of the species in an area. Some evidence indicates that elevated water temperatures in freshwater streams and rivers in the northeastern United States may be responsible for increased algal growth, which has been suggested as a possible factor in the diminished stocks of rainbow smelt (Moring 2005).

Release of contaminants

Irrigation discharges are often associated with contaminants and toxic materials (e.g., metals, pesticides, fertilizers, salts, and nutrients) and possibly introduced pathogens, all of which stress the habitat and aquatic organisms (USEPA 2003). Studies evaluating pesticides in runoff and streams generally find that concentrations can be relatively high near the application site and soon after application but are significantly reduced further downstream and with time (USEPA 2003). However, some pesticides used in the past (e.g., dichlorodiphenyl trichloroethane [DDT]) are known to persist in the environment for years after application.

Soil transported from irrigated croplands and rangelands usually contains a higher percentage of fine and less dense particles, which tend to have a higher affinity for adsorbing pollutants such as insecticides and herbicides (Duda 1985; USEPA 2003). In addition, irrigation water has a natural base load of dissolved mineral salts, and return flows convey the salt to the receiving streams or groundwater reservoirs. If the amount of salt in the return flow is low in comparison to the total stream flow, water quality may not be degraded to the extent that aquatic functions are impaired. However, if the process of water diversion and the return flow of saline drainage water is repeated many times along a stream or river, downstream habitat quality can become progressively degraded (USEPA 2003).

Siltation and sedimentation

Water diversions can alter sediment and nutrient transport processes (Christie et al. 1993; Fajen and Layzer 1993), which can hinder benthic processes and communities. Suspended sediments in aquatic environments can reduce the availability of sunlight to aquatic plants, interfere with filtering capacity of filter feeders, and clog and harm the gills of fish (USEPA 2003). Increased suspended sediments may degrade or eliminate spawning and rearing habitats, impede feeding, negatively affect the food sources of fishes, severely alter the aquatic food web, and thus negatively affect the growth and survival of diadromous fish. Fine sediments are potentially detrimental to Atlantic salmon development and survival during all life stages. For example, sediments can fill interstitial spaces, embedding the substrate and preventing oxygenated water from reaching the incubating eggs within redds and inhibiting the removal of waste metabolites; eliminate refuge utilized by fry and parr to avoid predators; create a homogeneous environment which can lead to lower fish densities; reduce macroinvertebrate abundance; and decrease the depth and area of pools utilized by juveniles and adults (Danie et al. 1984; Fay et al. 2006). In addition, Breitburg (1988) found the predation rates of striped bass larvae on copepods to decrease by 40% when exposed to high turbidity conditions in the laboratory.

Loss of wetlands and flood storage

Healthy riparian corridors are well vegetated, support abundant prey items, maintain nutrient fluxes, provide LWD that creates channel structure and cover for fish, and provide shade, which controls stream temperatures (Bilby and Ward 1991; Hanson et al. 2003). Riparian wetland vegetation can be affected by long-term or frequent changes in water levels caused by water withdrawals and diversions. Removal of riparian vegetation can impact fish habitat by reducing cover and shade, by reducing water temperature fluctuations, and by affecting the overall stability of water quality characteristics (Christie et al. 1993). As river and stream water levels recede because of withdrawals, fringing wetlands may be lost and armoring or other erosion control methods may be needed to protect newly exposed stream banks. The results are less refuge for fish, fundamental changes in channel structure (e.g., loss of pool habitats), instability of stream banks, and alteration of nutrient and prey sources within the river system (Hanson et al. 2003). The changes to the natural habitat caused by irrigation water discharges can potentially lead to large-scale aquatic community changes. Changes in flow patterns may affect the availability of prey and forage species. In conjunction with anthropogenic watershed changes, water diversions and associated riparian impacts have been associated with the increase in some harmful algal blooms (Boesch et al. 1997), which further impact an array of aquatic habitat characteristics. Lost wetlands correlate to a loss of floodplain and flood storage capacity, and thus a reduced ability to act as flood control during storm events.

For additional information on water diversion impacts, refer to the Physical Affects: Water Intake and Discharge Facilities, Chemical Affects: Water Discharge Facilities, and Agriculture and Silviculture chapters in this report.

Conservation measures and best management practices for water withdrawal/ diversion (adapted from Hanson et al. 2003)

1. Design projects to create flow conditions adequate to provide for passage, water quality, proper timing for all life history stages, and avoidance of juvenile stranding and redd (i.e., spawning nest) dewatering, as well as to maintain and restore properly functioning channel, floodplain, riparian, and estuarine conditions.
2. 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.
3. Establish adequate instream flow conditions for anadromous fish.
4. Design intakes with minimal flows to prevent impingement/entrainment (e.g., ≤0.5 feet per second).
5. Screen water diversions on fish-bearing streams, as needed.
6. Design thermal discharges such that ambient stream temperatures are maintained or a zone of passage is provided to maintain suitable temperatures for fish passage.
7. Incorporate juvenile and adult fish passage facilities on all water diversion projects.
8. Whenever possible, contaminants and sediments should be removed from water discharge prior to entering rivers and other aquatic habitats.
9. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in water withdrawal project review processes.

Dredging and Filling

The dredging and filling of riparian and freshwater wetlands directly remove potentially important habitat and alter the habitat surrounding the developed area. Expansion of navigable waterways is associated with economic growth and development and generally adversely affects benthic and water-column habitats. Routine dredging is required to maintain the desirable depth as the created channel fills with sediment. Direct removal of riverine habitat from dredge and fill activities may be one of the biggest threats to riverine habitats and anadromous species (NEFMC 1998).

Dredge and fill activities in riverine and riparian habitats can affect fisheries habitat in a number of ways, including: (1) reducing the ability of the wetland to retain floodwater; (2) reducing the uptake and release of nutrients; (3) decreasing the amount of detrital food source, an important food source for aquatic invertebrates (Mitsch and Gosselink 1993); (4) converting habitats by altering water depth or the substrate type (i.e., substrate conversion); (5) removing aquatic vegetation and preventing natural revegetation; (6) hindering physiological processes to aquatic organisms (e.g., photosynthesis, respiration) caused by increased turbidity and sedimentation (Arruda et al. 1983; Cloern 1987; Dennison 1987; Barr 1993; Benfield and Minello 1996; Nightingale and Simenstad 2001); (7) directly eliminating sessile or semimobile aquatic organisms via entrainment or smothering (Larson and Moehl 1990; McGraw and Armstrong 1990; Barr 1993; Newall et al. 1998); (8) altering water quality parameters (i.e., temperature, oxygen concentration, and turbidity); (9) releasing contaminants such as petroleum products, metals, and nutrients (USEPA 2000); (10) reducing dissolved oxygen through reduced photosynthesis and through chemical processes associated with the release of reactive compounds in the sediment (Nightingale and Simenstad 2001).

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) because of 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 (Mitsch and Gosselink 1993).

Flood storage capacity

Impervious surfaces decrease the capacity of a watershed to absorb pulses of freshwater input (e.g., heavy rain, snowmelt). Similarly, stormwater drain systems decrease the storage by directing water directly into a nearby wetland or river system. The rate and volume of stormwater runoff from land into rivers and streams is greater in watersheds with high percentages of impervious surface cover and extensive drainage systems, which reduce the stormwater storage capacity (American Rivers 2002). Measurable adverse changes in the physical and chemical environment were observed when the impervious cover exceeded 10-20% of the land cover (Holland et al. 2004). Flashy, high-velocity pattern of flows and associated pulse of contaminants from upland sources can have long-term, cumulative impacts on freshwater wetlands and riverine, estuarine, and marine ecosystems. As development continues throughout the region, the ability to minimize loss of flood storage capacity and mitigate consequences of increasing coverage of impervious surfaces will be significant planning issues (American Rivers 2002). Refer to the Coastal Development chapter for additional information on stormwater runoff and nonpoint source pollution.

Impacts associated with dredging and filling of aquatic habitats and wetlands are discussed in greater detail in the Offshore Dredging and Disposal Activities, Marine Transportation, and Coastal Development chapters of this report.

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

1. Avoid the filling of wetlands and riparian habitat whenever possible. Ensure proposed dredge and fill projects in wetlands are water-dependent.
2. Utilize best management practices (BMPs) to limit and control the amount and extent of turbidity and sedimentation. Standard BMPs may include constructing silt fences, coffer dams, and operational modification (e.g., hydraulic dredge rather than mechanical dredge).
3. Require the use of multiple-season biological sampling data (both pre- and post-construction) when appropriate to assess the potential and resultant impacts on habitat and aquatic organisms.
4. Test sediment compatibility for open-water disposal per the US Environmental Protection Agency (US EPA) and US Army Corps of Engineers requirements for inshore and offshore, unconfined disposal.
5. Plan dredging and filling activities to avoid submerged aquatic vegetation and special aquatic sites. This may include the placement of pipes for hydraulic dredging and anchoring of barges and other vessels associated with the dredging project.
6. Design the dredge footprint to avoid littoral zone habitat, and appropriate buffers should be in place to protect these areas from wind driven waves and boat wakes.
7. Schedule dredging activities when the fewest species and least vulnerable life stages are present. Appropriate work windows can be established based on the multiple season biological sampling. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
8. Reference all dredging projects in a geographical information system (GIS) compatible format for long-term evaluation.
9. Identify sources of sedimentation within the watershed that may exacerbate repetitious maintenance activities. Implement appropriate management techniques to control these sources.
10. Address cumulative impacts of past, present, and foreseeable future dredging operations on aquatic habitats by considering them in the review process.

Mining

Most modern mining operations in the northeast US region involve bulk mineral commodities (aggregates such as sand, gravel, and crushed stone), but the region has a long history of mineral mining for mica, feldspar, copper, iron, gold, silver, and coal, as well as peat (Lepage et al. 1991; Boudette 2005; VADMME 2007). While some mineral mining continues in this region, many operations have ceased entirely (Lepage 1991). Some of these abandoned mines have become a source of groundwater or surface water contamination and have been identified by the US EPA's Superfund Program (USEPA 2007) and other nonfederal programs for cleanup. Currently, the US EPA Superfund Program lists cleanup sites on the Susquehanna River in Pennsylvania from coal mining and tributaries leading to East Penobscot Bay in Maine and the Connecticut River in Vermont from copper and other metal mining.

Few active mining sites in the northeast US region currently affect fishery resources as they generally are not located adjacent to or in rivers that support didromous fish. In addition, because access for diadromous fish to historic spawning grounds has been adversely affected by dams and poor water quality throughout the region (Moring 2005), the potential adverse effects of mining operations on these species have been reduced in recent times. Nonetheless, some sand and gravel extraction projects occur within rivers and their tributaries of the northeast US region. Although limited information is available on this subject, it appears the number of active sand and gravel operations that may adversely affect diadromous fish in the northeast US region is relatively small compared to other regions of the United States. However, considering the potential direct and indirect effects from historic and current mining activities on long-term water quality and health of diadromous species, a brief discussion on this topic is warranted in this section.

Mining within riverine habitats may result in direct and indirect chemical, biological, and physical impacts to habitats within the mining site and surrounding areas during all stages of operations (NEFMC 1998). On-site mining activities include exploration, site preparation, mining and milling, waste management, decommissioning and reclamation, and abandonment. Mining operations often occur in urban settings or around existing or historic mining sites; however, mining in remote settings where human activity has caused little disruption and aquatic resources are most productive may cause significant impacts (NRC 1999). Existing state and federal regulations have been established to restrict various environmental impacts associated with mining operations. However, the nature of mining will always result in some alteration of habitat and natural resources (NRC 1999).

Some of the impacts associated with the extraction of material from within or near a stream or river bed include: (1) disruption of preexisting balance between sediment supply and transporting capacity, leading to channel incision and bed degradation; (2) increased suspended sediment, sediment transport, turbidity, and gravel siltation; (3) alteration in the morphology of the channel and decreased channel stability; (4) direct impacts to fish spawning and nesting habitats (redds), juveniles, and prey items; (5) alteration of the channel hydraulics during high flows caused by material stockpiled or left abandoned; (6) removal of instream roughness, including LWD; (7) reduced groundwater elevations and stream flows caused by dry pit or wet pit mining; and (8) destruction of the riparian zone during extraction operations (Pearce 1994; Packer et al. 2005). In addition, structures used in mining extraction and transportation often cause additional impacts to wetland and riverine habitats (Starnes and Gasper 1996). Other impacts include fragmentation and conversion of habitat, alteration of temperature regimes, reduction in oxygen concentration, and the release of toxic materials.

Mineral mining

Although there is a long history of mining in the northeast region of the United States, few active mineral mining operations remain that are located in or adjacent to streams or rivers in this region, and even fewer mineral mining operations occur in streams and rivers utilized by diadromous fish. Nonetheless, mineral mining has occurred in the northeast US region in the past, as evidenced by a number of completed and ongoing remediation sites in areas that have supported or historically supported diadromous fish (USEPA 2007). A brief discussion on the potential impacts to aquatic habitats is provided below.

The effects of mineral mining on riverine habitat depend on the type, extent, duration, and location of the mining activity. Surface mining typically involves suction dredging, hydraulic mining, panning, sluicing, strip mining, and open-pit mining. Surface mining has a greater potential impact on riverine habitat than does underground or shaft mining, depending on other aspects of the mining activities, including processing and degree of disturbance (Spence et al. 1996; Hanson et al. 2003). Elimination of vegetation, topographic alterations, alteration of soil and subsurface geological structure and alteration of surface and groundwater hydrologic regimes are potential effects of surface mining (Starnes and Gasper 1996). Soil erosion and sediment runoff may be the greatest impact of surface mining, contributing a greater sediment load per area of disturbance compared with other activities because of the degree of soil, topographic, and vegetation disturbance (Nelson et al. 1991).

Sand and gravel mining

Sand and gravel are the most valuable and extensively exploited nonfuel mineral resources in the eastern US region and are mined in all states from Virginia to Maine (Bolen 2007). According to Starnes and Gasper (1996), sand and gravel extraction is the least regulated of all mining industries, and approximately 80% of this resource is extracted under jurisdiction of state and local laws only. These authors state that sand and gravel mining is "widely used in large US rivers and can increase the sediment bed load through resuspension, physically eliminate benthic organisms, and destroy fish spawning and nursery areas, all of which ultimately change aquatic community composition" (Starnes and Gasper 1996); however, they do not identify specific rivers that are affected or state whether the rivers support diadromous fish species. The Virginia Department of Mines, Minerals and Energy states, "Sand and gravel are extracted from coastal sand pits, river terraces or dredged from the rivers themselves" (VADMME 2007). In 2005, over 15,000 tons of sand were mined from two operations along the Roanoke River in Virginia (VADMME 2007). In addition, a dredge and fill permit was granted by the US Army Corps of Engineers to allow sand extraction in the St. John River, ME, for use in road sanding operations (USACE 2005). Although sand and gravel mining may not be a significant threat to diadromous fish in the northeast US region at this time, at least some activity is currently taking place, and any increase in activity represents potential future threat.

Gravel and sand mining operations can involve wet-pit mining (i.e., removal of material below the water table); dry pit mining on beaches, exposed bars, and ephemeral streambeds; or subtidal mining. Impacts associated with sand and gravel mining in riverine environments are similar to mineral mining impacts and include: turbidity plumes and resuspension of sediment and nutrients, removal of spawning habitat, and alteration of stream channel morphology. These physical perturbations often lead to alteration of migration patterns, physical and thermal barriers to upstream and downstream migration, increased fluctuation in water temperature, decrease in dissolved oxygen, high mortality of early life stages, increased susceptibility to predation, and loss of suitable habitat (Packer et al. 2005). For information pertaining to impacts associated with mining and dredging in marine habitats refer to the chapter on Offshore Dredging and Disposal Activities.

Peat mining

Peat is mined in the United States primarily for horticultural and industrial purposes, including a filtration medium to remove toxic materials and a fuel/oil absorbent (Jasinski 2007). Peat mining occurs in a number of states in the northeast US region, although at relatively small scales. In Maine, at least one peat mining operation exists in the Narraguagus River watershed, which burns mixtures of peat and wood chips to generate electricity (Lepage et al. 1991; USFWS and NMFS 1999).

The impacts associated with peat mining include the release of contaminants (i.e., peat fiber, arsenic residues, and other toxic chemicals), siltation, increased stormwater runoff from roads and other unvegetated areas, and altered hydraulic flow regimes (NEFMC 1998; USFWS and NMFS 1999). Peat mining has been associated with acidic conditions in eastern Maine watersheds, such as Narraguagus River, and has been identified as a potential contributor to Atlantic salmon declines (USFWS and NMFS 1999).

Alteration of stream bed and stream morphology

Surface mining can alter channel morphology by making the stream channel wider and shallower and removing the natural sediment load. Consequently, the suitability of stream reaches as rearing habitat may decrease, especially during summer low-flow periods when deeper waters are important for survival. Gravel bar skimming or "scalping," which involves the removal of the surface from gravel bars without excavating below the low water flow level, can significantly impact aquatic habitat (Packer et al. 2005). Bar skimming creates a wide, flat cross section in the stream channel, which eliminates confinement of the low flow channel. A reduction in pool frequency may adversely affect migrating adults that require holding pools (Spence et al. 1996). Changes in the frequency and extent of bedload movement and increased erosion and turbidity can also remove spawning substrates, scour redds, result in a direct loss of eggs and young, or reduce their quality by deposition of increased amounts of fine sediments. These changes can affect the early life stages of Atlantic salmon, which exhibit an affinity for specific habitat types (Fitzsimons et al. 1999; Hedger et al. 2005). Extraction of sand and gravel in riverine ecosystems can directly eliminate the amount of gravel available for spawning if the extraction rate exceeds the deposition rate of new gravel in the system. Gravel excavation also reduces the supply of gravel to downstream habitats. The extent of suitable spawning habitat may be reduced where degradation reduces gravel depth or exposes bedrock (Spence et al. 1996). Associated with stream morphology alterations are resultant increased temperatures from a reduction in summer base flows; altered width to depth ratios; decreased riparian vegetation; decreased dissolved oxygen concentration as water temperatures increase; decreased nutrients from loss of floodplain connection and riparian vegetation; and decreased food production (e.g., loss of invertebrate prey populations) (Spence et al. 1996).

Sedimentation and siltation

Sedimentation effects of mining may be immediate or delayed. During gravel extraction, for example, fine material can travel long distances downstream in the form of turbidity plumes. Silt can also be released during peat mining operations (USFWS and NMFS 1999). Sedimentation may be a delayed effect because gravel removal typically occurs at low flow when the stream has the least capacity to transport fine sediments out of the system. Increased sedimentation results when the spring freshet inundates an extraction area that is less stable than before mining operations. The extent and duration of sedimentation and siltation is likely to be higher than normal as unstable sediment washes freely into the system during higher rates of flow, acting as a migratory barrier to anadromous fish, such as Atlantic salmon, and increasing entrainment of sediment in downstream habitat. The result can be a degradation or loss of spawning and rearing habitat within the system (Spence et al. 1996).

Release of contaminants

Peat mining can negatively impact diadromous fish, including Atlantic salmon, from the discharge of low pH water containing peat silt and dissolved metals and pesticides (USFWS and NMFS 1999). However, only one peat mining operation has been identified on the Narraguagus

River in Maine, and monitoring efforts at the site suggests that impacts are being controlled (USFWS and NMFS 1999).

Although current mineral mining operations in the northeast region of the United States are not a significant threat to rivers supporting diadromous fish, the effects of historic mining operations continue to be remediated (USEPA 2007). Harmful or toxic materials can be released directly from mining operations, including processing and machinery. Mining can introduce high levels of metals, sulfuric acid, mercury, cyanide, arsenic, and processing reagents into waterways. Water pollution by metals and acids is associated with mineral mining because ores, rich in sulfides, are commonly mined to extract gold, silver, copper, zinc, and lead (NRC 1999). In combination with anoxic conditions, sulfur-containing sediments can create additional levels of toxicity in addition to acid conditions (Brouwer and Murphy 1995). The improper handling or discharge of tailings and settling ponds can result in a direct loss of living aquatic resources as a result of decreased water quality and increased concentration levels of toxic substances. Locating settling ponds in unstable or landslide prone upland sites makes them prone to dangerous, instantaneous releases of large quantities of toxins. Groundwater and surface water may be incidentally contaminated by leaching of toxic substances from upland settling ponds.

Conservation measures and best management practices for mining (adapted from Hanson et al. 2003 and Packer et al. 2005)

1. Use upland aggregate sources before beginning any mining activities in active channels or floodplains.
2. Avoid mining operations in rivers and streams identified as important migratory pathways, spawning, and nursery habitat for anadromous fish.
3. Conduct a thorough assessment and characterization of aquatic resources, sediments, and potential sources of point and nonpoint contaminants prior to gravel removal.
4. Design, manage, and monitor sand and gravel mining operations to minimize potential direct and indirect impacts to riverine habitat if operations cannot be avoided. This includes, but is not limited to, migratory corridors, foraging and spawning areas, and stream/river banks.
5. Minimize the spatial extent and the depth of mine extraction operation to the maximum extent practicable.
6. Schedule necessary in-water activities when the fewest species and least vulnerable life stages are present. 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.
7. Identify upland or off-channel (where channel will not be captured) gravel extraction sites as alternatives to gravel mining in or adjacent to rivers and streams identified as important pathways for anadromous fish, if possible.
8. Utilize best management practices to avoid spills of dirt, fuel, oil, toxic materials, and other contaminants. Prepare a spill prevention plan and maintain appropriate spill containment and water repellent/oil absorbent cleanup materials on the project location.
9. Treat wastewater (e.g., acid neutralization, sulfide precipitation, reverse osmosis, electrochemical, or biological treatments) and recycle onsite to minimize discharge to streams. Treat wastewater before discharge for compliance with state and federal clean water standards.
10. Reclaim mining wastes that contain contaminants such as metal, acids, arsenic, or other substances if leachate could enter aquatic habitats through surface or groundwater.

11. Use best management practices to minimize opportunities for sediment to enter streams and waterways. Methods such as contouring, mulching, silt curtains, and settling ponds should be part of the operations plan. Monitor turbidity during operations and alter operations if turbidity levels reach or exceed a predetermined level.
12. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in mining project review processes.

Emerging Issues for Freshwater Systems

Endocrine disruptors, pharmaceuticals, and nanoparticles

Growing concerns have mounted in response to the effects of endocrine-disrupting chemicals on humans, fish, and wildlife (Kavlock et al. 1996; Kavlock and Ankley 1996). These chemicals act as "environmental hormones" that may mimic the function of the sex hormones androgen and estrogen (Thurberg and Gould 2005). One of the sources of endocrine disrupting compound is the effluent of residential and commercial wastewater treatment facilities, as well as agricultural runoff (USGS 2002). Some of the chemicals shown to be estrogenic include polychlorinated biphenyl (PCB), dieldrin, DDT, phthalates, and alkylphenols (Thurberg and Gould 2005), which have had or still have applications in agriculture and may be present in irrigation water. Metals have also been implicated in disrupting endocrine secretions of marine organisms, potentially disrupting natural biotic processes (Brodeur et al. 1997). Adverse effects include reduced or altered reproductive functions, which could result in population-level impacts. Refer to the Chemical Effects: Water Discharge Facilities chapter for more information on endocrine disruptors. In addition to endocrine disrupting compounds, recent studies have found municipal wastewater effluent entering streams and rivers containing human and veterinary pharmaceuticals, including antibiotics and natural and synethetic hormones (USGS 2002).

Other recent concerns are the release of substances referred to as nanoparticles into the aquatic environment. Nanoparticles, such as fullerenes (e.g., 60-carbon molecules often referred to as "buckyballs") may have great potential for use in the pharmaceutical, lubricant, and semiconductor industries, as well as applications in energy conversion. However, the micro-fine particulate waste generated from the production and use of nanoparticles may adversely affect the distribution, feeding, ecology, respiration, and nutrient regeneration of microorganisms, such as bacterivorous and herbivorous protozoa, protists, and phagotrophic or mixotrophic microalgae (Colvin 2003).

Harmful algal blooms

Impervious surfaces and stormwater drain systems can increase the rate and volume of stormwater runoff into rivers and streams. This direct flushing of water generates large pulses of runoff into rivers and streams, carrying with it nutrients and a wide-range of pollutants that flow into estuaries and coastal areas. Nutrient-rich waters have been associated with harmful algal blooms (HABs), which can deplete the oxygen in the water during bacterial degradation of algal tissue and can result in hypoxic or anoxic "dead zones" and large-scale fish kills in rivers, estuaries, and coastal areas (Deegan and Buchsbaum 2005; MDDNR 2007). For example, HABs have been responsible for fish kills in the freshwater portions of the Potomac River in Virginia and the Corsica River in Maryland, as well as in the Potomac and Chesapeake Bay estuaries (MDDNR 2007). HABs affecting Gulf of Maine waters have resulted in shellfish bed closures and mortalities to endangered marine mammals (NOAA 2008; WHOI 2008). While the causes of HABs in coastal waters of New England are unclear, large pulses of freshwater rivers and streams in the region as a result of elevated rainfall and snowmelt in the spring are being examined as contributing factors in creating conditions favorable for algal growth (NOAA 2008). Refer to the Coastal Development and Introduced/Nuisance Species and Aquaculture chapters for more information on HABs.

Introduced and nusiance species

Introductions of nonnative nuisance species are a significant threat to freshwater and coastal ecosystems in the United States (Carlton 2001). Nonnative species may be released intentionally (i.e., fish stocking and pest control programs) or unintentionally during industrial shipping activities (e.g., ballast water releases), aquaculture operations, recreational boating, biotechnology, or from aquarium discharge (Hanson et al. 2003; Niimi 2004). For example, increased competition for food sources between the invasive exotic zebra mussel (*Dreissena polymorpha*) and open-water commercial and recreational species have altered the trophic structure in the Hudson River estuary, NY, by withdrawing large quantities of phytoplankton and zooplankton from the water column, thus increasing competition with planktivorous fish (Strayer et al. 2004). Refer to the Introduced/Nuisance Species and Aquaculture chapter for information on introduced and nuisance species.

References for Alteration of Freshwater Systems

Abbe TB, Montgomery DR. 1996. Large woody debris jams, channel hydraulics and habitat formation in large rivers. Regulated Rivers-Research and Management 12(2-3):201-21.

American Rivers, Natural Resources Defense Council, Smart Growth America. 2002. Paving our way to water shortages: how sprawl aggravates the effects of drought. [Internet]. Washington (DC): Smart Growth America. [cited 2007 Dec 21]. 31 p. Available from: http://www.smartgrowthamerica.org/waterandsprawl.html.

Arruda JA, Marzolf GR, Faulk RT. 1983. The role of suspended sediments in the nutrition of zooplankton in turbid reservoirs. Ecology 64(5):1225-35.

Babbitt B. 2002. What goes up, may come down: learning from our experiences with dam construction in the past can guide and improve dam removal in the future. BioScience 52(8):656-8.

Barr BW. 1993. Environmental impacts of small boat navigation: vessel/sediment interactions and management implications. In: Magoon OT, editor. Coastal Zone '93: proceedings of the eighth Symposium on Coastal and Ocean Management; 1993 Jul 19-23; New Orleans, LA. American Shore and Beach Preservation Association. p 1756-70.

Belford DA, Gould WR. 1989. An evaluation of trout passage through six highway culverts in Montana. North American Journal of Fisheries Management 9(4):437-45.

Benfield MC, Minello TJ. 1996. Relative effects of turbidity and light intensity on reactive distance and feeding of an estuarine fish. Environmental Biology of Fishes 46:211-6.

Beschta RL, Bilby RE, Brown GW, Holtby LB, Hofstra TD. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. In: Salo EO, Cundy TW, editors. Streamside management: forestry and fishery interactions. Seattle (WA): University of Washington, College of Forest Resources. p 191-232.

Bilby RE, Ward JW. 1991. Characteristics and function of large woody debris in streams draining old-growth, clearcut, and second-growth forests in southwestern Washington. Canadian Journal of Fisheries and Aquatic Sciences 48:2499-508.

Bodi FL, Erdheim E. 1986. Swimming upstream: FERC's failure to protect anadromous fish. Ecology Law Quarterly 13(1):7-50.

Boesch DF, Anderson DM, Horner RA, Shumway SE, Tester PA, Whitledge TE. 1997. Harmful algal blooms in coastal waters: options for prevention, control and mitigation. Silver Spring (MD): NOAA Coastal Ocean Office. NOAA Coastal Ocean Program Decision Analysis Series No. 10. 46 p. + appendix.

Bolen W. 2007. Sand and Gravel, Construction. US Geological Survey Minerals Yearbook-2005. [Internet]. US Department of the Interior, US Geological Survey. [updated 2008 May 19; cited 2008 Jul 15]. 21 p. Available from:

http://minerals.usgs.gov/minerals/pubs/commodity/sand_&_gravel_construction/sandgmyb05.pdf.

Boudette EL. 2005. Gold in New Hampshire. [Internet]. Concord (NH): New Hampshire Department of Environmental Services. Environmental Fact Sheet CO-GEO-1. [cited 2008 Jul 15]. 4 p. Available from: http://www.des.state.nh.us/factsheets/geo/geo-1.htm.

Breitburg DL. 1988. Effects of turbidity on prey consumption by striped bass larvae. Transactions of the American Fisheries Society 117(1):72-7.

Brodeur JC, Sherwood G, Rasmussen JB, Hontela A. 1997. Impaired cortisol secretion in yellow perch (*Perca flavescens*) from lakes contaminated by heavy metals: *in vivo* and *in vitro* assessment. Canadian Journal of Fisheries and Aquatic Sciences 54(12):2752-8.

Brouwer H, Murphy T. 1995. Volatile sulfides and their toxicity in freshwater sediments. Environmental Toxicology and Chemistry 14 (2):203-8.

Buchsbaum R. 2005. The role of overfishing, pollution, and habitat degradation on marine fish and shellfish populations of New England: summary and conclusions. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. 175 p.

Bushaw-Newton KL, Hart DD, Pizzuto JE, Thomson JR, Egan J, Ashley JT, Johnson TE, Horwitz RJ, Keeley M, Lawrence J, and others. 2002. An integrative approach towards understanding ecological responses to dam removal: the Manatawny Creek study. Journal of the American Water Resources Association 38 (6):1581-99.

Carlton JT. 2001. Introduced species in U.S. coastal waters: Environmental impacts and management priorities. Arlington (VA): Pew Oceans Commission. 28 p.

Chabreck RA. 1988. Coastal marshes: ecology and wildlife management. Minneapolis (MN): University of Minnesota Press. 138 p.

Christie RW, Allen DE, Jobsis GJ. 1993. Diversions. In: Bryan CF, Rutherford DA, editors. Impacts on warmwater streams: guidelines for evaluation. Little Rock (AR): Southern Division, American Fisheries Society. p 181-186.

Clancy CG, Reichmuth DR. 1990. A detachable fishway for steep culverts. North American Journal of Fisheries Management 10(2):244-6.

Cloern JE. 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. Continental Shelf Research 7(11-12):1367-81.

Collins BD, Montgomery DR, Haas AD. 2002. Historical changes in the distribution and functions of large wood in Puget lowland rivers. Canadian Journal of Fisheries and Aquatic Sciences 59:66-76.

Collins BD, Montgomery DR, Sheikh AJ. 2003. Reconstructing the historic riverine landscape of the Puget lowland. In: Bolton S, Montgomery DR, Booth D, Wall L, editors. Restoration of Puget Sound rivers. Seattle (WA): University of Washington Press. p 79-128.

Colvin VL. 2003. The potential environmental impact of engineered nanomaterials. Nature Biotechnology 21:1166-70.

Dadswell MJ, Rulifson RA. 1994. Macrotidal estuaries: a region of collision between marine animals and tidal power development. Biological Journal of the Linnean Society 51(1-2):93-113.

Danie DS, Trial JG, Stanley JG. 1984. Species profiles: life histories and environmental requirements of coastal fish and invertebrates (North Atlantic)-Atlantic salmon. Washington (DC): US Fish and Wildlife Service, FWS/OBS-82/11.22 and the US Army Corps of Engineers, TR EL-82-4. 19 p.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Dennison WC. 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. Aquatic Botany 27(1):15-26.

De Laney TA. 1995. Benefits to downstream flood attenuation and water quality as a result of constructed wetlands in agricultural landscapes. Journal of Soil and Water Conservation 50:620-6.

Duda AM. 1985. Environmental and economic damage caused by sediment from agricultural nonpoint sources. Water Resources Bulletin 21(2):225-34.

Evans WA, Johnston B. 1980. Fish migration and fish passage: a practical guide to solving fish passage problems. Washington (DC): US Department of Agriculture, Forest Service. 163 p.

Fajen OF, Layzer JB. 1993. Agricultural practices. In: Bryan CF, Rutherford DA, editors. Impacts on warmwater streams: guidelines for evaluation. Little Rock (AR): Southern Division, American Fisheries Society. p 257-67.

Fay C, Bartron M, Craig S, Hecht A, Pruden J, Saunders R, Sheehan T, Trial J. 2006. Status review for anadromous Atlantic salmon (*Salmo salar*) in the United States. [Internet]. Report to the National Marine Fisheries Service and US Fish and Wildlife Service. [cited 2008 Jul 15]. 294 p. Available from: http://www.nmfs.noaa.gov/pr/pdfs/statusreviews/atlanticsalmon.pdf.

Fitzsimons JD, Fodor G, Williston B, Carl L, Gentile T. 1999. Spawning habitat selection and suitability for Atlantic salmon. In: IAGLR '99. The 42nd Conference of the International Association for Great Lakes Research: Great Lakes, Great Science, Great Cities; 1999 May 24-28; Cleveland (OH): Case Western Reserve University. p A32-3.

Fletcher R. 2003. Loss of wetlands: how are bird communities affected? [Internet]. American Institute of Biological Sciences, c2000-2007. [cited 2007 Dec 21]. Available from: http://www.actionbioscience.org/environment/fletcher.html.

Furniss MJ, Roelofs TD, Yee CS. 1991. Road construction and maintenance. In: Meehan WR, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. Bethesda (MD): American Fisheries Society. Special Publication 19. p 297-323.

Graf WL. 1993. Landscapes, commodities, and ecosystems: the relationship between policy and science for American rivers. In: Sustaining our water resources (1993) Commission on geosciences, environment and resources (CGER). Washington (DC): National Academy Press. p 11-42.

Gregory SV, Bisson PA. 1997. Degradation and loss of anadromous salmonid habitat in the Pacific Northwest. In: Stouder DJ, Bisson PA, Naiman RJ, editors. Pacific Salmon and their ecosystems: status and future options. New York (NY): Chapman and Hall. p 277-314.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Haro A, Castro-Santos T, Noreika J, Odeh M. 2004. Swimming performance of upstream migrant fishes in open-channel flow: a new approach to predicting passage through velocity barriers. Canadian Journal of Fisheries and Aquatic Sciences 61(9):1590-601.

Hedger RD, Dodson JJ, Bergeron NE, Caron F. 2005. Habitat selection by juvenile Atlantic salmon: the interaction between physical habitat and abundance. Journal of Fish Biology 67:1054-71.

Heinz Center. 2002. Dam removal: science and decision making. Washington (DC): The H. John Heinz III Center for Science, Economics, and the Environment. 236 p.

Holland AF, Sanger DM, Gawle CP, Lerberg SB, Santiago MS, Riekerk GHM, Zimmerman LE, Scott GI. 2004. Linkages between tidal creek ecosystems and the landscape and demographic attributes of their watersheds. Journal of Experimental Marine Biology and Ecology 298(2):151-78.

Howard-Williams C. 1985. Cycling and retention of nitrogen and phosphorus in wetlands: theoretical and applied perspective. Freshwater Biology 15(4):391-431.

Jackson SD. 2003. Design and construction of aquatic organism passage at road-stream crossings: ecological considerations in the design of river and stream crossings. In: Irwin CL, Garrett P, McDermott KP, editors. 2003 Proceedings of the International Conference on Ecology and Transportation; 2003 Aug 24-29; Lake Placid, NY. Raleigh (NC): Center for Transportation and the Environment: North Carolina State University. p 20-9.

Jasinski SM. 2007. USGS Minerals Yearbook: peat. [Internet]. US Department of the Interior, US Geological Survey. [cited 2008 Jul 15]. 6 p. Available from: http://minerals.usgs.gov/minerals/pubs/commodity/peat/myb1-2006-peat.pdf.

Kavlock RJ, Ankley GT. 1996. A perspective on the risk assessment process for endocrine-disruptive effects on wildlife and human health. Risk Analysis 16(6):731-9.

Kavlock RJ, Daston GP, DeRosa C, Fenner-Crisp P, Gray LE, Kaattari S, Lucier G, Luster M, Mac MJ, Maczka C and others. 1996. Research needs for the risk assessment of health and environmental effects of endocrine disruptors: a report of the US EPA-sponsored workshop. Environmental Health Perspectives 104 (Suppl 4):715-40.

Kerry J, Boorse D, Buchsbaum R. 2004. Effect of nutrient enrichment and salinity on salt marsh invertebrates in the Plum Island Estuary. Biological Bulletin 207:174.

Larinier M. 2001. Environmental issues, dams and fish migration. In: Dams, fish and fisheries: opportunities, challenges and conflict resolution. Rome (Italy): FAO Fisheries Technical Paper No. 419. p 45-89.

Larson K, Moehl CE. 1990. Entrainment of anadromous fish by hopper dredge at the mouth of the Columbia River. In: Simenstad CA Jr, editor. Effects of dredging on anadromous Pacific coast fishes. Seattle (WA): Washington Sea Grant, University of Washington. p 102-12.

Lepage CA, Foley ME, Thompson WB. 1991. Mining in Maine: past, present, and future. [Internet]. Augusta (ME): Maine Geological Survey. Open-File 91-7. [updated 2005 Oct 6; cited 2007 Dec 21]. 11 p. Available from: http://www.maine.gov/doc/nrimc/mgs/explore/mining/minemaine.htm.

[MEDOT] Maine Department of Transportation. 2004. Fish passage policy and design guide. 2nd ed. [Internet]. Augusta (ME): Maine Department of Transportation. 80 p. [updated 2005 Jan 3; cited 2007 Dec 21]. Available from: http://www.maine.gov/mdot-stage/environmental-office-homepage/documents.

[MDDNR] Maryland Department of Natural Resources. 2007. Harmful algae blooms in Maryland. [Internet]. [cited 2007 Dec 21]. Five articles available from: http://www.dnr.state.md.us/bay/hab/index.html.

Maser C, Sedell JR. 1994. From the forest to the sea: the ecology of wood in streams, rivers, estuaries, and oceans. Delray Beach (FL): St. Lucie Press. 200 p.

[MRP] Massachusetts Riverways Program. 2005. Massachusetts stream crossings handbook. [Internet]. Boston (MA): Commonwealth of Massachusetts, Executive Office of Environmental Affairs, Department of Fish and Game. [cited 2007 Dec 21]. Available from: http://www.mass.gov/dfwele/river/programs/rivercontinuity/guidancedoc.htm.

McGraw K, Armstrong D. 1990. Fish entrainment by dredges in Grays Harbor, Washington. In: Simenstad CA Jr, editor. Effects of dredging on anadromous Pacific coast fishes. Seattle (WA): University of Washington Sea Grant. p 113-31.

Mitsch WJ, Gosselink JG. 1993. Wetlands. 2nd ed. New York (NY): Van Nostrand Reinhold. 722 p.

Montgomery DR, Smith RD, Schmidt KM, Pess GR. 1995. Pool spacing in forest channels. Water

Resources Research 31(4):1097-105.

Moring J. 2005. Recent trends in anadromous fishes. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 25-42.

Morita K, Yamomoto S. 2002. Effects of habitat fragmentation by damming on the persistence of stream-dwelling charr populations. Conservation Biology 16(5):1318-23.

Morita K, Yokota A. 2002. Population viability of stream-resident salmonids after habitat fragmentation: a case study with white-spotted charr (*Salvelinus leucomaenis*) by an individual based model. Ecological Modelling 155:85-94.

Naiman RJ, editor. 1992. Watershed management: balancing sustainability and environmental change. New York (NY): Springer-Verlag New York. 542 p.

[NOAA] National Oceanic and Atmopheric Administration. 2008. Worst New England Harmful Algal Bloom in 30 Years. [Internet]. Silver Spring (MD): NOAA; Center for Sponsored Coastal Ocean Research (CSCOR). [updated 2008 May 14; cited 2008 Sep 22]. Available from: http://www.cop.noaa.gov/news/fs/ne_hab_200505.html.

[NRC] National Research Council. 1996. Upstream: salmon and society in the Pacific Northwest. Washington (DC): National Academy Press. 452 p.

[NRC] National Research Council. 1999. Hardrock mining on federal lands. Appendix B. Potential environmental impacts of hardrock mining. Washington (DC): National Academy Press. p 149-168.

Nechvatal M, Granata T. 2004. Dam removal as a solution to increase river water quality. In: Self sustaining solutions for streams, wetlands, and watersheds; 2004 Sep 12-15; St. Paul, MN. St. Joseph (MI): American Society of Agricultural and Biological Engineers.

Nelson RL, McHenry M, Platts WS. 1991. Mining. In: Meehan WR, editor. Influences of forest and rangeland management in salmonid fishes and their habitats. Bethesda (MD): American Fisheries Society. Special Publication 19. p 425-58.

Newell RC, Seiderer LJ, Hitchcock DR. 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. In: Ansell AD, Gibson RN, Barnes M, editors. Oceanography and marine biology: an annual review. Vol. 36. Oban, Argyll (Scotland): UCL Press. p 127-78.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol 1.

Nightingale B, Simenstad CA Jr. 2001. Dredging activities: marine issues. [Internet]. Olympia (WA): Washington Department of Fish and Wildlife. White Paper. [cited 2008 Jul 15]. 119 p + appendices. Available from: http://wdfw.wa.gov/hab/ahg/finaldrg.pdf.

Niimi AJ. 2004. Role of container vessels in the introduction of exotic species. Marine Pollution Bulletin 49(9-10):778-82.

Nislow KH, Magilligan FJ, Fassnacht H, Bechtel D. 2002. Effects of dam impoundment on the flood regime of natural floodplain communities in the upper Connecticut River. Journal of the American Water Resources Association 38(6):1533-48.

Ormerod SJ. 2003. Current issues with fish and fisheries: editor's overview and introduction. Journal of Applied Ecology 40(2):204-13.

[PFMC] Pacific Fisheries Management Council. 1999. Appendix A: identification and description of essential fish habitat, adverse impacts, and recommended conservation measures for salmon. Amendment 14 to the Pacific Coast Salmon Plan. [Internet]. Portland (OR). [updated 2007 May 8; cited 2007 Dec 21]. Available from: http://www.pcouncil.org/salmon/salfmp/a14.html.

Packer DB, Griffin K, McGlynn KE. 2005. National Marine Fisheries Service national gravel extraction guidance. [Internet]. Washington (DC): US Department of Commerce. [Internet]. NOAA Technical Memorandum NMFS-F/SPO-70. [cited 2008 Jul 15]. 27 p. Available from: http://www.nmfs.noaa.gov/habitat/habitatprotection/anadfish/gravel.htm.

Parrish DL, Behnke RJ, Gephard SR, McCormick SD, Reeves GH. 1998. Why aren't there more Atlantic salmon (*Salmo salar*)? Canadian Journal of Fisheries and Aquatic Sciences 55(Suppl. 1):281-7.

Pearce JB. 1994. Mining of seabed aggregates. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 48-50.

Pizzuto J. 2002. Effects of dam removal on river form and process. BioScience 52(8):683-91.

Poff NL, Allan JD, Bain MB, Karr JR, Prestegaard KL, Richter BD, Sparks RE, Stromberg JC. 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience 47:769-84.

Raymond HL. 1979. Effects of dams and impoundments on migrations of juvenile Chinook salmon and steelhead from the Snake River, 1966 to 1975. Transactions of the American Fisheries Society 108:505-29.

Ruggles CP. 1980. A review of the downstream migration of Atlantic salmon. Halifax (Nova Scotia): Freshwater and Anadromous Division, Resource Branch, Department of Fisheries and Oceans. Canadian Technical Report of the Fisheries and Aquatic Sciences No. 952. ix + 39 p.

Spence BC, Lomnicky GA, Hughes RM, Novitzki RP. 1996. An ecosystem approach to salmonid conservation. [Internet]. Corvallis (OR): ManTech Environmental Research Services Corp. TR-4501-96-6057. [cited 2008 Jul 9]. Available from: http://www.nwr.noaa.gov/Publications/Reference-Documents/ManTech-Report.cfm.

Starnes LB, Gasper DC. 1996. Effects of surface mining on aquatic resources in North America. Fisheries 21(5):24-6.

Strayer DL, Hattala KA, Kahnle AW. 2004. Effects of an invasive bivalve (*Dreissena polymorpha*) on fish in the Hudson River estuary. Canadian Journal of Fisheries and Aquatic Sciences 61(6):924-41.

Thomson JR, Hart DD, Charles DF, Nightengale TL, Winter DM. 2005. Effects of removal of a small dam on downstream macro invertebrate and algal assemblages in a Pennsylvania stream. Journal of the North American Benthological Society 24(1):192-207.

Thurberg FP, Gould E. 2005. Pollutant effects upon cod, haddock, pollock, and flounder of the inshore fisheries of Massachusetts and Cape Cod Bays. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 43-66.

Travnichek VH, Zale AV, Fisher WL. 1993. Entrainment of ichthyoplankton by a warmwater hydroelectric facility. Transactions of the American Fisheries Society 122(5):709-16.

[USACE] US Army Corps of Engineers. 2005. Maine programmatic general permit no. NAE-2005-584 (2005 Jul 5). Concord (MA): Army Corps of Engineers, New England District.

[USEPA] US Environmental Protection Agency. 2000. Environmental screening checklist and workbook for the water transportation industry. Washington (DC): US EPA Office of Water. EPA-305-B-00-004.

[USEPA] US Environmental Protection Agency. 2003. National management measures for the control of non-point pollution from agriculture. [Internet]. Washington (DC): US EPA Office of Water. EPA-841-B-03-004. [cited 2008 Jul 15]. Available from: http://www.epa.gov/owow/nps/agmm/index.html.

[USEPA] US Environmental Protection Agency. 2007. US EPA Superfund Program. [Internet]. [updated 2007 Dec 18; cited 2007 Dec 21]. Available from: http://www.epa.gov/superfund/index.htm.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1999. Status review of anadromous Atlantic salmon in the United States. Hadley (MA): USFWS. p 1-131.

[USGAO] US General Accounting Office. 2001. Restoring fish passage through culverts on Forest Service and BLM lands in Oregon and Washington could take decades. Washington (DC): US GAO. GAO-02-136. 29 p.

[USGS] US Geological Survey. 2002. Pharmaceuticals, hormones, and other organic wastewater contaminants in U.S. streams. [Internet]. Reston (VA): US Geological Survey. USGS Fact Sheet FS-027-02. [cited 2008 Jul 15]. 2 p. Available from: http://toxics.usgs.gov/pubs/FS-027-02/pdf/FS-027-02.pdf.

[VADMME] Virginia Department of Mines, Minerals, and Energy. 2007. Mineral Resources. [Internet]. Richmond (VA): VADMME. [cited 2007 Dec 28]. Fourteen articles available from: http://geology.state.va.us/DMR3/Mineralresources.shtml.

Wilbur AR, Pentony MW. 1999. Human-induced nonfishing threats to essential fish habitat in the New England region. American Fisheries Society Symposium 22:299-321.

Wilson WJ, Gallaway BJ. 1997. Synthesis in applied fish ecology: twenty years of studies on effects of causeway development on fish populations in the Prudhoe Bay Region, Alaska. American Fisheries Society Symposium 19:326-39.

[WHOI] Woods Hole Oceanographic Institute. 2008. Northeast PSP, New England harmful algal bloom/red tide information. [Internet]. Woods Hole (MA): WHOI. [updated 2008 May 30; cited 2008 Sep 22]. Available from: http://www.whoi.edu/northeastpsp/.

Zwick P. 1992. Stream habitat fragmentation-a threat to biodiversity. Biodiversity and Conservation 1:80-97.

CHAPTER FIVE: MARINE TRANSPORTATION

Introduction

The demand for increased capacity of marine transportation vessels, facilities, and infrastructure is a global trend that is expected to continue in the future. This demand is fueled by a need to accommodate growing vessel operations for cargo handling activities and human population growth in coastal areas. As coastal areas continue to grow, there is a concomitant increase in the demand for water transportation services and recreational opportunities.

It is also important to note that coastal areas under high developmental pressure are often located adjacent to productive and sensitive aquatic environments. Historically, human settlements in the northeastern United States were probably established on the basis of availability to food resources and marine transportation. Coastal features such as estuaries and embayments satisfied these needs as they are highly productive ecosystems ideal for fishing, farming, or hunting and are sheltered waters that provide access to rivers and the ocean for transportation purposes. Today, urban growth and development in coastal areas are growing at a rate approximately five times that of other areas of the country and over one-half of all Americans live within 50 miles of the coast (Markham 2006). The continued demand on the coast today is likely attributed to the highly desirable aesthetic quality and recreational opportunities, including access to fishing, beaches, and boating.

The expansion of port facilities, vessel operations, and commercial and recreational marinas can have adverse impacts on fishery habitat. The growth of the marine transportation industry is accompanied by land-use changes, including over-water or in-water construction, filling of aquatic habitat and wetlands, and increased maintenance activities. Although some categories of habitat impacts resulting from activities related to port and marina construction and maintenance and vessel operations may be minimal and site specific, the cumulative effects of these activities over time can have substantial impacts on habitat.

The construction of new ports and marinas typically involves the removal of sediments by dredging from intertidal and subtidal habitats in order to create navigational channels, turning basins, anchorages, and berthing docks for the size and types of vessels expected to use the facilities. For existing ports and marinas, dredging is generally conducted on a routine basis in order to maintain the required depths as sediment is transported and deposited into the channels, basins, anchorages, and docks. The construction of new ports and marinas, or the expansion of existing facilities, is often referred to as “improvement” dredging; whereas, dredging existing ports and marinas in order to maintain an assigned or authorized depth is generally referred to as “maintenance” dredging. Because the chemical, physical, and biological impacts associated with both “improvement” and “maintenance” dredging are similar in nature, both types of dredging are discussed in the Navigation Dredging section of this chapter. Other impacts associated with newly constructed and expanded ports and marinas are covered under the Construction and Expansion of Ports and Marinas section of this chapter.

Construction and Expansion of Ports and Marinas

Construction of ports and marinas can change physical and chemical habitat parameters such as tidal prism, depth, water temperature, salinity, wave energy, sediment transport, and current velocity. Alterations to physical characteristics of the coastal ecosystems can cause adverse effects to biological parameters, such as the composition, distribution, and abundance of shellfish and submerged aquatic vegetation (SAV). These changes can impact the distribution of nearshore habitats and affect aquatic food webs.

Loss and conversion of habitat

Port and marina facilities are typically located in areas containing highly productive intertidal and subtidal habitats, including saltmarsh wetlands and SAV. Coastal wetlands provide a number of important ecological functions, including foraging, spawning/breeding, protection from predators, as well as nutrient uptake and release and retention of storm and floodwaters. Vegetated wetlands and intertidal habitats are some of the most highly productive ecosystems in the world, and support one or more life stages of important commercial and recreational fishery resources in the United States (Dahl 2006). One of the most obvious habitat impacts related to the construction of a port or marina facility is alteration or loss of physical space taken up by the structures required for such a facility. The construction of ports and marinas can alter or replace salt marsh, SAV, and intertidal mud flat habitat with "hardened" structures such as concrete bulkheads and jetties that provide relatively few ecological functions. Boston Harbor, MA, exemplifies a northeastern coastal port transformed by expansive dredging and filling of former shallow estuarine waters and salt marsh wetlands. Between 1775 and 1980, wetland filling within the harbor extensively altered the shoreline, with the airport alone amounting to 2,000 acres of filled intertidal salt marsh wetlands (Deegan and Bushbaum 2005).

Over-water structures, such as commercial and residential piers and docks, floating breakwaters, barges, rafts, booms, and mooring buoys are associated with port and marina facilities and are constructed over both subtidal and intertidal habitats. Although they generally have less direct physical contact with benthic habitats than in-water structures, float, raft, and barge groundings at low tides and the scouring of the substrate by the structures and anchor chains can be substantial. Piles and other in-water structures can alter the substrate below and adjacent to the structures by providing a surface for encrusting communities of mussels and other sessile organisms, which can create shell deposits and shift the biota normally associated with sand, gravel, mud, and eelgrass substrates to those communities associated with shell hash substrates (Penttila and Doty 1990; Nightingale and Simenstad 2001a).

Shoreline armoring is an in-water activity associated with the construction and operation of marinas and ports, intended to protect inland structures from storm and flood events and to prevent erosion that is often a result of increased boat traffic. Armoring of shorelines to prevent erosion and maintain or create shoreline development simplifies habitats, reduces the amount of intertidal habitat, and affects nearshore processes and the distribution of aquatic communities (Williams and Thom 2001). Hydraulic effect alterations to the shoreline include increased energy seaward of the armoring from reflected wave energy, which can exacerbate erosion by coarsening the substrate and altering sediment transport (Williams and Thom 2001). Installation of breakwaters and jetties can also result in community changes, including burial or removal of resident biota, changes in cover, preferred prey species, predator interaction, and the movement of larvae (Williams and Thom 2001). Chapman (2003) found a paucity of mobile species associated with seawalls in a tropical estuary, compared with surrounding areas.

Altered light regimes and loss of submerged aquatic vegetation

Alteration of the light regimes in coastal waters can affect primary production, including the distribution and density of SAV, as well as the feeding and migratory behavior of fish. Over-water structures shade the surface of the water and attenuate the sunlight available to the benthic habitat under and adjacent to the structures. The height, width, construction materials used, and the orientation of the structure in relation to the sun can influence how large a shade footprint an over-water structure may produce and how much of an adverse impact that shading effect may have on the localized habitat (Fresh et al. 1995; Burdick and Short 1999; Shafer 1999; Fresh et al. 2001). High, narrow piers and docks produce more diffuse shadows which have been shown to reduce shading impacts to SAV (Burdick and Short 1999; Shafer 1999).

The density of pilings can also determine the amount of light attenuation created by dock structures. Piling density is often higher in larger, commercial shipping ports than in smaller recreational marinas, as larger vessels and structures often require a greater number of support structures such as fenders and dolphin piles. Light limitations caused by pilings can be reduced through adequate spacing of the pilings and the use of light reflecting materials (Thom and Shreffler 1996; Nightingale and Simenstad 2001a). In addition, piers constructed over solid structures, such as breakwaters or wooden cribs, would further limit light transmittance and increase shading impacts on SAV.

Although shading impacts are greatest directly under a structure, the impacts on SAV may extend to areas adjacent to the structure as shadows from changing light conditions and adjacent boats or docks create light limitations (Burdick and Short 1999; Smith and Mezich 1999). A decrease in SAV and primary productivity can impact the nearshore food web, alter the distribution of invertebrates and fish, and reduce the abundance of prey organisms and phytoplankton in the vicinity of the over-water structure (Kahler et al. 2000; Nightingale and Simenstad 2001a; Haas et al. 2002).

The sharp light contrasts created by over-water structures because of shading during the day and artificial lighting at night can alter the feeding, schooling, predator avoidance, and migratory behaviors of fish (Nightingale and Simenstad 2001a; Hanson et al. 2003). Fish, especially juveniles and larvae, rely on visual cues for these behaviors. Shadows create a light-dark interface which may increase predation by ambush predators and increase starvation through limited feeding ability (Able et al. 1999; Hanson et al. 2003). In addition, the migratory behavior of some species may favor deeper waters away from shaded areas during the day and lighted areas may affect migratory movements at night, contributing to increased risk of predation (Nightingale and Simenstad 2001a).

Altered temperature regimes

Shoreline modifications, including the construction of seawalls and bulkheads, can alter nearshore temperature regimes and natural communities. Modified shorelines invariably contain less shoreline vegetation than do natural shorelines, which can reduce shading in the nearshore intertidal zone and cause increases in water temperatures (Williams and Thom 2001). Conversely, seawalls and bulkheads constructed along north facing shorelines may unnaturally reduce light levels and reduce water temperatures in the water column adjacent to the structures (Williams and Thom 2001).

Siltation, sedimentation, and turbidity

The construction of a new port or marina facility is usually associated with profound changes in land use and in-water activities. Because a large proportion of the shoreline associated with a port is typically replaced with impervious surfaces such as concrete and asphalt, stormwater runoff is exacerbated and can increase the siltation and sedimentation loads in estuarine and marine habitats. The upland activities related to building roads and buildings may cause erosion of topsoil which can be transported through stormwater runoff to the nearshore aquatic environment, increasing sedimentation and burying benthic organisms. Construction and expansion of ports and marinas generally include dredging channels, anchorages, and berthing areas for larger and greater numbers of vessels, which contribute to localized sedimentation and turbidity. In addition, the use of underwater explosives to construct bulkheads, seawalls, and concrete docks may temporarily resuspend sediments and cause excessive turbidity in the water column and impact benthic organisms. Refer to the section on Navigation Dredging later in this chapter for information on channel dredging.

Impacts associated with increased suspended particles in the water column include high turbidity levels, reduced light transmittance, and sedimentation which may lead to reductions or loss of SAV and other benthic habitats. Elevated suspended particles have also been shown to adversely affect the respiration of fish, reduce filtering efficiencies and respiration of invertebrates, reduce egg buoyancy, disrupt ichthyoplankton development, reduce the growth and survival of filter feeders, and decrease the foraging efficiency of sight-feeders (Messieh et al. 1991; Barr 1993).

Structures such as jetties and groins may be constructed to reduce the accretion of sediment in navigable channels, so by design they alter littoral sediment transport and change sedimentation rates. These structures may reduce sand transport, cause beach and shoreline erosion to down drift areas, and may also interfere with the dispersal of larvae and eggs along the coastline (Williams and Thom 2001). Substrate disturbance from pile driving and removal can increase turbidity, interfere with fish respiration, and smother benthic organisms in adjacent areas (Mulvihill et al. 1980). In addition, contaminants in the disturbed sediments may be resuspended into the water column, exposing aquatic organisms to potentially harmful compounds (Wilbur and Pentony 1999; USEPA 2000; Nightingale and Simenstad 2001b). Refer to the Coastal Development chapter for a more detailed discussion on impacts related to pile driving and removal.

Contaminant releases

The construction of ports and marinas can alter natural currents and tidal flushing and may exacerbate poor water quality conditions by decreasing water circulation. Bulkheads, jetties, docks, and pilings can create water traps that accumulate contaminants or nutrients washed in from land based sources, vessels, and facility structures. These conditions may create areas of low dissolved oxygen, dinoflagellate blooms, and elevated toxins.

Contaminants can be released directly into the water during construction activities associated with new ports and marinas or indirectly through storm water runoff from land-based operations. Accidental and incidental spills of petroleum products and other contaminants, such as paint, degreaser, detergents, and solvents, can occur during construction operations of a facility. Large amounts of impervious surfaces at ports and marinas can increase, and in some cases direct, stormwater runoff and contaminants into aquatic habitats. The use of certain types of underwater explosives to construct bulkheads, seawalls, and concrete docks may release toxic chemicals (e.g., ammonia) in the water column that can impact aquatic organisms.

Wood pilings and docks used in marina and port construction are often treated with chemicals such as chromated copper arsenate, ammoniacal copper zinc, and creosote to help extend the service of the structures in the marine environment. These preservatives can leach harmful chemicals into the water that have been shown to produce toxic affects on fish and other organisms (Weis et al. 1991). Creosote-treated wood for pilings and docks has also been used in marine environments and has been shown to release polycyclic aromatic hydrocarbons (PAH) continuously and for long periods of time after installation or treatment; whereas other chemicals that are applied to the wood, such as ammoniacal copper zinc arsenate (ACZA) and chromated copper arsenate (CCA), tend to leach into the environment for shorter durations (Poston 2001). Affects from exposure of aquatic organisms to PAH include carcinogenesis, phototoxicity, immunotoxicity, and disturbance of hormone regulation (Poston 2001). The rate and duration that these preservatives can be leached into marine waters after installation are highly variable and dependent on many factors, including the length of time since the treatment of the wood and the type of compounds used in the preservatives. The toxic effects of metals such as copper on fish are well known and include body lesions, damage to gill tissue, and interrupted cellular functions (Gould et al. 1994). 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 may cause localized reductions in species richness and diversity (Weis and Weis 2002). Concrete, steel, or nontreated wood are relatively inert and generally do not leach contaminants into the water.

Dredging and filling of intertidal and subtidal habitats can resuspend sediments into the water column that may have been contaminated by nearby industrial activities. Information on contaminant releases from dredging can be found in the Navigation Dredging section of this chapter and the Chemical Effects: Water Discharge Facilities chapter of the report.

Altered tidal, current, and hydrologic regimes

One of the primary functions of a marina or port is to shelter and protect boats from wave energy. In-water structures of ports and marinas such as bulkheads, breakwaters, jetties, and piles result in localized changes to tidal and current patterns. These alterations may exacerbate poor water quality conditions in these facilities by reducing water circulation. In addition, in-water structures interfere with longshore sediment transport processes resulting in altered substrate amalgamation, bathymetry, and geomorphology. Changing the type and distribution of sediment may alter key plant and animal assemblages, starve nearshore detrital-based foodwebs, and disrupt the natural processes that build spits and beaches (Nightingale and Simenstad 2001a; Hanson et al. 2003).

The protected, low energy nature of marinas and ports may alter fish behavior as juvenile fish show an affinity to structure and may congregate around breakwaters or bulkheads (Nightingale and Simenstad 2001a). These alterations in behavior may make them more susceptible to predation and may interfere with normal migratory movements.

Underwater blasting and noise

Noise from underwater blasting and in-water construction generates intense underwater sound pressure waves that may adversely affect marine organisms. These pressure waves have been shown to injure and kill fish (Caltrans 2001; Longmuir and Lively 2001; Stotz and Colby 2001). Fish are known to use sound for prey and predator detection as well as social interaction (Richard 1968; Myrberg 1972; Myrberg and Riggio 1985; Hawkins 1986; Kalmijn 1988), and underwater blasting and noise may alter the their distribution and behavior (Feist et al.1996).

Generally, aquatic organisms that possess air cavities (i.e., lungs and swim bladders) are more susceptible to underwater blasts than those without (Keevin et al. 1999). In addition, smaller fish are more likely to be impacted by the shock wave of underwater blasts than are larger fish, and the eggs and embryos tend to be particularly sensitive; however, fish larvae tend to be less sensitive to blasts than eggs or post-larvae fish, probably because the larvae stages do not yet possess air bladders (Wright 1982; Keevin et al. 1999).

Blasting may be used for dredging new navigation channels and boat basins or expanding existing channels in areas containing rock substrates, boulders, and ledges. The construction of new in-water structures, such as bulkheads, seawalls, and concrete docks also may involve blasting. Blasting represents a single point of disturbance with a restricted, and often predictable, mortality zone. In addition, blasting engineers purposefully focus the blast energy towards fracturing rock substrate and prevent excess energy from being released into the water column (Keevin et al. 1999). Techniques used to prevent blasting damage to structures in the vicinity of a project, such as bubble curtains, may be effective mitigation measures for reducing blasting impacts on aquatic biota (Keevin et al. 1999). Although the use of bubble curtains have been shown to be effective at minimizing pressure wave impacts on fish (Keevin et al. 1997; Longmuir and Lively 2001), the difficulty of deploying bubble curtains in field conditions may reduce the efficacy of this technology in mitigating these effects (Keevin et al. 1997).

Unlike blasting, pile driving is a repeating sound disturbance that can last for extended periods of time during construction. There are several factors which affect the type and intensity of sound pressure waves during pile driving, including the size and material of the piling, the firmness of the substrate, and the type of pile-driving hammer that is used (Hanson et al. 2003). Wood and concrete piles produce lower sound pressures than do steel piles. Pile driving in firmer substrate, which requires more energy, will produce more intense sound pressures (Hanson et al. 2003). Both impact hammers and vibratory hammers are commonly used when driving pilings into the substrate. Vibratory hammers produce sounds with more energy in the lower frequencies (15-26 Hz), compared to higher frequency noise generated by impact hammers (100-800 Hz) (Carlson et al. 2001). The behavioral response elicited by fish differs in these two ranges of sound frequencies. Fish respond to sounds similar to vibratory hammers by consistently displaying an avoidance response and not habituating to the sound despite repeated exposure (Dolat 1997; Knudsen et al. 1997; Sand et al. 2000). In contrast to vibratory hammers, fish may be initially startled by an impact hammer but eventually become habituated and no longer respond to the stimuli. Acclimation to the sound may place fish in more danger as they remain in range of potentially harmful sound pressure waves (Dolat 1997). Refer to the chapter on Global Effects and Other Impacts for additional information on underwater noise impacts to aquatic organisms.

Conservation recommendations and best management practices for construction and expansion of ports and marinas

1. Encourage federal, state, and local authorities to assist port authorities and marinas in developing management plans that avoid and minimize impacts to the coastal environment and that are consistent with coastal zone management plans.
2. Encourage implementation of environmental management systems for ports and marinas that incorporate strong operational controls and best management practices (BMPs) into existing job descriptions and work instruction.
3. Encourage marinas to participate in NOAA/US EPA’s Coastal Nonpoint Program and the Clean Marina Initiative.
4. Explore alternative port developments such as satellite ports and offshore terminals, which may decrease some impacts associated with traditional inshore port facility developments.
5. Conduct site suitability analyses for new or proposed expansion of port and marina facilities to reduce and avoid habitat degradation or loss. Some of the analyses that should be conducted include identifying alterations to current and circulation patterns, water quality, bathymetric and topographic features, fisheries utilization and species distributions, and substrate features.
6. Conduct pre- and post-project biological surveys over multiple growing seasons to assess impacts on submerged and emergent aquatic vegetation communities.
7. Site new or expansions of port and marina facilities in deep-water areas to the maximum extent practicable to avoid the need for dredging. Areas that are subject to rapid shoaling or erosion will likely require more frequent maintenance dredging and should be avoided.

8. Avoid areas identified as supporting high abundance and diversity of species (e.g., SAV beds, intertidal mudflats, emergent wetlands, fish spawning areas) when locating new or expanded port and marina facilities.
9. Encourage the use of preproject surveys by qualified biologists/botanists to identify and map invasive plants within the proposed project area, and develop and implement an eradication plan for nonnative species.
10. Consider excavating uplands as a less-damaging alternative for new or expanded port and marina facilities instead of dredging intertidal or shallow subtidal habitat. However, water quality modeling should be conducted to evaluate potential impacts associated with enclosed and poorly flushed marinas.
11. Retain and preserve marine riparian buffers to maintain intertidal microclimate, flood and stormwater storage capacity, and nutrient cycle.
12. Consider low-wake vessel technology and appropriate vessel routes in the facility design and permitting process to minimize impacts to shorelines and shallow water habitats. Vessel speeds should be adapted to minimize wake damage to shorelines, and no-wake zones should be considered in highly sensitive areas, such as fish spawning habitat and SAV beds.
13. Do not locate new port and marina facilities in areas that have reduced tidal exchange and/or shallow water habitats, such as enclosed bays, salt ponds, and tidal creeks.
14. Implement construction designs for new ports and marinas to facilitate good tidal exchange and surface water movement and provide an adequate migratory corridor for fish. When possible, structures that impede tidal exchange and that may interfere with the movement of marine organisms, such as solid breakwaters, should be avoided.
15. Ensure that new ports and marinas incorporate BMPs in the construction operation plans that prevent and minimize the release of contaminants and debris caused by construction equipment and activities. The plan should include a spill response plan and training, and spill response equipment should be installed and maintained properly on-site.
16. Implement seasonal restrictions when necessary to avoid construction-related impacts to habitat during species' critical life history stages (e.g., spawning and egg development periods).
17. For structures located over SAV, the amount of light reaching vegetation below the dock should be maximized by providing adequate height over the water, minimizing the width of the dock, and orienting the length of the dock in a north-south direction.
18. The use of wood preservatives, such as creosote, ACZA and CCA should be avoided, where possible. If CCA treated wood must be used, the wood can be presoaked for several weeks or the wood can be coated with plastic sheath to reduce/eliminate leaching. Concrete and steel pilings are generally considered to be less damaging, since they reflect light more than wood docks and generally do not release contaminants into the aquatic environment. However, concrete pilings and docks generally increase the overall size of the overwater structure and may not be preferable in areas containing SAV.
19. Site floating docks, which limit light transmittance more than elevated structures, only in nonvegetated areas. When used, floating docks should either be located in areas of adequate depth so that adequate clearance between the float and the bottom is maintained, or fitted with structures (i.e., float stops) that prevent the float from contacting the bottom. Float stops should be designed to provide a minimum of 2 feet 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.
20. Orient night lighting such that illumination of the surrounding waters is avoided.

21. Reduce sound pressure impacts during pile installation by using wood or concrete piles, rather than hollow steel piles which produce intense, sharp spikes of sound that are more damaging to fish.
22. Use technologies that have been designed to reduce the adverse effects of underwater sound pressure waves such as air bubble curtains and metal or fabric sleeves to surround the pile. Air bubble systems must have adequate airflow, and the pile should be fully contained to ensure that sound attenuation is successful.
23. Conduct pile driving during low tides in intertidal and shallow subtidal areas.
24. Employ vibratory hammers when removing old piles to help minimize the release of suspended sediments, silt, and contaminants into the water column; these may be preferable over direct pull or the use of a clamshell dredge.
25. Reduce or eliminate the amount of sediment released into the water column by cutting the pile off below the mudline and leaving the stub in place when removing old piles.
26. Mitigate impacts to marine organisms, particularly those with air cavities (i.e., swim bladders and lungs), from underwater blasting by employing BMPs such as focusing the blast energy towards a solid rock substrate rather than towards the water column; installing noise attenuating devices such as air curtains; conducting the blasting during periods of low-water or low-tide; using delayed blasts that produce sequenced, lesser-charged explosions that reduce the shockwave; stemming (capping) the charge bore hole with material that contains the blast; and repelling charges that frighten fish from the blast area prior to blasting (Keevin 1998).
27. Consult federal and state resource agencies prior to work that involves blasting to assess the marine resource utilization of the area. Biological surveys may be required to assess the presence of fishery resources. Time-of-year restrictions should be employed to avoid impacting sensitive species and life history stages that use the area. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
28. Integrate measures to reduce nonpoint source (NPS) pollution, such as a stormwater management plan into the design, maintenance, and operation of a port or marina. Some examples of BMPs for stormwater management include (adapted from Amaral et al. 2005):
    a. Minimize the amount of impervious surfaces surrounding the port or marina facility and maintain a buffer zone between the coastal zone and upland facilities.
    b. Implement runoff control strategies to decrease the amount of contaminants entering marine waters from upland sources. This can be accomplished by using alternative surface materials such as crushed gravel, decreasing the slope of surfaces towards the waters' edge, and installing filtering systems or settling ponds.
    c. Designate specific enclosed areas for maintenance activities such as sanding, painting, engine repairs. Use tarp enclosures or spray booths for abrasive blasting to prevent residue from reaching surface waters.
    d. Provide and maintain appropriate storage, transfer, containment, and disposal facilities for liquid hazardous material, such as solvents, antifreeze, and paints.
29. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in port and marina review processes.

Operation and Maintenance of Ports and Marinas

Existing ports and marinas can be a source of impacts to fishery resources and habitat that may differ from those relating to construction and expansion of new facilities. These impacts may be associated with the operation of the facilities, equipment impacts, and stormwater runoff. Examples of port or marina impacts include chronic pollution releases, underwater noise, altered light regimes, and repeated physical disturbances to benthic habitats.

Contaminant release and storm water runoff

Ports and marinas can be a source of contaminants directly associated with facility activities and by stormwater runoff from the facility and the surrounding urbanized areas. The long-term operation of a marina or port can provide a chronic presence of contaminants to the localized area that can have an adverse effect on the quality of fishery habitat and population dynamics (Wilbur and Pentony 1999).

The oil and fuel that accumulates on dock surfaces, facilities properties, adjacent parking lots, and roadways may enter coastal waters through stormwater runoff and snowmelt. Oil and fuel contains PAH and other contaminants that are known to bioaccumulate in marine organisms and impact the marine food web (Nightingale and Simenstad 2001a; Amaral et al. 2005). In addition, these contaminants can persist in bottom sediments where they can be resuspended through a variety of activities such as propeller scouring and dredging. Marina activities such as vessel refueling, engine repair, and accidental vessel sinking may increase the risk of fuel and oil contamination of the surrounding environment (Amaral et al. 2005).

Marina facilities such as storage areas for paint, solvents, detergents, and other chemicals may pose a risk of introducing additional contaminants to the marine environment resulting in both acute and chronic toxicity to marine biota (Amaral et al. 2005). These products are often a routine and essential part of marina or port operations, and if handled and stored improperly they can increase the risk of accidental spillage. Various port and vessel maintenance activities may contribute to metal contamination to the surrounding waters. For example, elevated levels of copper are often associated with ports and marinas, especially those with a high density of recreational boats because of the type of antifouling paints used on those boats. A number of other metals have been detected in the sediments and surface waters of marinas, including arsenic (used in paints and wood preservatives), zinc (leached from anodes used to reduce corrosion of boat hulls and motors), mercury (used in float switches for bilge and other storage tank pumps), lead (used in batteries), nickel, and cadmium (used in brake linings) (USEPA 2001). However, stormwater runoff may be the primary source of copper in most marinas in urban areas (Warnken et al. 2004).

Wooden pilings and docks in marinas and ports are typically treated with some type of preservative, such as chromated copper arsenate, ammoniacal copper zinc, and creosote. These preservatives can leach harmful chemicals into the water that have been shown to have toxic effects on fish and other organisms (Weis et al. 1991). Concrete, steel, or nontreated wood are relatively inert and do not leach contaminants into the water. Refer to this chapter’s section on Construction and Expansion of Ports and Marinas and the Coastal Development chapter for more information on the affects of copper and other wood preservatives on aquatic resources.

Because marinas and ports typically contain large areas of impervious surfaces and are located at the interface between land and water, stormwater runoff can be greater at these facilities compared with other types of land uses. The organic particulates that are washed into marine waters from the surrounding surfaces can add nutrients to the water and cause eutrophication in bays and estuaries. A number of sources of organic matter from ports and marinas can degrade water quality and reduce dissolved oxygen concentrations, including sewage discharges from recreational and commercial boats, trash tossed overboard, fish wastes disposed of into surface waters, pet wastes, fertilizers, and food wastes (USEPA 2001). Eutrophication often leads to abnormally high phytoplankton populations, which in turn can reduce the available light to SAV beds. Changes in water quality caused by eutrophication can sometimes have a more severe impact on seagrass populations than shading from over-water structures or physical uprooting by vessel and float groundings (Costa et al. 1992; Burdick and Short 1999).

Release of debris

Solid waste is another problematic issue associated with port and marina operations. A great deal of solid waste is generated through daily operations of a commercial port as well as the recreational activities of a marina. This waste may include plastics such as fishing line, bottles, tarps, food containers, and shopping bags, or paper products and other materials, which can be released as debris into the surface waters through accidental loss from vessels or through stormwater runoff from upland facilities. Activities such as sanding, pressure washing, sand blasting, and discarding rags and oil/fuel filters can contribute to marine debris if improper handling and disposal is allowed (USEPA 2001). If this waste is collected and disposed of properly the impacts to the environment can be minimized (Amaral et al. 2005). Plastics are a large component of the trash released into marine waters, accounting for 50-60% of marine debris collected from the Gulf of Maine (Hoagland and Kite-Powell 1997). Plastics contain toxic substances that can persist in the environment and bioaccumulate through the food web, impairing metabolic functions in fish and invertebrates that use habitats polluted by plastic debris. Some chemicals found in plastics, known as "endocrine disruptors," may interfere with the endocrine system of aquatic organisms (Kavlock et al. 1996; Kavlock and Ankley 1996). These chemicals act as "environmental hormones" that may mimic the function of the sex hormones androgen and estrogen (Thurberg and Gould 2005). Adverse effects include reduced or altered reproductive functions, which could result in population-level impacts.

Marine debris can directly affect fish and invertebrates that may consume or become entangled by the debris. Plastic debris may be ingested by seabirds, fish and invertebrates, sea turtles, and marine mammals, which can cause infections and death of the animal (Cottingham 1988). Debris can be transported by currents to other areas where it can become snagged and attached to benthic habitat, damaging sensitive reef habitat. Additional information on impacts associated with marine debris can be found under Operation and Maintenance of Vessels section of this chapter and in the Coastal Development chapter of this report.

Underwater noise

The ambient noises emanating from ports and marinas are from a combination of boat propellers, engines, pumps, generators, and other equipment within vessels and shore-side equipment. In coastal areas the sounds of cargo and tanker traffic are multiplied by complex reflected paths from scattered and reverberated noises caused by littoral geography. Commercial and private fishing boats, pleasure craft, personal watercraft (i.e., jet skis), industrial vessels, public transport ferries, and shipping safety and security services such as tugs boats, pilot boats, enforcement vessels, and coastal agency support craft generate sounds that can impact marine organisms, particularly fish and marine mammals. Exposure to continuous noise may also create a shift in hearing thresholds for marine organisms resulting in hearing losses at certain frequency ranges (Jasny et al. 1999). Refer to the Global Effects and Other Impacts chapter and the Operation and Maintenance of Vessels and the Construction and Expansion of Ports and Marinas sections in this chapter for more information on underwater noise.

Derelict structures

Increased vessel activity in and around port and marina operations increase the probability of the grounding of vessels, which may not always be removed immediately from the aquatic environment. In addition to being public health and navigational hazards, derelict or abandoned vessels can cause various impacts to coastal habitats. Grounded vessels can physically damage and smother benthic habitats, create changes in wave energy and sedimentation patterns, and scatter debris across sensitive habitats (Precht et al. 2001; Zelo and Helton 2005). However, the most common environmental threat of a derelict or abandoned vessel is the release of oil or other pollutants. These hazardous materials may be part of a vessel's cargo, fuel and oil related to vessel operations, or chemicals contained within the vessel's structure which may be released over time through decay and corrosion. Refer to the Operation and Maintenance of Vessels section of this chapter for more information on impacts associated with derelict structures and grounded vessels.

Mooring and floating dock impacts

Vessel mooring impacts, although localized, can reduce habitat quality and complexity. Accidental vessel groundings can smother or crush shellfish, scour vegetation, and disturb substrates (Nightingale and Simenstad 2001a). Disturbance of substrates can lead to increased turbidity, reduced light penetration, decreased dissolved oxygen levels, and the possible resuspension of contaminants. In addition, moored vessels contacting the bottom during low tides can cause the bottom habitat in the area of the mooring to be unavailable for fish and other marine biota during the time the vessel is resting on the bottom. Vessels that contact the bottom can create scouring of the substrate and result in permanent alteration or loss of benthic habitats, such as eelgrass. Demersal eggs (e.g., Atlantic herring [*Clupea harengus*]) and larvae that utilize an area can also be destroyed from the impact of the vessel or shading. Floating piers and docks may also alter wave energy, current patterns, and longshore sediment transport, especially in areas that experience strong current velocities (Nightingale and Simenstad 2001a).

Depending upon the type and configuration, the mooring tackle itself may cause impacts to substrate and benthos, including SAV. Typical vessel moorings consist of an anchor connected to a surface buoy by a long length of heavy chain. In most moorings, some portion of the anchor chain drags and often scours the bottom and forms a depression in the sediment surface (Walker et al. 1989). In areas influenced strongly by tides and currents or wind, the bottom scouring takes on a circular or "V" configuration when the anchor chain is allowed to drag along the bottom as the vessel or buoy swings with the tide or wind (Nightingale and Simenstad 2001a). The resulting scour holes allow further erosion and loss of the physical integrity of the habitat, which can lead to fragmentation of seagrass meadows (Walker et al. 1989; Hastings et al. 1995). Hastings et al. (1995) attributed an approximate 18% direct loss of seagrass habitat from boat moorings in one bay in Western Australia. Refer to the Coastal Development chapter of this report for a more detailed discussion on impacts from overwater structures.

Alteration of light regimes

As discussed in other sections of this chapter, overwater structures shade the surface of the water and attenuate the light available to benthic habitat under and adjacent to the structures. The height, width, construction materials used, and orientation of the structure in relation to the sun can influence how large a shade footprint an over-water structure may produce and how much of an adverse impact that shading effect may have on the benthic habitat (Burdick and Short 1999; Shafer 1999; Fresh et al. 2001; Nightingale and Simenstad 2001a). Refer to the chapter on Coastal

Development and the Construction and Expansion of Ports and Marinas section of this chapter for more information on docks structures and light attenuation.

Conservation recommendations and best management practices for the operation and maintenance of ports and marinas (adapted from Amaral et al. 2005; Hanson et al. 2003)

1. Consider environmental impacts through port development and operations plans, including:
    a. assess all activities at facility and identify potential environmental impacts
    b. determine compatibility with port environmental practices and assess available control technologies
    c. evaluate and monitor effectiveness of control technologies
    d. develop and implement environmental management
2. Encourage marinas to participate in NOAA/US EPA’s Coastal Nonpoint Program and the Clean Marina Initiative.
3. Ensure that marina and port facility operations have an oil spill response plan in place, which has been shown to improve the response and recovery times of oil spills.
4. Ensure that marina or port facilities have adequate oil spill response equipment accessible and clearly marked. Oil spill response equipment may include oil booms, absorbent pads, and oil dispersant chemicals.
5. Use dispersants that remove oils from the environment, rather than those that simply move them from the surface to the ocean bottom.
6. Install automatic shut-off nozzles at fuel dispensing sites and require the use of fuel/air separators on air vents or tank stems of inboard fuel tanks to reduce the amount of fuel oil spilled into surface waters by vessels using fuel stations.
7. Promote the use of oil-absorbing materials in the bilge areas of all boats with inboard engines.
8. Place containment berms around fixed pieces of machinery that use oil and gas within the facility.
9. Encourage public education and signage to promote proper disposal of solid debris and polluting materials.
10. Encourage the proper disposal of materials produced and used by the operation, cleaning, maintenance, and repair of boats to limit the entry of solid and contaminated waste into surface waters.
11. Recommend the placement of garbage containers to supervised areas and use containers that have lids in order to reduce the potential for litter to enter the marine environment.
12. Promote the use of pumpout facilities and restrooms at marinas and ports to reduce the release of sewage into surface waters. Ensure that these facilities are maintained and operational, and provide these services at convenient times, locations, and reasonable cost. In addition, promote the use of these facilities through public education and signage.
13. Develop a harbor management plan which addresses the maintenance and operation of pumpout facilities.
14. Prevent the disposal of fish waste or other nutrient laden material in marina or port basins through the use of public education, signage, and by providing alternate fish waste management practices.
15. Ensure that measures to reduce NPS pollution, such as a stormwater management plan, are integrated into the maintenance and operation of a port or marina.

16. Recommend site-specific solutions to NPS pollution by considering the frequency of marina operations and potential pollution sources. Management practices should be tailored to the specific issues of each marina.
17. Encourage the removal of unnecessary impervious surfaces surrounding the port or marina facility and maintain a buffer zone between the aquatic zone and upland facilities.
18. Ensure that stormwater runoff from parking lots and other impervious surfaces is collected and treated to remove contaminants prior to delivery to any receiving waters. This can be accomplished by using alternative surface materials such as crushed gravel, decreasing the slope of surfaces towards the water's edge, and installing filtering systems or settling ponds.
19. Recommend that specific, enclosed areas are designated for maintenance activities such as sanding, painting, engine repairs. Using tarp enclosures or spray booths for abrasive blasting will also prevent residue from reaching surface waters.
20. Ensure that facilities provide for appropriate storage, transfer, containment, and disposal facilities for harmful liquid material, such as solvents, antifreeze, and paints.
21. Recommend that facilities provide a containment system and a filtering and treatment system for vessel wash down wastewater.
22. Ensure that floating structures, including barges, mooring buoys, and docks are located in adequate water depths to avoid propeller scour and grounding of vessel and floating structures. When floating docks cannot be located in adequate depth to avoid contact on the bottom at low tides, recommend that float stops (structural supports to prevent the float from resting on the bottom) are installed. Float stops should be designed to provide a minimum of 2 feet 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.
23. Recommend anchoring techniques and mooring designs that avoid scouring from anchor chains. For example, anchors that do not require chains (e.g., helical anchors) or moorings that use subsurface floats to prevent anchor chains from dragging the bottom are some designs that should be considered.
24. When moorings with anchor chains cannot be avoided, recommend that areas prone to high current and wind velocity be avoided, where the sweep of the anchor chain on the bottom can cause the greatest damage.
25. Recommend the use of concrete, nontreated wood or steel dock materials to avoid the leaching of contaminants associated with wood preservatives.

Operation and Maintenance of Vessels

Vessel activity in coastal waters is generally proportional to the degree of urbanization and port and harbor development within a particular area. Benthic, shoreline, and pelagic habitats may be disturbed or altered by vessel use, resulting in a cascade of cumulative impacts in heavy traffic areas (Barr 1993). The severity of boating-induced impacts on coastal habitats may depend on the geomorphology of the impacted area (e.g., water depth, width of channel or tidal creek), the current velocity, the sediment composition, the vegetation type and extent of vegetative cover, as well as the type, intensity, and timing of boat traffic (Yousef 1974; Karaki and vanHoften 1975; Barr 1993). Recreational boating activity mainly occurs during the warmer months which coincide with increased biological activity in east coast estuaries (Stolpe and Moore 1997; Wilbur and Pentony 1999). Similarly, frequently traveled routes such as those traveled by ferries and other transportation vessels can impact fish spawning, migration, and recruitment behaviors through noise and direct disturbance of the water column (Barr 1993).

Other common impacts of vessel activities include vessel wake generation, anchor chain and propeller scour, vessel groundings, the introduction of invasive or nonnative species, and the discharge of contaminants and debris (Hanson et al. 2003).

Impacts to benthic habitat

Vessel operation and maintenance activities can have a wide range of impacts to benthic habitat, ranging from minor (e.g., shading of SAV) to potentially large-scale impacts (e.g., ship groundings and fuel or toxic cargo spills). Direct disturbances to bottom habitat can include propeller scouring and vessel wake impacts on SAV and other sensitive benthic habitats and direct contact by groundings or by resting on the bottom at low tides while moored. Propeller scarring can result in a loss of benthic habitat, decrease productivity, potentially fragment SAV beds, and lead to further erosion and degradation of the habitat (Uhrin and Holmquist 2003). Eriksson et al. (2004) found that boating activities can have direct and indirect impacts on SAV, including drag and tear on plant tissues resulting from increased wave-action, reduction in light availability caused by elevated turbidity and resuspension of bottom sediments, and altered habitat and substrate that causes plants to be uprooted and can inhibit recruitment. The disturbance of sediments and rooted vegetation decreases habitat suitability for fish and shellfish resources and can effect the spatial distribution and abundance of fauna (Nightingale and Simenstad 2001a; Uhrin and Holmquist 2003; Eriksson et al. 2004).

Resuspension of bottom sediments/turbidity

The degree of sediment resuspension and turbidity that is produced in the water column from vessel activity is complex but is generally dependent upon the wave energy and surge produced by the vessel, as well as the size of the sediment particles, the water depth, and the number of vessels passing through an area (Karaki and vanHoften 1975; Barr 1993). These activities typically increase turbidity and sedimentation on SAV and other sensitive benthic habitats (Klein 1997; Barr 1993; Nightingale and Simenstad 2001a; Eriksson et al. 2004). Studies investigating sedimentation impacts on eelgrass have found that experimental burial of 25% of the plant height can result in greater than 50% mortality (Mills and Fonseca 2003). Klein (1997) reported that turbidity generated by boats operating in shallow waters can exceed safe levels by up to 34-fold.

The resuspension of sediments can affect habitat suitability for fish and shellfish resources and effect the spatial distribution and abundance of fauna (Nightingale and Simenstad 2001a; Uhrin and Holmquist 2003; Eriksson et al. 2004). The egg and larval stages of marine and estuarine fish are generally highly sensitive to suspended sediment exposures (Wilber and Clark 2001), and juvenile fish may be susceptible to gill injury when suspended sediment levels are high (Klein 1997). Sedimentation and turbidity impacts associated with boating may be more pronounced in areas that contain shallow water habitat where the bottom is composed of fine sediments (Klein 1997).

Shoreline erosion

Wave energy caused by industrial and recreational shipping and transportation can have substantial impacts on aquatic shoreline and backwater areas which can eventually cause the loss and disturbance of shoreline habitats (Karaki and vanHoften 1975; Barr 1993; Klein 1997). Vessel wakes along frequently traveled routes can cause shoreline erosion, damage aquatic vegetation, disturb substrate, and increase turbidity. Wave energy and surge produced by vessels are dependent upon a number of factors, including the size and configuration of the vessel hull, the size of the vessel, and the speed of the vessel (Karaki and vanHoften 1975; Barr 1993). The degree of erosion on shorelines caused by vessels is complex, but it is generally dependent upon the wave energy and surge produced by the vessel and the slope of the shoreline, the type of sediment (e.g., clay, sand), and the type and amount of shoreline vegetation, as well as the characteristics of the water body (e.g., water depth and bottom topography) and distance between the vessel and shoreline (Karaki and vanHoften 1975; Barr 1993).

Contaminant spills and discharges

A variety of substances can be discharged or accidentally spilled into the aquatic environment, such as gray water (i.e., sink, laundry effluent), raw sewage, engine cooling water, fuel and oil, vessel exhaust, sloughed bottom paint, boat washdown water, and other vessel maintenance and repair materials that may degrade water quality and contaminate bottom sediments (Cardwell et al. 1980; Cardwell and Koons 1981; Krone et al. 1989; Waite et al. 1991; Hall and Anderson 1999; Hanson et al. 2003).

Industrial shipping and recreational boating can be sources of metals such as arsenic, cadmium, copper, lead, and mercury (Wilbur and Pentony 1999). Metals are known to have toxic effects on marine organisms. For example, 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 impairment of vertical migration for larvae at copper concentrations greater than 300 μg/L (Blaxter 1977). Copper may also bioaccumulate in bacteria and phytoplankton (Milliken and Lee 1990). Metals may enter the water through various vessel maintenance activities such as bottom washing, paint scraping, and application of antifouling paints (Amaral et al. 2005). For example, elevated copper concentrations in the vicinity of shipyards have been associated with vessel maintenance operations such as painting and scraping of boat hulls (Milliken and Lee 1990). Studies have shown a positive relationship between the number of recreational boats in a marina and the copper concentrations in the sediments of that marina (Warnken et al. 2004). Copper and an organotin, called tributyltin (TBT), are common active ingredients in antifouling paints (Milliken and Lee 1990). The use of TBT is primarily used for large industrial vessels to improve the hydrodynamic properties of ship's hulls and fuel consumption, while recreational vessels typically use copper-based antifouling paints because of restrictions introduced in the Organotin Antifouling Paint Control Act of 1988 (33 U.S.C. 2401), which bans its use on vessels less than 25 m in length (Milliken and Lee 1990; Hofer 1998).

Herbicides are also used in some antifouling paints to inhibit the colonization of algae and the growth of seaweeds on boat hulls and intake pipes (Readman et al. 1993). Similar to copper, the highest concentrations of herbicides in nearshore waters are associated with recreational marinas, which may be because of a higher frequency of use of these types of antifouling paints for pleasure boats compared to commercial vessels (Readman et al. 1993). The leaching of these chemicals into the marine environment could affect community structure and phytoplankton abundance (Readman et al. 1993).

Fuel and oil spills can affect animals directly or indirectly through the food chain. Fuel, oil, and some hydraulic fluids contain PAH which can cause acute and chronic toxicity in marine organisms (Neff 1985). Toxic effects of exposure to PAH have been identified in adult finfish at concentrations of 5-50 ppm and the larvae of aquatic species at concentrations of 0.1-1.0 ppm (Milliken and Lee 1990). Small, but chronic oil spills are a potential problem because residual oil can build up in sediments and affect living marine resources. Even though individual releases are small, they are also frequent and when combined they contribute nearly 85% of the total input of oil into aquatic habitats from human activities (ASMFC 2004). Incidental fuel spills involving small vessels are probably common events, but these spills typically involve small amounts of material and may not necessarily adversely affect fishery resources. Larger spills may have significant acute adverse affects, but these events are relatively rare and usually involve small geographic areas.

Outboard engines, as opposed to inboard engines that are generally used for larger, commercial vessels, are unique in that their exhaust gases cool rapidly and leave some hydrocarbon components condensed and in the water column rather than being released into the atmosphere (Moore and Stolpe 1995). Outboard engine pollution, particularly from two-cycle engines, can contribute to the concentrations of hydrocarbons in the water column and sediment (Milliken and Lee 1990). Two-cycle outboard engines accomplish fuel intake and exhaust in the same cycle and tend to release unburned fuel along with the exhaust gases. In addition, two-cycle engines mix lubricant oil with the fuel, so this oil is released into the water along with the unburned fuel. There are over 100 hydrocarbon compounds in gasoline, including additives to improve the efficiency of the fuel combustion (Milliken and Lee 1990). Once discharged into the water, petroleum hydrocarbons may remain suspended in the water column, concentrate on the surface, or settle to the bottom (Milliken and Lee 1990).

Any type of fuel or oil spill has the potential to cause impacts to organisms and habitats in the water column, on the bottom, and on the shoreline, but it is unknown to what extent these effects are individually or cumulatively significant. Effects on fish from low-level chronic exposure may increase embryo mortality, reduce growth, or alter migratory patterns (Heintz et al. 2000; Wertheimer et al. 2000). For more details on the impacts of oil or fuel spills, see the chapter on Energy-related Activities.

Gray water and sewage discharge from boats may impact water quality by increasing nutrient loading and biological oxygen demand of the local area and through the release of disease causing organisms and toxic substances (Thom and Shreffler 1996; Klein 1997). Positive correlations between boating activity levels and elevated levels of fecal coliform bacteria in nearshore coastal waters have been reported (Milliken and Lee 1990). Although the Clean Water Act (CWA) of 1972 makes it illegal to discharge untreated wastes into coastal waters and the Federal Water Pollution Control Act requires recreational boats be equipped with marine sanitation devices (MSDs), it is legal to discharge treated wastes, and illegal discharges of untreated waste may be common (Milliken and Lee 1990; Amaral et al. 2005). Despite these laws, many vessels may not be equipped with MSDs and on-shore pumpout stations are not common (Amaral et al. 2005). Impacts from vessel waste discharges may be more pronounced in small, poorly flushed waterways where pollutant concentrations can reach unusually high levels (Klein 1997).

Underwater noise

The noise generated by vessel operations is usually concentrated in ports, marinas, and heavily used shipping lanes or routes and may impact fish spawning, migration, and recruitment behaviors (Hildebrand 2004). Exposure to continuous noise may also create a shift in hearing thresholds for marine organisms resulting in hearing losses at certain frequency ranges (Jasny et al. 1999). Reducing vessel noise is a difficult task because of the economic incentives that encourage the expansion of commercial shipping and the lack of alternatives for efficient global transport of large and high tonnage material (Hildebrand 2004).

Small craft with high-speed engines and propellers (e.g., recreational boats with outboard engines) typically produce higher frequency noise than do larger vessels that generate substantial low-frequency noise because of their size and large, slow-speed engines and propellers (Kipple and

Gabriele 2004). A noise study of three size-classes of vessels (i.e., small, 17-30 feet; medium, 50-100 feet; and large, >100 feet) in Glacier Bay, AK, found that, on average, overall sound levels were higher for the larger vessel categories (Kipple and Gabriele 2004). However, vessel sound levels in this study were generally measured at vessel speeds less than 10 knots, and the investigators found increasing sound levels with greater vessel speed (Kipple and Gabriele 2004). Scholik and Yan (2002) reported significant elevation of the auditory threshold of the fathead minnow (*Pimephales promelas*), after exposure to noise from an idling 55 horsepower outboard motor. Furthermore, the frequencies of the noise from the outboard engine corresponded to the frequencies of the fish's auditory threshold shifts, specifically in this species' most sensitive hearing range (1.0-2.0 kHz).

Commercial shipping vessels are a major source of low frequency (5-500 Hz) noise in the marine environment and may be one of the most pervasive sources of anthropogenic ocean noise (Jasny et al. 1999; Stocker 2002; Hildebrand 2004). Low frequencies travel long distances in the marine environment, which is probably why these frequencies are also used by marine mammals for communication (Jasny et al. 1999). Ship noise is generated from the use of engines and other on-board mechanical devices such as pumps, cooling systems, and generators, as well as movement of water across the hull and propellers (Stocker 2002; Hildebrand 2004). These sounds are amplified and transferred to the water through the ship's hull (Stocker 2002). The size and frequency of use for commercial vessels traversing the ocean and nearshore waters may explain why they are considered a major source of noise impacts compared to the more numerous fishing and pleasure craft found in coastal waters (Hildebrand 2004).

There are several factors which influence sound attenuation in shallow coastal waters including temperature variations or thermoclines, bottom geography, and sediment composition. Vessel noise may reverberate or scatter off geological features and anthropogenic structures in the water (Stocker 2002).

Sonar is another source of anthropogenic noise attributed to vessel operation. It is used for various purposes such as depth sounding and fish finding and can vary in range depending on the use (15-200 kHz for commercial navigation, 1-20 kHz for other positioning and navigation, and 100-3,000 Hz for long range sonar) (Stocker 2002). Refer to the Global Effects and Other Impacts chapter of this report for more information on ocean noise.

Release of debris

As discussed in the Operation and Maintenance of Ports and Marinas section of this chapter, the release of solid waste in coastal waters is a considerable concern. Billions of pounds of debris are dumped into the oceans each year (Milliken and Lee 1990), and vessel traffic is a significant source of this waste because of accidental loss, routine practices of dumping waste, and illegal dumping activities (Cottingham 1988). Entanglement in or ingestion of this debris can cause fish, marine mammals, and sea birds to become impaired or incapacitated, leading to starvation, drowning, increased vulnerability to predators, and physical wounds (Milliken and Lee 1990). Marine debris can also cause direct physical damage to habitat features through smothering or physical disturbance.

Plastics are an especially persistent form of solid waste. Plastics tend to concentrate along coastal areas because they float on the surface and can be transported by ocean currents (Milliken and Lee 1990). Commercial fishing, merchant vessel, cruise ship, and recreational boats are major contributors to marine plastic debris (Cottingham 1988; Milliken and Lee 1990). Cottingham (1988) estimated that merchant vessels are the primary source of plastic refuse in New England. Refer to the Operation and Maintenance of Ports and Marinas section in this chapter for information on plastic debris and the Coastal Development chapter of this report for more information on general marine debris.

Abandoned and derelict vessels

Derelict or abandoned vessels can cause a variety of impacts to habitats and are public health and navigational hazards. Grounded vessels may physically damage and smother benthic habitats, create changes in wave energy and sedimentation patterns, and scatter debris across sensitive habitats (Precht et al. 2001; Zelo and Helton 2005). The potential impact footprint of a grounded vessel can be much larger than the vessel itself as vessels move or break up during storm events, which can scour bottom habitat, amplify impacts, and complicate removal (Zelo and Helton 2005). The physical impacts of a grounded vessel can be greater in shallow water since the wreck is more likely to be unstable and move, may break up more rapidly because of wave and current forces, and is more likely to need urgent removal because of navigation concerns which may lead to additional resource impacts (Michel and Helton 2003). Refer to the Offshore Dredging and Disposal Activities chapter of this report for information regarding intentional sinking of vessels for disposal and creation of artificial reefs.

The most common environmental threat of a derelict or abandoned vessel is the release of oil or other pollutants. These hazardous materials may be part of a vessel's cargo, fuel and oil related to vessel operations, or chemicals contained within the vessel's structure which may be released through decay and corrosion over time. Rusting vessel debris can also cause iron enrichment in enclosed areas, which has been associated with harmful algal blooms (Helton and Zelo 2003; Michel and Helton 2003).

The historical focus of laws regarding derelict or abandoned vessels was the protection of the property rights of shipowners and the recovery of cargo (Michel and Helton 2003). Existing federal laws and regulations do not provide clear authority or funding to any single agency for the removal of grounded or abandoned vessels that harm natural resources but which are not otherwise obstructing or threatening to obstruct navigation or threatening a pollution discharge (Helton and Zelo 2003). In many cases vessels are abandoned and are left to continually damage the marine environment because a responsible party cannot be identified or a funding source for removal cannot be secured (Zelo and Helton 2005). Physical impacts, in particular, can persist for decades when vessels are left in the marine environment, and in some cases simply removing a vessel is enough to allow natural recolonization of benthic organisms (Zelo and Helton 2005).

Removal of a derelict vessel will ensure that the vessel does not become a navigation hazard to other ships and that hazardous materials are not released during storms which can damage the wreckage further. It also ensures that abandoned vessels do not become illegal dumpsites for oil, industrial waste, and other hazardous materials, including munitions (Helton and Zelo 2003). Salvage and wreck removal activities can result in unintended habitat impacts. For example, fuel spillage may occur during salvage operations of a wrecked vessel. The potential for collateral impacts should be considered when planning a salvage operation (Michel and Helton 2003). Wrecks in shallow water are often removed and scuttled in deep water to prevent further damage to more vulnerable, nearshore benthic habitats and to avoid the risks involved in bringing an unstable vessel into port (Michel and Helton 2003).

Although many of the habitat impacts described above can be averted if derelict vessels are removed while still afloat, abandoned and neglected floating vessels can also create habitat impacts (Zelo and Helton 2005). These vessels may shade seagrass beds, scour subtrates with anchor chains, or release pollutants from decaying hull materials and paints (Sunda 1994; Negri et al. 2002; Smith et al. 2003; Zelo and Helton 2005).

Nonnative and invasive species

Nonnative species, some of which are invasive, have been introduced to coastal areas through industrial shipping and recreational boating (Omori et al. 1994; Wilbur and Pentony 1999; Hanson et al. 2003; Pertola et al. 2006). These introductions can be in the form of fouling organisms on the bottom of vessels as they are transported between water bodies or through the release of ballast water from large commercial vessels. Modern ships can carry 10 to 200 thousand tons of ballast water at a time and transport marine organisms across long distances and in relatively short time periods (Hofer 1998). This expeditious travel increases the risk that the organisms taken up in ballast water will be viable when introduced into a distant port or marina during deballasting (Wilbur and Pentony 1999). Pertola et al. (2006), in an investigation of dinoflagellates and other phytoplankton from the ballast tank sediments of ships at ports in the northeastern Baltic Sea, found a large assemblage of germinated dinoflagellate cysts in 90% of all ships and at all ports sampled. Ship traffic can transport, in large numbers, nonnative and invasive species of phytoplankton that can be harmful to native aquatic species (Pertola et al. 2006). The nonnative green algae (*Codium fragile*), is an example of a species that has invaded the northeastern US coast, the eastern Atlantic Ocean, Mediterranean Sea, and New Zealand and has displaced native species of *Codium* (Walker and Kendrick 1998; Tyrrell 2005). Shipping has been implicated as the major agent of spread of this species (Walker and Kendrick 1998), as well as of the zebra mussel (*Dreissena polymorpha*) (Strayer et al. 2004). This invasive species has been shown to have had an adverse effect on the populations of some native species of fish (e.g., *Alosa* spp.), as well as phytoplankton, zooplankton, aquatic vegetation, water chemistry, and zoobenthos (Strayer et al. 2004).

Introduced species can adversely impact habitat qualities and functions by altering the community structure, competing with native species, and introducing exotic diseases (Omori et al. 1994; Wilbur and Pentony 1999; Carlton 2001). Additional discussion of the effects of introduced species can be found in the chapters on Introduced/Nuisance Species and Aquaculture and Physical Effect: Water Intake and Discharge Facilities.

Conservation recommendations and best management practices for vessel operation and maintenance

1. Encourage marinas to participate in NOAA/US EPA's Coastal Nonpoint Program and the Clean Marina Initiative.
2. Ensure that commercial ships and port facilities have oil-spill response plans in place which improve response and recovery in the case of accidental spillage.
3. Ensure that commercial ships and or port facilities have adequate oil-spill response equipment accessible and clearly marked.
4. Use dispersants that remove oils from the environment rather than dispersants that simply move them from the surface to the ocean bottom.
5. Promote the use of oil-absorbing materials in the bilge areas of all boats with inboard engines.
6. Promote the use of fuel/air separators on air vents or tank stems of inboard fuel tanks to reduce the amount of fuel and oil spilled into surface waters during fueling of boats.
7. Encourage recreational boats to be equipped with marine sanitation devices (MSDs) to prevent untreated sewage to be pumped overboard.
8. Encourage ship designs that include technologies capable of reducing noise generated and transmitted to the water column, such as the use of muffling devices already required for land-based machinery that may help reduce the impacts of vessel noise.

9. The effects of proposed and existing vessel traffic and associated underwater noise should be assessed for potential impacts to sensitive areas such as migration routes and spawning areas for marine animals.
10. Exclude vessels or limit specific vessel activities such as high intensity, low-frequency sonar, to known sensitive marine areas if evidence indicates that these activities have a substantial adverse effect to marine organisms.
11. Promote education and signage on all vessels to encourage proper disposal of solid debris at sea.
12. Encourage the use of innovative cargo securing and stowing designs that may reduce solid debris in the marine environment from the transportation of commercial cargo.
13. Use appropriate equipment and techniques to salvage and remove grounded vessels and follow all necessary state and federal laws and regulations. If possible, avoid using the propulsion systems of salvage tugs that can cause propeller wash and scour the bottom. Instead, moor the tugs and use a ground tackle system to provide maneuvering and pull.
14. Minimize additional seafloor damage when a derelict vessel has to be dragged across the seafloor to deep water by following the same ingress path. Alternatively, identify the least sensitive, operationally feasible towpath. Dismantling derelict vessels in place when stranded close to shore may cause less environmental impact than dredging or dragging a vessel across an extensive shallow habitat.
15. Reduce the risk of a sudden release of the entire cargo when a submerged derelict vessel contains hazardous aqueous solutions that pose limited environmental risks, such as mild acids and bases, by allowing the release of the cargo under controlled conditions. The controlled release plan can include water-quality monitoring to validate the calculated dilution rates and plume distance assumptions. All applicable state and federal laws and regulations regarding the release of chemicals into the water should be followed.
16. Develop a contingency plan for uncontrolled releases during vessel salvage operations. The salvage plan should include a risk assessment to determine the most likely release scenarios and use the best practices of the industry.
17. Schedule nonemergency salvage operations while including environmental considerations to minimize potential impacts on natural resources. Environmental considerations include periods when few sensitive species are present, avoidance of critical reproductive periods, and weather patterns that influence the trajectory of potential releases during operations
18. Choose a scuttling site for a derelict vessel in a deep-water location in federal or Exclusive Economic Zone (EEZ) waters that does not contain any sensitive resources or geological hazards. Ensure that all proposed disposal of vessels in the open ocean adheres to state and federal guidance and regulations, including section 102(a) of the Marine Protection, Research, and Sanctuaries Act (Ocean Dumping Act), and under 40 CFR § 229.3 of the US EPA regulations. Refer to the Offshore Dredging and Disposal Activities chapter for additional recommendations and BMPs for the disposal of vessels.

Navigation Dredging

Introduction

Channel dredging is a ubiquitous and chronic maintenance activity associated with port and harbor operation and vessel activity (Barr 1987; NEFMC 1998). Navigational dredging occurs in rivers, estuaries, bays, and other areas where ports, harbors, and marinas are located (Messieh and El-Sabh 1988). The locations of these facilities often coincide with sensitive aquatic habitats that are vital for supporting fishery production (Newell et al. 1998).

For the purposes of navigation, dredging can be generally classified as either creating new or expanded waterways with greater profiles, depths, and scope or as maintenance of existing waterways for the purpose of maintaining established profiles, depths, and scope. Although the latter category represents the most common dredging scenario, new construction, or "improvement" dredging as it is sometimes called, has become increasingly common at larger ports and harbors throughout the United States. Several corresponding factors have likely led to greater need for navigational "improvements" and increases in the operating depths and the sizes of existing ports and harbors, including: (1) increased demand for marine cargo and transportation; (2) expansion of commercial fleets; (3) increased demand for larger capacity commercial and recreational vessels; and (4) increased urbanization and infrastructure development along the coast (Messieh et al. 1991; Wilbur and Pentony 1999; Nightingale and Simenstad 2001b). In particular, this demand for larger capacity commercial cargo vessels has led to an increased competition among the major coastal ports to provide facilities to accommodate these vessels. Improvement dredging may occur in areas that have not previously been subjected to heavy vessel traffic and dredging activities, such as new commercial marinas or the creation of a new channel or turning basin in an existing port or marina facility. Because improvement dredging is often conducted in areas that have been less affected by previous dredging and vessel activities, the impacts are generally more severe than the impacts associated with regular maintenance dredging activities unless the sediments involved in the maintenance dredging contain high levels of contaminants (Allen and Hardy 1980).

Maintenance dredging is generally required in most navigation channels and port and marina facilities because of the continuous deposition of sediments from freshwater runoff or littoral drift. Navigation channels require maintenance dredging to remove accumulated sediments, typically conducted on a temporal scale of one to ten years (Nightingale and Simenstad 2001b). Alterations in sedimentation patterns of estuaries resulting from increased coastal development and urbanization often increases the sediment influx and the frequency for maintaining existing channels and ports. Dredging for other purposes, such as aggregate mining for sand and gravel, conveyance of flood flows, material for beach nourishment, and removal of contaminated sediments or construction of subtidal confined disposal of contaminated sediments, may be done separately or in conjunction with navigation dredging (Nightingale and Simenstad 2001b). Refer to the Offshore Dredging and Disposal Activities chapter of this report for more information on offshore aggregate mining and to the Coastal Development chapter of this report contains information on the affects of beach nourishment and other coastal development activities.

There is a variety of methods and equipment used in navigation dredging, and a detailed explanation and assessment is beyond the scope of this report. However, one can categorize dredging activities as either using hydraulic or mechanical equipment. The type of equipment used for navigation dredging primarily depends on the nature of the sediments to be removed and the type of disposal required. Some of the factors that determine the equipment type used are the characteristics of the material to be dredged, the quantities of material to be dredged, the dredging depth, the distance to the disposal area, the physical environmental factors of the dredging and disposal area, the contamination level of sediments, the methods of disposal, the production (i.e., rate of material removed) required, and the availability of the dredge equipment (Nightingale and Simenstad 2001b).

Hydraulic dredging involves the use of water mixed with sediments that forms a slurry, which is pumped through a pipeline onto a barge or a hopper bin for off-site disposal. To increase the productivity of the dredging operation (i.e., maximizing the amount of solid material transported to the disposal site), some of the water in the sediment slurry may be allowed to overflow out of the hopper which can increase the turbidity in the surrounding water column. If the disposal site is relatively close to the dredge site, the slurry may be pumped through a pipeline directly to the disposal site (e.g., beach disposal).

Mechanical dredging typically involves the use of a clamshell dredge, which consists of a bucket of hinged steel that is suspended from a crane. The bucket, with its jaws open, is lowered to the bottom and as it is hoisted up, the jaws close and carry the sediments to the surface. The sediments are then placed in a separate barge for transport to a disposal site. Bucket dredges tend to increase the suspended sediment concentrations compared to hydraulic dredges because of the resuspension created as sediment spills through the tops and sides of the bucket when the bucket contacts the bottom, during withdrawal of the bucket through the water column, and when it breaks the water's surface (Nightingale and Simenstad 2001b). Closed or "environmental" buckets are designed to reduce the sediment spill from the bucket by incorporating modifications such as rubber seals or overlapping plates and are often used in projects involving contaminated sediments.

The location and method of disposal for dredged material depends on the suitability of the material determined through chemical, and often, biological analyses conducted prior to the dredging project. Generally, sediments determined to be unacceptable for open water disposal are placed in confined disposal facilities or contained aquatic disposal sites and capped with uncontaminated sediments. Sediments that are determined to be uncontaminated may be placed in open-water disposal sites or used for beneficial uses. Beneficial uses are intended to provide environmental or other benefits to the human environment, such as shoreline stabilization and erosion control, habitat restoration/enhancement, beach nourishment, capping contaminated sediments, parks and recreation, agriculture, strip mining reclamation and landfill cover, and construction and industrial uses (Nightingale and Simenstad 2001b). Open water disposal sites can be either predominantly nondispersive (i.e., material is intended to remain at the disposal site) or dispersive (i.e., material is intended to be transported from the disposal site by currents and/or wave action (Nightingale and Simenstad 2001b). The potential for environmental impacts is dependent upon the type of disposal operation used, the physical characteristics of the material, and the hydrodynamics of the disposal site. Refer to the chapter on Offshore Dredging and Disposal Activities for more detailed information on dredge material disposal.

Dredging to deepen or maintain ports, marinas, and navigational channels involves a number of environmental effects to fishery habitats, including the direct removal or burial of demersal and benthic organisms and aquatic vegetation, alteration of physical habitat features, the disturbance of bottom sediments (resulting in increased turbidity), contaminant releases in the water column, light attenuation, releases of oxygen consuming substances and nutrients, entrainment of living organisms in dredge equipment, noise disturbances, and the alteration of hydrologic and temperature regimes. Dredging is often accompanied by a significant decrease in the abundance, diversity, and biomass of benthic organisms in the affected area and an overall reduction in the aquatic productivity of the area (Allen and Hardy 1980; Newell et al. 1998). The rate of recovery of the benthic community is dependent upon an array of environmental variables which reflect interactions between sediment particle mobility at the sediment-water interface and complex associations of chemical and biological factors operating over long time periods (Newell et al. 1998).

Loss or conversion of benthic habitat and substrate

Alterations in bathymetry, benthic habitat features, and substrate types caused by navigational dredging activities may have long-term effects on the functions of estuarine and other aquatic environments. The effects of an individual project are proportional to the scale and time required for a project to be completed, with small-scale and short-term dredging activities having less impact on benthic communities than long-term and large-scale dredging projects (Nightingale and Simenstad 2001b). Dredging can have cumulative effects on benthic communities, depending upon the dredging interval, the scale of the dredging activities, and the ability of the environment to recover from the impacts. The new exposed substrate in a dredged area may be composed of material containing more fine sediments than before the dredging, which can reduce the recolonization and productivity of the benthos and the species that prey upon them.

The impacts to benthic communities vary greatly with the type of sediment, the degree of disturbance to the substrate, the intrinsic rate of reproduction of the species, and the potential for recruitment of adults, juveniles, eggs, and larvae (Newell et al. 1998). Following a dredging event, sediments may be nearly devoid of benthic infauna, and those that are the first to recolonize are typically opportunistic species which may have less nutritional value for consumers (Allen and Hardy 1980; Newell et al. 1998).

In general, dredging can be expected to result in a 30-70% decrease in the benthic species diversity and 40-95% reduction in number of individuals and biomass (Newell et al. 1998). Recovery of the benthic community is generally defined as the establishment of a successional community which progresses towards a community that is similar in species composition, population density, and biomass to that previously present or at nonimpacted reference sites (Newell et al. 1998). The factors which influence the recolonization of disturbed substrates by benthic infauna are complex, but the suitability of the postdredging sediments for benthic organisms and the availability of adjacent, undisturbed communities which can provide a recruitment source are important (Barr 1987; ICES 1992). Rates of benthic infauna recovery for disturbed habitats may also depend upon the type of habitat being affected and the frequency of natural and anthropogenic disturbances. Benthic infauna recovery rates may be less than one year for some fine-grained mud and clay deposits, where a frequent disturbance regime is common, while gravel and sand substrates, which typically experience more stability, may take many years to recover (Newell et al. 1998). Post-dredging recovery in cold waters at high latitudes may require additional time because these benthic communities can be comprised of large, slow-growing species (Newell et al. 1998).

Loss of submerged aquatic vegetation

Submerged aquatic vegetation provides food and shelter for many commercially and recreationally important species, attenuates wave and current energy, and plays an important role in the chemical and physical cycles of coastal habitats (Thayer et al. 1997). The loss of vegetated shallows results in a reduction in important rearing and refugia functions utilized by migrating and resident species. Seagrass beds are more difficult to delineate and map than some other subtidal habitats because of their spatial and temporal dynamic nature, making these habitats more vulnerable to being inadvertently dredged (Thayer et al. 1997; Deegan and Buchsbaum 2005). Dredging causes both direct and indirect impacts to SAV. The physical removal of plants through dredging is a direct impact, while the reduction in light penetration and burial or smothering that is a result of the turbidity plumes and sedimentation created by the dredge are indirect impacts (Deegan and Buchsbaum 2005). While SAV may regrow in a dredged area if the exposure to excessive suspended sediments is not protracted and most of the accumulated sediments are removed by currents and tides after dredging ceases (Wilber et al. 2005), the recolonization by SAV may be limited if the bottom sediments are destabilized or the composition of the bottom sediments is altered (Thayer et al. 1997). Even when bottom sediments are stabilized and are conducive to SAV growth, channel deepening may result in the area having inadequate light regimes necessary for the recolonization of SAV (Barr 1987).

Dredge and fill operations require a permit review process which is regulated by state and federal agencies. Advancement in understanding the physical impacts of dredging on SAV and recognition of the ecological significance of these habitats has allowed special consideration for SAV beds during the permit review process. Most reviewing agencies discourage dredging activities in or near SAV beds as well as in areas that have been historically known to have SAV and areas that are potential habitats for SAV recruitment (Orth et al. 2002).

While the physical disturbance to SAV beds from dredge activities may have significant localized effects, water quality problems such as eutrophication, pollution and sedimentation have resulted in large-scale declines to SAV in some areas of the northeastern US coast (Goldsborough 1997; Deegan and Buchsbaum 2005; Wilber et al. 2005). The small, localized disturbance of SAV associated with dredging may be viewed as a significant impact in the context of diminished regional health and distribution resulting from stressors such as poor water quality and cumulative effects such as dredging, boating (propeller scour), and shoreline alteration (Goldsborough 1997; Thayer et al. 1997; Deegan and Buchsbaum 2005). The environmental effects of excess nutrients and sediments are the most common and significant causes of SAV decline worldwide (Orth et al. 2006).

Loss of intertidal habitat and wetlands

Intertidal habitats (e.g., mud and sand flats) and wetlands (e.g., salt marsh) are valuable coastal habitats which support high densities and diversities of biota by supporting biological functions such as breeding, juvenile growth, feeding, predator avoidance, and migration (Nightingale and Simenstad 2001b). These valuable habitats are also some of the most vulnerable to alterations through coastal development, urbanization, and the expansion of ports and marinas.

The loss of intertidal habitat and the deepening of subtidal habitat during dredging for marina development and for navigation can alter or eliminate the plant and animal assemblages associated with these habitats, including SAV and shellfish beds (Nightingale and Simenstad 2001b; MacKenzie 2007). Dredging in intertidal habitats can alter the tidal flow, currents, and tidal mixing regimes of the dredged area as well as other aquatic habitats in the vicinity, leading to changes in the environmental parameters necessary for successful nursery habitats (Barr 1987). Dredging in tidal wetlands can also encourage the spread of nonnative invasive organisms by removing or disturbing the native biota and altering the physical and chemical properties of the habitat (Hanson et al. 2003; Tyrrell 2005).

Navigational dredging converts shallow subtidal or intertidal habitats into deeper water environments through the removal of sediments (Nightingale and Simenstad 2001b, Deegan and Buchsbaum 2005). The historical use of dredged materials was to infill wetland, salt marshes, and tidal flats in order to create more usable land. The Boston Harbor, MA, area is a prime example of this historical trend, where thousands of acres of salt marsh and intertidal wetlands have been filled over time (Deegan and Buchsbaum 2005). Filling wetlands eliminates the biological, chemical, and physical functions of intertidal habitat such as flood control, nutrient filter or sink, and nursery habitat. Although direct dredging and filling within intertidal wetlands are relatively rare in recent times, the lost functions and values of intertidal wetlands and the connectivity between upland and subtidal habitat is difficult and costly to create and restore (Nightingale and Simenstad 2001b).

Underwater noise

Fish can detect and respond to sounds for many life history requirements, including locating prey and avoiding predation, spawning, and various social interactions (Myrberg 1972; Myrberg and Riggio 1985; Kalmijn 1988). The noise generated by pumps, cranes, and by the mechanical action of the dredge itself has the ability to alter the natural behavior of fish and other aquatic organisms. Feist et al. (1996) reported that pile-driving operations had an affect on the distribution and behavior of juvenile pink salmon (*Oncorhynchus gorbuscha*) and chum salmon (*Oncorhynchus keta*). Fish may leave an area for more suitable spawning grounds or may avoid a natural migration path because of noise disturbances.

The noise levels and frequencies produced from dredging depend on the type of dredging equipment being used, the depth and thermal variations in the surrounding water, and the topography and composition of the surrounding sea floor (Nightingale and Simenstad 2001b; Stocker 2002). However, dredging activities from both mechanical and hydraulic dredges produce underwater sounds that are strongest at low frequencies and because of rapid attenuation of low frequencies in shallow water, dredge noise normally is undetectable underwater at ranges beyond 20-25 km (Richardson et al. 1995). Although the noise levels from large ships may exceed those from dredging, single ships usually do not produce strong noise in one area for a prolonged period of time (Richardson et al. 1995). The noise created during dredging can produce continuous noise impacts for extended periods of time (Nightingale and Simenstad 2001b).

Siltation, sedimentation, and turbidity

Dredging degrades habitat quality through the resuspension of sediments which creates turbid conditions and can release contaminants into the water column, in addition to impacting benthic organisms and habitat through sedimentation. Turbidity plumes ranging in the hundreds to thousands mg/L are created and can be transported with tidal currents to sensitive resource areas. Alterations in bottom sediments, bottom topography, and altered circulation and sedimentation patterns related to dredge activities can lead to shoaling and sediment deposition on benthic resources such as spawning grounds, SAV, and shellfish beds (Wilber et al. 2005; MacKenzie 2007). Early life history stages (eggs, larvae, and juveniles) and sessile organisms are the most sensitive to sedimentation impacts (Barr 1987; Wilber et al. 2005). Some estuarine and coastal habitats are prone to natural sediment loads and sediment resuspension because of the relatively dynamic nature of the ecosystems; therefore, most organisms adapted to these environments have tolerance to some level of suspended sediments and sedimentation (Nightingale and Simenstad 2001b).

The reconfiguration of sediment type and the removal of biogenic structure during dredging may decrease the stability of the bottom and increase the ambient turbidity levels (Messieh et al. 1991). This increased turbidity and sedimentation can reduce the light penetration of the water column which then can adversely affect SAV and reduce primary productivity (Cloern 1987; Dennison 1987; Wilbur and Pentony 1999; Mills and Fonseca 2003; Wilbur et al. 2005). The combination of decreased photosynthesis and the interaction of the suspended material with dissolved oxygen in the water may result in short-term oxygen depletion (Nightingale and Simenstad 2001b).

If suspended sediment loads remain high, fish may experience respiratory distress and reduced feeding ability because of sight limitations, while filter feeders may suffer a reduction in growth and survival (Messieh et al.1991; Barr 1993; Benfield and Minello 1996; Nightingale and Simenstad 2001b). Prolonged exposure to suspended sediments can cause gill irritation, increased mucus production, and decreased oxygen transfer in fish (Nightingale and Simenstad 2001b; Wilber et al. 2005). Reduced dissolved oxygen concentrations and increased water temperatures may be cumulative stressors that exacerbate the effects of respiratory distress on fish from extended exposure to suspended sediments (Nightingale and Simenstad 2001b). In addition, mobile species may leave an area for more suitable feeding or spawning grounds, or avoid migration paths because of turbidity plumes created during navigational dredging.

Increased turbidity and sedimentation may also bury benthic organisms and demersal fish eggs. The depth of burial and the density of the substrate may limit the natural escape response of some organisms that are capable of migrating vertically through the substrate (Barr 1987; Wilber et al. 2005). In addition, anoxic conditions in the disturbed sediments may decrease the ability of benthic organisms to escape burial (Barr 1987). Short-term burial, where sediment deposits are promptly removed by tides or storm events, may have minimal effects on some species (Wilber et al. 2005). However, even thin layers of fine sediment have been documented to decrease gas exchange in fish eggs and adversely affect the settlement and recruitment of bivalve larvae (Wilber et al. 2005). An in-situ experiment with winter flounder (*Pseudopleuronectes americanus*) eggs exposed to sediment deposition from a navigational dredging project found a slightly lower larval survival rate compared to control sites, but the differences were not statistically significant (Klein-MacPhee et al. 2004). However, the viability of the larvae in this experiment was not monitored beyond burial escapement. Similarly, laboratory experiments with winter flounder eggs buried to various depths (i.e., control, <0.5 mm, and up to 2 mm) indicated a decreased hatch success and delayed hatch with increasing depth; but differences were not statistically significant (Berry et al. 2004). The same study also exposed winter flounder eggs to both clean, fine-grained sediment and highly contaminated, fine-grained sediment at various depths from 0.5-6.0 mm. The investigators found that eggs buried to depths of 4 mm with clean sediments did not hatch, while eggs buried to depths of 3 mm with contaminated sediments had little or no hatching success (Berry et al. 2004). Although there are clearly adverse effects to sessile benthic organisms and life stages from sedimentation from dredging activities, additional investigations are needed to assess lethal and sublethal thresholds for more species and under different sediment types and quality. In addition, better understanding about the relationship between natural and anthropogenic sources of suspended sediments and population-level effects is needed.

The use of certain types of dredging equipment can result in greatly elevated levels of fine-grained particles in the water column. Mechanical dredging techniques such as clam shell or bucket dredges usually increase suspended sediments at the dredge site more than hydraulic dredge techniques such as hopper or cutterheads, unless the sediment and water mixture (slurry) removed during hydraulic dredging is allowed to overflow from the barge or hopper and into the water column, a technique often used to reduce the number of barge trips required (Wilber and Clarke 2001). Mechanical dredges are most commonly used for smaller projects or in locations requiring maneuverability such as close proximity to docks and piers or in rocky sediments (Wilber et al. 2005), although small hydraulic dredges can be used to reduce suspended sediment concentrations in the dredging area and minimize impacts on adjacent benthic habitats, such as SAV or shellfish beds.

Seasonal or time-of-year (TOY) restrictions to dredging activities are used to constrain the detrimental affects of dredging to a timeframe that minimizes impacts during sensitive periods in the life history of organisms, such as spawning, egg development, and migration (Nightingale and Simenstad 2001b; Wilber et al. 2005). Segregating dredging impacts by life history stages provides a means for evaluating how different impacts relate to specific organisms and life history strategies (Nightingale and Simenstad 2001b). The application of TOY restrictions should be based upon the geographic location, species and life history stages present, and the nature and scope of the dredging project. Because the employment of TOY restrictions may have some negative effects, such as extending the overall length of time required for dredging and disposal, increasing the impacts on less economically valuable or poorly studied species, and increasing the economic costs of a project, the benefits of TOY restrictions should be evaluated for each individual dredging project (Wilber et al. 2005; Nightingale and Simenstad 2001b).

Contaminant release and source exposure

Contaminated sediments are a concern because of the risk of transport of the contaminants and the exposure to aquatic organism and humans through bioaccumulation and biomagnification (Nightingale and Simenstad 2001b). Navigation dredging can create deep channels where currents are reduced and fine sediments may be trapped. Nutrients and contaminants can bind to fine particles such as those that may settle in these deep channels (Newell et al. 1998; Messiah et al. 1991). Dredging and disposal causes resuspension of the sediments into the water column and the contaminants that may be associated with the sediment particles. The disturbance of bottom sediments during dredging can release metals (e.g., lead, zinc, mercury, cadmium, copper), hydrocarbons (e.g., PAH), hydrophobic organics (e.g., dioxins), pesticides, pathogens, and nutrients into the water column and allow these substances to become biologically available either in the water column or through trophic transfer (Wilbur and Pentony 1999; USEPA 2000; Nightingale and Simenstad 2001b). Generally, the resuspension of contaminated sediments can be reduced by avoiding dredging in areas containing fine sediments. In addition, the biological and/or chemical testing requirements under the Marine Protection, Research, and Sanctuaries Act and the Clean Water Act are designed to minimize adverse effects of dredge material disposal on the environment. For additional information regarding the affects of contaminants associated with resuspended sediments, refer to the chapters on Offshore Dredging and Disposal Activities and Chemical Affects: Water Discharge Facilities in this report.

Release of nutrients/eutrophication

Dredging can degrade water quality through resuspension of sediments and the release of nutrients and other contaminants into the water column. Nutrients and contaminants may adhere to these fine particles (Newell et al. 1998; Messieh et al. 1991). The resuspension of this material creates turbid conditions and decreases photosynthesis. The combination of decreased photosynthesis and the release of organic material with high biological oxygen demand can result in short-term oxygen depletion to aquatic resources (Nightingale and Simenstad 2001b). Long-term anoxia can occur if highly organic sediments are dredged or discharged into estuaries, particularly in enclosed or confined bodies of water. The loss of SAV is linked to poor water quality from increased turbidity and nutrient loading (Deegan and Buchsbaum 2005; Wilber et al. 2005).

Entrainment and impingement

Entrainment is the direct uptake of aquatic organisms by the suction field created by hydraulic dredges. Benthic infauna are particularly vulnerable to entrainment by dredging, although some mobile epibenthic and demersal species such as shrimp, crabs, and fish can be susceptible to entrainment as well (Nightingale and Simenstad 2001b). Elicit avoidance responses to suction dredge entrainment has been reported for some demersal and pelagic mobile species (Larson and Moehl 1990; McGraw and Armstrong 1990). The susceptibility to entrainment for some pelagic species may be related to the degree of waterway constriction in the area of the dredging, which makes it more difficult for fish to avoid the dredge operation (Larson and Moehl 1990; McGraw and Armstrong 1990).

Altered tidal, current, and hydrologic regimes

Large channel deepening projects can potentially alter ecological relationships through a change in freshwater inflow, tidal circulation, estuarine flushing, and freshwater and saltwater mixing (Nightingale and Simenstad 2001b). Dredging may also modify longshore current patterns by altering the direction or velocity of water flow from adjacent estuaries. These changes in water circulation are often accompanied by changes in the transport of sediments and siltation rates resulting in alteration of local habitats used for spawning and feeding (Messieh et al. 1991).

Altered circulation patterns around dredged areas can also lead to changes in sediment composition and deposition and in the stability of the seabed. The deep channels created during navigational dredging may experience reduced current flow that allows the area to become a sink for fine particles as they settle out of the water column or slump from the channel walls (Newell et al. 1998). In some cases this may change the sediment composition from sand or shell substrate to a substrate consisting of fine particles which flocculate easily and are subject to resuspension by waves and currents (Messieh et al. 1991). This destabilization of the seabed can lead to changes in sedimentation rates and a reduction in benthic resources, such as shellfish beds and SAV (Wilber et al. 2005). In addition, changes in substrate type can smother demersal eggs, affect larval settlement, and increase predation on juveniles adapted to coarser bottom substrates (Messieh et al. 1991; Wilber et al. 2005).

Navigational dredging can remove natural benthic habitat features, such as shoals, sand bars, and other natural sediment deposits. The removal of such features can alter the water depth, change current direction or velocity, modify sedimentation patterns, alter wave action, and create bottom scour or shoreline erosion (Barr 1987). Channel dredging can alter the estuarine hydrology and the mixing zone between fresh and salt water, leading to accelerated upland run-off, lowered freshwater aquifers, and greater saltwater intrusion into aquifers, as well as reduce the buffering capabilities of wetlands and shallow water habitats (Barr 1987; Nightingale and Simenstad 2001b).

Navigational channels that are substantially deeper than surrounding areas can become anoxic or hypoxic as natural mixing is decreased and detrital material settles out of the water column and accumulates in the channels. This concentration of anoxic or hypoxic water can stress nearshore biota when mixing occurs from a storm event (Allen and Hardy 1980). The potential for anoxic conditions can be reduced in areas that experience strong currents or wave energy, and sediments are more mobile (Barr 1987; Newell et al. 1998).

Altered temperature regimes

Channel and port dredging can alter bottom topography, increase water depths, and change circulation patterns in the dredged area, which may increase stratification of the water column and reduce vertical mixing. This thermal layering of water may create anoxic or hypoxic conditions for benthic habitats. Deepened or new navigation channels may create deep and poorly flushed areas that experience reduced light penetration and water temperatures. Temperature influences biochemical processes and deep channels may create zones of poor productivity that can serve as barriers to migration for benthic and demersal species and effectively fragment estuarine habitats.

Conservation recommendations and best management practices for navigational dredging

1. Avoid new dredging to the maximum extent practicable. Activities that would likely require dredging (such as placement of piers, docks, marinas, etc.) should instead be located in deep water or designed to alleviate the need for maintenance dredging.
2. Reduce the area and volume of material to be dredged to the maximum extent practicable.

3. Ensure that the volumes of dredge material are appropriately considered and that the identified disposal sites are adequate in containing the material. For example, the volume of material removed for the allowable over-depth dredging (usually 2 feet below the authorized or target depth) should be included in the disposal volume calculations.
4. Ensure that areas proposed for dredging are necessary in order to maintain the necessary and authorized target depths of the channel. Recent bathymetric surveys should be reviewed to evaluate the existing depths of the area proposed for dredging. Areas within the proposed dredge area that are at or deeper than the target depths should be avoided, whenever practicable.
5. Identify sources of erosion in the watershed that may be contributing to excessive sedimentation and the need for regular maintenance dredging activities. Implement appropriate management techniques to ensure that actions are taken to curtail those causes.
6. Use settling basins to act as sediment traps to prevent accretion of sediments in the navigational channel, when appropriate. This reduces the need for frequent maintenance dredging of the entire channel.
7. Consider the effects of increased boat traffic to an area when assessing a new dredging project or expanding existing channels. Increases in the speed, size, and density of boat traffic in an area may require increased frequency of maintenance dredging and produce a number of secondary impacts, such as shoreline erosion, sedimentation, and turbidity.
8. Identify the user group during the planning process to ensure that the dredging project meets the basic needs of the target user without exceeding an appropriate size and scope, or encouraging inappropriate use.
9. Consider time-of-year dredging restrictions, which may reduce or avoid impacts to sensitive life history stages, such as migration, spawning, or egg and young-of-year development. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
10. Avoid projects that involve dredging intertidal and wetland habitat.
11. Avoid dredging in areas with SAV, areas which historically supported SAV, and areas which are potential habitat for recolonization by SAV.
12. Conduct both historic surveys of the area and predredge surveys because of the spatial and temporal dynamic nature of SAV beds.
13. Avoid dredging in areas supporting shellfish beds.
14. Consider beneficial uses for uncontaminated sediments when practicable and feasible. Priority should be given to beneficial uses of material that contributes to habitat restoration and enhancement, landscape ecology approach, and includes pre- and post-disposal surveys.
15. Avoid beneficial use projects that impose unnatural habitats and features and involve habitat trade-offs (substituting one habitat type for another).
16. Ensure that sediments are tested for contaminants and meet or exceed US EPA requirements and standards prior to dredging and disposal.
17. Assess cumulative impacts for current activities in the vicinity of a proposed dredging project, as well as for activities in the past and foreseeable future.
18. Ensure that bankward slopes of the dredged area are slanted to acceptable side slopes (e.g., 3:1 ratio) to ensure that sloughing of the channel side slopes does not occur.
19. Avoid placing pipelines and accessory equipment used in conjunction with dredging operations close to algae beds, eelgrass beds, estuarine/salt marshes, and other high value habitat areas.
20. Use silt curtains in some locations to reduce impacts of suspended sediments on adjacent benthic resources.
21. Avoid dredging in fine sediments when possible to reduce turbidity plumes and the release of nutrients and contaminants which tend to bind to fine particles.

22. Include information on control sites and predredging sampling for comparison and monitoring of impacts in environmental assessments for dredging projects.
23. Ensure that disposal sites are properly sited (i.e., avoid sensitive resources and habitats) and are appropriate for the type of dredge material proposed for disposal.
24. Ensure that disposal sites are being properly managed (e.g., disposal site marking buoys, inspectors, the use of sediment capping and dredge sequencing) and monitored (e.g., chemical and toxicity testing, benthic recovery) to minimize impacts associated with dredge material.

References for Marine Transportation

Able KW, Manderson JP, Studholme AL. 1999. Habitat quality for shallow water fishes in an urban estuary: the effects of man-made structures on growth. Marine Ecology Progress Series 187:227-35.

Allen KO, Hardy JW. 1980. Impacts of navigational dredging on fish and wildlife a literature review. Washington (DC): Biological Services Program, USFWS.

Amaral M, Lee V, Rhodes J. 2005. Environmental guide for marinas: controlling nonpoint source and storm water pollution in Rhode Island: best management practices for marinas. [Internet]. Narragansett (RI): University of Rhode Island, Coastal Resources Center, Rhode Island Sea Grant. [cited 2007 Dec 28]. Available from: http://seagrant.gso.uri.edu/BMP/BMP.html.

[ASMFC] Atlantic States Marine Fisheries Commission. 2004. NRC report outlines the impacts of oil on marine fish habitat. Habitat Hotline Atlantic 11(3):1-5.

Barr BW. 1987. The dredging handbook: a primer for dredging in the coastal zone of Massachusetts. Boston (MA): Massachusetts Coastal Zone Management. Publication No. 14, 870-181-500-6-87-CR.

Barr BW. 1993. Environmental impacts of small boat navigation: vessel/sediment interactions and management implications. In: Magoon OT, editor. Coastal Zone '93: proceedings of the eighth Symposium on Coastal and Ocean Management; 1993 Jul 19-23; New Orleans, LA. American Shore and Beach Preservation Association. p 1756-70.

Benfield MC, Minello TJ. 1996. Relative effects of turbidity and light intensity on reactive distance and feeding of an estuarine fish. Environmental Biology of Fishes 46:211-6.

Berry WJ, Hinchey EK, Rubinstein NI, Klein-MacPhee G. 2004. Winter flounder, *Pseudopleuronectes americanus*, hatching success as a function of burial depth in the laboratory. Ninth flatfish biology conference- poster presentation; 2004 Dec 1-2; Westbrook, CT. Woods Hole (MA): Northeast Fisheries Science Center Reference Document 04-13.

Blaxter JHS. 1977. The effect of copper on the eggs and larvae of plaice and herring. Journal of the Marine Biological Association of the United Kingdom 57:849-59.

Burdick DM, Short FT. 1999. The effects of boat docks on eelgrass beds in coastal waters of Massachusetts. Environmental Management 23(2):231-40.

Caltrans. 2001. San Francisco-Oakland Bay Bridge East Span seismic safety project, pile installation demonstration project, fisheries impact assessment. Sacramento (CA): California Department of Transportation. 59 p.

Cardwell RD, Koons RR. 1981. Biological considerations for the siting and design of marinas and affiliated structures in Puget Sound. Olympia (WA): Washington Department of Fisheries. Technical Report No. 60. 31 p.

Cardwell RD, Olsen SJ, Carr MI, Sanborn EW. 1980. Biotic, water quality and hydrologic characteristics of skyline marina in 1978. Olympia (WA): Washington Department of Fisheries. Technical Report No. 54. 103 p.

Carlson TJ, Ploskey G, Johnson RL, Mueller RP, Weiland MA, Johnson PN. 2001. Observations of the behavior and distribution of fish in relation to the Columbia River navigation channel and channel maintenance activities. Richland (WA): US Army Corps of Engineers, Portland District. 35 p + appendices.

Carlton JT. 2001. Introduced species in U.S. coastal waters: Environmental impacts and management priorities. Arlington (VA): Pew Oceans Commission. 28 p.

Chapman MG. 2003. Paucity of mobile species on constructed seawalls: effects of urbanization on biodiversity. Marine Ecology Progress Series 264:21-9.

Cloern JE. 1987. Turbidity as a control on phytoplankton biomass and productivity in estuaries. Continental Shelf Research 7(11-12):1367-81.

Costa JE, Howes BL, Giblin AE, Valiela I. 1992. Monitoring nitrogen and indicators of nitrogen loading to support management action in Buzzards Bay. In: McKenzie DH, Hyatt DE, McDonald VJ, editors. Ecological indicators. New York (NY): Elsevier Applied Science. p 499-531.

Cottingham D. 1988. Persistent marine debris: challenge and response: the federal perspective. Fairbanks (AK): Alaska Sea Grant. Report No. 88-1.

Dahl TE. 2006. Status and trends of wetlands in the conterminous United States 1998 to 2004. [Internet]. Washington (DC): US Department of the Interior; Fish and Wildlife Service. [cited 2008 Jul 21]. 112 p. Available from: http://wetlandsfws.er.usgs.gov/status_trends/National_Reports/trends_2005_report.pdf.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Dennison WC. 1987. Effects of light on seagrass photosynthesis, growth and depth distribution. Aquatic Botany 27(1):15-26.

Dolat SW. 1997. Acoustic measurements during the Baldwin Bridge demolition. Waterford (CT): Prepared for White Oak Construction by Sonalysts. 34 p + appendices.

Eriksson BK, Sandstrom A, Iscus M, Schreiber H, Karas P. 2004. Effects of boating activities on aquatic vegetation in the Stockholm archipelago, Baltic Sea. Estuarine, Coastal and Shelf Science 61(2):339-49.

Feist BE, Anderson JJ, Miyamoto R. 1996. Potential impacts of pile driving on juvenile pink (*Oncorhynchus gorbuscha*) and chum (*O. keta*) salmon behavior and distribution. Seattle (WA): Fisheries Research Institute-University of Washington. FRI-UW-9603. 58 p.

Fresh KL, Williams B, Penttila D. 1995. Overwater structures and impacts on eelgrass (*Zostera marina*) in Puget Sound, Washington. In: Puget Sound Research '95 Proceedings; 1995 Jan 12-14; Bellevue, WA. Olympia (WA): Puget Sound Water Quality Authority. p 537-43.

Fresh KL, Williams BW, Wyllie-Echeverria S, Wyllie-Echeverria T. 2001. Mitigating impacts of overwater floats on eelgrass *Zostera marina* L. in Puget Sound, Washington. In: Proceedings of Puget Sound Research 2001-The fifth Puget Sound Research Conference; 2001 Feb 12-14; Bellevue, WA. Olympia (WA): Puget Sound Action Team.

Goldsborough WJ. 1997. Human impacts on SAV-a Chesapeake Bay case study. In: Stephan CD, Bigford TE, editors. Atlantic coastal submerged aquatic vegetation: a review of its ecological role, anthropogenic impacts, state regulation, and value to Atlantic coastal fisheries. Washington (DC): ASMFC Habitat Management Series #1. p 68 + appendices.

Gould E, Clark PE, Thurberg FP. 1994. Pollutant effects on dermersal fishes. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 30-41.

Haas MA, Simenstad CA Jr, Cordell JR, Beauchamp DA, Miller BS. 2002. Effects of large overwater structures on epibenthic juvenile salmon prey assemblages in Puget Sound, WA. [Internet]. Seattle (WA): Washington State Transportation Center (TRAC), University of Washington, and Washington State Department of Transportation. Final Research Report WA-RD 550.1. [cited 2008 Jul 9]. 114 p. Available from: http://depts.washington.edu/trac/bulkdisk/pdf/550.1.pdf.

Hall LW Jr, Anderson RD. 1999. A deterministic ecological risk assessment for copper in European saltwater environments. Marine Pollution Bulletin 38(3):207-18.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Hastings K, Hesp P, Kendrick G. 1995. Seagrass loss associated with boat moorings at Rottnest Island, Western Australia. Ocean & Coastal Management 26:225-46.

Hawkins AD. 1986. Underwater sound and fish behaviour. In: Pitcher TJ, editor. The behavior of teleost fishes. Baltimore (MD): The Johns Hopkins University Press. p 114-51.

Heintz RA, Rice SD, Wertheimer AC, Bradshaw RF, Thrower FP, Joyce JE, Short JW. 2000. Delayed effects on growth and marine survival of pink salmon *Oncorhynchus gorbuscha* after exposure to crude oil during embryonic development. Marine Ecology Progress Series 208:205-16.

Helton D, Zelo I. 2003. Developing information and support necessary to prioritize and support removal of abandoned vessels impacting coral resources. In: Proceedings of the 13th

Biennial Coastal Zone Conference; 2003 Jul 13-17; Baltimore, MD. Charleston (SC): NOAA Coastal Services Center.

Hildebrand J. 2004. Sources of anthropogenic sound in the marine environment. [Internet]. In: Vos E, Reeves RR, editors. Report of an international workshop: policy on sound and marine mammals; 2004 Sep 28-30; London, England. Bethesda (MD): Marine Mammal Commission. [cited 2008 Jul 21]. 129 p. Available from: http://www.mmc.gov/sound/internationalwrkshp/pdf/hildebrand.pdf.

Hoagland P, Kite-Powell HL. 1997. Characterization and mitigation of marine debris in the Gulf of Maine. A report prepared for the U.S. Gulf of Maine Association under contract no. GM97-13. Gulf of Maine Council on the Marine Environment. 41 p.

Hofer T. 1998. Part II: Environmental and health effects resulting from marine bulk liquid transport. Environmental Science and Pollution Research 5(4):231-7.

[ICES] International Council for the Exploration of the Sea. 1992. Report of the ICES working group on the effects of extraction of marine sediments on fisheries. Copenhagen (Denmark): ICES Cooperative Research Report # 182.

Jasny M, Reynolds J, Horowitz C, Wetzler A. 1999. Sounding the depths: supertankers, sonar, and rise of undersea noise. Los Angeles (CA): Natural Resources Defense Council. Marine Mammal Protection Project. 38 p.

Kahler T, Grassley M, Beauchamp DA. 2000. A summary of the effects of bulkheads, piers, and other artificial structures and shorezone development on ESA listed salmonids in lakes. [Internet]. Bellevue (WA): Final Report to the City of Bellevue. [cited 2007 Dec 28]. 74 p. Available from: http://www.cityofbellevue.org/pdf/Utilities/dock_bulkhead.pdf.

Kalmijn AJ. 1988. Hydrodynamic and acoustic field detection. In: Atema J, Fay RR, Popper AN, Tavolga WN, editors. Sensory biology of aquatic animals. New York (NY): Springer-Verlag. p 83-130.

Karaki S, vanHoften J. 1975. Resuspension of bed material and wave effects on the Illinois and upper Mississippi Rivers caused by boat traffic. St. Louis (MO): US Army Engineer District. Cont. No. LMSSD 75-881.

Kavlock RJ, Ankley GT. 1996. A perspective on the risk assessment process for endocrine-disruptive effects on wildlife and human health. Risk Analysis 16(6):731-9.

Kavlock RJ, Daston GP, DeRosa C, Fenner-Crisp P, Gray LE, Kaattari S, Lucier G, Luster M, Mac MJ, Maczka C, and others. 1996. Research needs for the risk assessment of health and environmental effects of endocrine disruptors: a report of the US EPA-sponsored workshop. Environmental Health Perspectives 104 (Suppl 4):715-40.

Keevin TM. 1998. A review of natural resource agency recommendations for mitigating the impacts of underwater blasting. Reviews in Fisheries Science 6(4):281-313.

Keevin TM, Gaspin JB, Gitschlag GR, Hempen GL, Linton TL, Smith M, Wright DG. 1999. Underwater explosions: natural resource concerns, uncertainty of effects, and data needs. In: Proceedings of the twenty-fifth annual conference on explosives and blasting technique; 1999 Feb 7-10; Nashville, TN. Cleveland (OH): International Society of Explosive Engineers. p 105-16.

Keevin TM, Hempen GL, Schaeffer DJ. 1997. Use of a bubble curtain to reduce fish mortality during explosive demolition of Locks and Dam 26, Mississippi River. In: Proceedings of the twenty-third annual conference on explosives and blasting technique; 1997 Feb 2-5; Las Vegas, NV. Cleveland (OH): International Society of Explosive Engineers. p 197-206.

Kipple B, Gabriele C. 2004. Glacier Bay watercraft noise-noise characterization for tour, charter, private, and government vessels. Bermerton (WA): Naval Surface Warfare Center-Detachment Bermerton. Technical Report NSWCCD-71-TR-2004/545. 47 p.

Klein R. 1997. The effects of marinas and boating activities upon tidal waters. Owing Mills (MD): Community and Environmental Defense Services. 23 p.

Klein-MacPhee G, Macy WK, Berry W. 2004. *In situ* effects of suspended particulate loads produced by dredging on eggs of winter flounder (*Pseudopleuronectes americanus*). In: Ninth flatfish biology conference- oral presentation; 2004 Dec 1-2; Water's Edge Resort, Westbrook, CT. Woods Hole (MA): Northeast Fisheries Science Center Reference Document 04-13.

Knudsen FR, Schreck CB, Knapp SM, Enger PS, Sand O. 1997. Infrasound produces flight and avoidance responses in Pacific juvenile salmonids. Journal of Fish Biology 51:824-9.

Krone CA, Brown DW, Burrows DG, Chan S-L, Varanasi U. 1989. Butyltins in sediment from marinas and waterways in Puget Sound, Washington State, USA. Marine Pollution Bulletin 20(10):528-31.

Larson K, Moehl CE. 1990. Entrainment of anadromous fish by hopper dredge at the mouth of the Columbia River. In: Simenstad CA Jr, editor. Effects of dredging on anadromous Pacific coast fishes. Seattle (WA): Washington Sea Grant, University of Washington. p 102-12.

Longmuir C, Lively T. 2001. Bubble curtain systems for use during marine pile driving. New Westminister (British Columbia): Fraser River Pile and Dredge LTD. 9 p.

MacKenzie CL Jr. 2007. Causes underlying the historical decline in eastern oyster (*Crassostrea virginica* Gmelin, 1791) landings. Journal of Shellfish Research 26(4):927-38.

Markham VD. 2006. U.S. national report on population and the environment. New Canaan (CT): Center for Environment and Population. 67 p.

McGraw K, Armstrong D. 1990. Fish entrainment by dredges in Grays Harbor, Washington. In: Simenstad CA Jr, editor. Effects of dredging on anadromous Pacific coast fishes. Seattle (WA): University of Washington Sea Grant. p 113-31.

Messieh SN, El-Sabh MI. 1988. Man-made environmental changes in the southern Gulf of St. Lawrence, and their possible impact on inshore fisheries. In: El-Sabh MI, Murty TS, editors. Natural and man-made hazards. Boston (MA): D. Reidel Publishing Co. p 499-523.

Messieh SN, Rowell TW, Peer DL, Cranford PJ. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. Continental Shelf Research 11(8-10):1237-63.

Michel J, Helton D. 2003. Environmental considerations during wreck removal and scuttling. In: Abandoned Vessel Program (AVP) Conference Proceedings-National Maritime Salvage Conference; 2003 Sep 9-11; Crystal City, VA. Arlington (VA): American Salvage Association.

Milliken AS, Lee V. 1990. Pollution impacts from recreational boating: a bibliography and summary review. Narragansett (RI): Rhode Island Sea Grant. National Sea Grant Depository Publication # RIU-G-002.

Mills KE, Fonseca MS. 2003. Mortality and productivity of eelgrass *Zostera marina* under conditions of experimental burial with two sediment types. Marine Ecology Progress Series 255:127-34.

Moore M, Stolpe NE. 1995. Boating workshop raises tough questions. Commercial Fisheries News 22(7):6B-7B and 15B.

Mulvihill EL, Francisco CA, Glad JB, Kaster KB, Wilson RE. 1980. Biological impacts of minor shoreline structures in the coastal environment: state of the art review. Sidell (LA): US Fish and Wildlife. FWS/OBS-77/51. Vol. 1. 156 p.

Myrberg AA Jr. 1972. Using sound to influence the behavior of free-ranging marine mammals. In: Winn HE, Olla BL, editors. Behavior of marine animals. Vol. 2: Vertebrates. New York (NY): Plenum Press. p 435-68.

Myrberg AA Jr, Riggio RJ. 1985. Acoustically mediated individual recognition by a coral reef fish (*Pomacentrus partitus*). Animal Behaviour 33:411-6.

Neff JM. 1985. Polycyclic aromatic hydrocarbons. In: Rand GM, Petrocelli SR, editors. Fundamentals of aquatic toxicology. Washington (DC): Hemisphere Publishing. p 416-54.

Negri AP, Smith LD, Webster NS, Heyward AJ. 2002. Understanding ship-grounding impacts on a coral reef: potential effects of anti-foulant paint contamination on coral recruitment. Marine Pollution Bulletin 44:111-7.

Newell RC, Seiderer LJ, Hitchcock DR. 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. In: Ansell AD, Gibson RN, Barnes M, editors. Oceanography and marine biology: an annual review. Vol. 36. Oban, Argyll (Scotland): UCL Press. p 127-78.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol. 1.

Nightingale BJ, Simenstad CA Jr. 2001a. Overwater structures: marine issues. [Internet]. Olympia (WA): Washington Department of Fish and Wildlife. White Paper. [cited 2008 Jul 9]. 133 p + appendices. Available from: http://wdfw.wa.gov/hab/ahg/finalmar.pdf.

Nightingale B, Simenstad CA Jr. 2001b. Dredging activities: marine issues. [Internet]. Olympia (WA): Washington Department of Fish and Wildlife. White Paper. [cited 2008 Jul 17]. 119 p + appendices. Available from: http://wdfw.wa.gov/hab/ahg/finaldrg.pdf.

Omori M, VanderSpoel S, Norman CP. 1994. Impact of human activities on pelagic biogeography. Progress in Oceanography 34(2-3):211-9.

Orth RJ, Batiuk RA, Moore KA. 2002. A perspective on two decades of policies and regulations influencing the protection and restoration of submerged aquatic vegetation in Chesapeake Bay, USA. Bulletin of Marine Science 71(3):1391-403.

Orth RJ, Carruthers TJB, Dennison WC, Duarte CM, Fourqurean JW, Heck KL Jr, Hughes AR, Kendrick GA, Kenworthy WJ, Olyarnik S, Short FT, Waycott M, William SL. 2006. A global crisis for seagrass ecosystems. Bioscience 56(12):987-96.

Penttila D, Doty D. 1990. Results of 1989 eelgrass shading studies in Puget Sound. Olympia (WA): Washington Department of Fisheries. Draft Progress Report. 41 p.

Pertola S, Faust MA, Kuosa H. 2006. Survey on germination and species composition of dinoflagellates from ballast tanks and recent sediments in ports on the South Coast of Finland, North-Eastern Baltic Sea. Marine Pollution Bulletin 52(8):900-11.

Poston T. 2001. Treated wood issues associated with overwater structures in freshwater and marine environments. [Internet]. Olympia (WA): Washington State Departments of Transportation, Fish and Wildlife, and Ecology. White Paper. [cited 2008 Jul 9]. 85 p. Available from: http://wdfw.wa.gov/hab/ahg/finaltw.pdf.

Precht WF, Aronson RB, Swanson DW. 2001. Improving scientific decision-making in the restoration of ship-grounding sites on coral reefs. Bulletin of Marine Science 69(2):1001-12.

Readman JW, Kwong LLW, Grondin D, Bartocci J, Villeneuve JP, Mee LD. 1993. Coastal water contamination from triazine herbicide used in antifouling paints. Environmental Science and Technology 27(9):1940-2.

Richard JD. 1968. Fish attraction with pulsed low-frequency sound. Journal of the Fisheries Research Board of Canada 25(7):1441-52.

Richardson WJ, Greene CR, Malme CI, Thomas DH. 1995. Marine mammals and noise. San Diego (CA): Academic Press. 576 p.

Sand O, Enger PS, Karlsen HE, Knudsen F, Kvernstuen T. 2000. Avoidance responses to infrasound in downstream migrating European silver eels, *Anguilla anguilla*. Environmental Biology of Fishes 57(3):327-36.

Scholik AR, Yan HY. 2002. Effects of boat engine noise on the auditory sensitivity of the fathead minnow, *Pimephales promelas*. Environmental Biology of Fishes 63:203-9.

Shafer DJ. 1999. The effects of dock shading on the seagrass *Halodule wrightii* in Perdido Bay, Alabama. Estuaries 22(4):936-43.

Smith KN, Mezich RR. 1999. Blending management and research for protection of manatee habitat in Florida: the seagrass dock effect. In: Poster presentation at the 15th Biennial International Estuarine Research Federation Conference; 1999 Sep 25-30; New Orleans, LA. Port Republic (MD): Coastal and Estuarine Research Federation.

Smith LD, Negri AP, Philipp E, Webster NS, Heyward AJ. 2003. The effects of antifoulant-paint-contaminated sediments on coral recruits and branchlets. Marine Biology 143:651-7.

Stocker M. 2002. Fish, mollusks and other sea animals' use of sound, and the impact of anthropogenic noise in the marine acoustic environment. Journal of Acoustical Society of America 112(5):2431.

Stolpe NE, Moore M. 1997. Boating workshop raises tough questions. [Internet]. New Jersey Fishing. Commercial Fisheries News 22(7); c2007 [cited 2007 Dec 28]. Available from: http://www.fishingnj.org/artobm2.htm.

Stotz T, Colby J. 2001. January 2001 dive report for Mukilteo wingwall replacement project. Washington State Ferries Memorandum. 5 p + appendices.

Strayer DL, Hattala KA, Kahnle AW. 2004. Effects of an invasive bivalve (*Dreissena polymorpha*) on fish in the Hudson River estuary. Canadian Journal of Fisheries and Aquatic Sciences 61(6):924-41.

Sunda WG. 1994. Trace metal/phytoplankton interactions in the sea. In: Bidoglio G, Stumm W, editors. Chemistry of aquatic systems: local and global perspectives. Norwell (MA): Kluwer Academic Publishers Group. p 213-47.

Thayer WG, Fonseca MS, Kenworthy JW. 1997. Ecological value of seagrasses: a brief summary for the ASMFC habitat committee's SAV subcommittee. In: Stephan CD, Bigford TE, editors. Atlantic coastal submerged aquatic vegetation: a review of its ecological role, anthropogenic impacts, state regulation, and value to Atlantic coastal fisheries. Washington (DC): ASMFC Habitat Management Series #1. p 68 + appendices.

Thom RM, Shreffler DK. 1996. Eelgrass meadows near ferry terminals in Puget Sound: characterization of assemblages and mitigation impacts. Sequim (WA): Battelle Pacific Northwest Laboratories.

Thurberg FP, Gould E. 2005. Pollutant effects upon cod, haddock, pollock, and flounder of the inshore fisheries of Massachusetts and Cape Cod Bays. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 43-66.

Tyrrell MC. 2005. Gulf of Maine marine habitat primer. [Internet]. Gulf of Maine Council on the Marine Environment. vi + 54 p. [cited 2008 Jul 2]. Available from: www.gulfofmaine.org/habitatprimer/.

Uhrin AV, Holmquist JG. 2003. Effects of propeller scarring on macrofaunal use of the seagrass *Thalassia testudinum*. Marine Ecology Progress Series 250:61-70.

[USEPA] US Environmental Protection Agency. 2000. Environmental screening checklist and workbook for the water transportation industry. Washington (DC): US EPA Office of Water. EPA-305-B-00-004.

[USEPA] US Environmental Protection Agency. 2001. National management measures guidance to control nonpoint source pollution from marinas and recreational boating. Washington (DC): US EPA Office of Water. EPA-841-B-01-005.

Waite ME, Waldock MJ, Thain JE, Smith DJ, Milton SM. 1991. Reductions in TBT concentrations in UK estuaries following legislation in 1986 and 1987. Marine Environmental Research 32:89-111.

Walker DI, Kendrick GA. 1998. Threats to macroalgal diversity: marine habitat destruction and fragmentation, pollution, and introduces species. Botanica Marina 41(1):105-12.

Walker DI, Lukatelich RJ, Bastyan G, McComb AJ. 1989. Effect of boat moorings on seagrass beds near Perth, Western Australia. Aquatic Botany 36(1):69-77.

Warnken J, Dunn RJK, Teasdale PR. 2004. Investigation of recreational boats as a source of copper at anchorage sites using time-integrated diffusive gradients in thin film and sediment measurements. Marine Pollution Bulletin 49(9-10):833-43.

Weis JS, Weis P. 2002. Contamination of saltmarsh sediments and biota by CCA treated wood walkways. Marine Pollution Bulletin 44:504-10.

Weis P, Wies JS, Coohill LM. 1991. Toxicity to estuarine organisms of leachates from chromated copper arsenate treated wood. Archives of Environmental Contamination and Toxicology 20:118-24.

Wertheimer AC, Heintz RA, Thedinga JF, Maselko JM, Rice SD. 2000. Straying of adult pink salmon from their natal stream following embryonic exposure to weathered *Exxon Valdez* crude oil. Transactions of the American Fisheries Society 129(4):989-1004.

Wilber D, Brostoff W, Clarke D, Ray G. 2005. Sedimentation: potential biological effects of dredging operations in estuarine and marine environments. [Internet]. DOER Technical Notes Collection. Vicksburg (MS): US Army Engineer Research and Development Center. ERDC TN-DOER-E20. [cited 2008 Jul 22]. 14 p. Available from: http://el.erdc.usace.army.mil/dots/doer/pdf/doere20.pdf.

Wilber DH, Clarke DG. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21(4):855-75.

Wilbur AR, Pentony MW. 1999. Human-induced nonfishing threats to essential fish habitat in the New England region. American Fisheries Society Symposium 22:299-321.

Williams GD, Thom RM. 2001. Marine and estuarine shoreline modification issues. [Internet]. White Paper submitted to Washington Department of Fish and Wildlife, Washington Dept. of Ecology, Washington Dept. of Transportation. Sequim (WA): Battelle Memorial Institute. [cited 2008 Jul 9]. 99 p. Available from: http://wdfw.wa.gov/hab/ahg/marnrsrc.htm.

Wright DG. 1982. A discussion paper on the effects of explosives on fish and marine mammals in the waters of the Northwest Territories. Winnipeg (Manitoba): Western Region, Department of Fisheries and Oceans. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1052.

Yousef YA. 1974. Assessing effects on water quality by boating activity. Washington (DC): US Environmental Protection Agency. EPA-670/2-74-072. 60 p.

Zelo I, Helton D. 2005. Removal of grounded, derelict or abandoned vessels as site restoration. In: Abandoned Vessel Program (AVP) Conference Proceedings-International Oil Spill Conference; 2005 May 15-19; Miami Beach, FL. Seattle (WA): NOAA Ocean Service Office of Response and Restoration. 15 p.

CHAPTER SIX: OFFSHORE DREDGING AND DISPOSAL ACTIVITIES

Introduction

This chapter describes activities associated with offshore dredging and disposal and their potential effects on living marine resources and habitats in the northeast region of the United States. For purposes of this discussion, the "offshore" environment is defined as those waters and seabed areas considered to be "estuarine" environments and extending offshore to and occasionally beyond the edge of the continental shelf. For example, while the open waters of Chesapeake Bay, MD/VA, and Long Island Sound, NY/CT, are considered offshore for this discussion, the coves and embayments within those waters bodies are not. In addition, Raritan Bay, NY/NJ, (lower New York Harbor) and similar areas are considered offshore environments. Dredging and disposal activities within riverine habitats have been discussed in the Alteration of Freshwater Systems chapter of this report, and information on dredging within navigation channels can be reviewed in the Marine Transportation chapter of this report.

Offshore Mineral Mining

Introduction

There is an increasing demand for beach nourishment sand and a smaller, but growing, demand for construction and "stable fill" grade aggregates. As the historic landside sources of these materials have been reduced, there has been a corresponding move towards mining the continental shelf to meet this demand. It is expected that the shift to offshore mineral extraction will continue and escalate, particularly in areas where glacial movements have relocated the desired material to the continental shelf. Typically, these deposits are not contaminated because of their offshore location and isolation from anthropogenic pollution sources. Beginning in the mid-1970s, the US Geological Survey began mapping the nature and extent of the aggregate resources in coastal and nearshore continental shelf waters throughout the northeast beyond the 10-m isobath. Between 1995 and 2005, the Minerals Management Service (MMS), which oversees offshore mineral extractions, regulated the relocation of over 23 million cubic yards of sand from the Outer Continental Shelf (OCS) for beach nourishment projects (MMS 2005a). The OCS is defined as an area between the seaward extent of states' jurisdiction and the seaward extent of federal jurisdiction. Currently, the MMS, in partnership with 14 coastal states, is focusing on collecting and analyzing geologic and environmental information in the OCS in order to study sand deposits suitable for beach nourishment and wetlands protection projects and to assess the environmental impacts of OCS mining in general (Drucker et al. 2004). With the advances in marine mining and "at sea" processing, aggregate extraction can occur in waters in excess of 40 m **(**MMS 2005a**)**.

Mineral extraction is usually conducted with hydraulic dredges by vacuuming or, in some cases, by mechanical dredging with clamshell buckets in shallow water mining sites. Mechanical dredges can have a more severe but localized impact on the seabed and benthic biota, whereas hydraulic dredges may result in less intense but more widespread impact (Pearce 1994). The impacts of offshore mineral mining on living marine resources and their habitats include: (1) the removal of substrates that serve as habitat for fish and invertebrates; (2) creation of (or conversion to) less productive or uninhabitable sites such as anoxic depressions or highly hydrated clay/silt substrates; (3) release of harmful or toxic materials either in association with actual mining, or from incidental or accidental releases from machinery and materials used for mining; (4) burial of productive habitats during beach nourishment or other shoreline stabilization activities; (5) creation of harmful suspended sediment levels; and (6) modification of hydrologic conditions causing adverse impacts to desirable habitats (Pearce 1994; Wilber et al. 2003).

In addition, mineral extraction can potentially have secondary and indirect adverse effects on fishery habitat at the mining site and surrounding areas. These impacts may include accidental or intentional discharges of mining equipment and processing wastes and degradation or elimination of marine habitats from structures constructed to process or transport mined materials. These secondary effects can sometimes exceed the initial, direct consequences of the offshore mining.

Loss of benthic habitat types

Offshore benthic habitats occurring on or over target aggregates may be adversely affected by mining. The mineral extraction process can disrupt or eliminate existing biological communities within the mining or borrow areas for several years following the excavation. Filling in of the borrow areas and reestablishment of a stable sediment structure is dependent upon the ability of bottom currents to transport similar sediments from surrounding areas to the mining site (ICES 1992). The principal concern noted by the International Council for the Exploration of the Sea (ICES) Working Group on the Effects of Extraction of Marine Sediments on Fisheries was dredging in spawning areas of commercial fish species (ICES 1992). Of particular concern to the ICES Working Group are fishery resources with demersal eggs (e.g., Atlantic herring [*Clupea harengus*] and sand lance [*Ammodytes marinus*]). They report that when aggregates are removed, Atlantic herring eggs are taken with them, resulting in lost production to the stock. Stewart and Arnold (1994) list the impacts on Atlantic herring from offshore mining to include the entrainment of eggs, larvae, and adults; burial of eggs; and effects of the turbidity plume on demersal egg masses. Gravel and coarse sand have been identified as preferred substrate for Atlantic herring eggs on Georges Bank and in coastal waters of the Gulf of Maine (Stevenson and Scott 2005).

Conversion of substrate/habitat and changes in community structure

Disposal of residues ("tailings") of the mining process can alter the type, as well as the functions and values, of habitats which can then alter the survival and growth of marine organisms. The tailings are often fine-grained and highly hydrated, making them very dissimilar to the natural seafloor, particularly in depths where wave energy and currents are capable of winnowing or sorting sediments and relocating them to depositional areas. It has been found that wave forces are affecting habitats in the New York Bight at depths in excess of 22 m (USACE 2005a). In laboratory experiments, benthic dwelling flatfishes (Johnson et al. 1998a) and crabs (Johnson et al. 1998b) persistently avoided sediments comprised of mine tailings.

Additionally, there can be adverse impacts from aggregate and/or mineral mining on nearby habitats associated with the removal and disturbance of substrate (Scarrat 1987). Seabed alteration can fragment habitat, reduce habitat availability, and disrupt predator/prey interactions, resulting in negative impacts to fish and shellfish populations. Not all offshore aggregate mining results in adverse impacts on seabed resources. Hitchcock and Bell (2004) conducted a detailed study of the effects from a small-scale, aggregate mining operation off the south coast of the United Kingdom and found physical impacts on the seabed to be limited to a downtide zone approximately 300 m from the dredge area. Related studies at this mining operation reported no detectable impact on the surrounding benthic communities, despite a small change in seabed particle size distribution (Hitchcock and Bell 2004).

Long-term mining can alter the habitat to such a degree that recovery may be extremely protracted and create habitat of limited value to benthic communities during the entire recovery period (van Dalfsen et al. 2000). For example, construction grade aggregate removal in Long Island Sound, Raritan Bay (lower New York Harbor) and the New Jersey portion of the intercoastal waterway have left borrow pits that are more than twice the depth of the surrounding area. The pits have remained chemically, physically, and biologically unstable with limited diversity communities for more than five decades. These pits were used to provide fill material for interstate transportation projects and have been investigated to assess their environmental impact (Pacheco 1984). Borrow pits in Raritan Bay were found to possess depressed benthic communities and elevated levels of highly hydrated and organically enriched sediments (Pacheco 1984). In one example, aggregate mining operations from the 1950s through the 1970s created a 20 m deep borrow pit in an area of Raritan Bay that, although the mining company was required to refill the pit, remains today as a rapid deposition area filling with fine-grained sediment and organic material emanating from the Hudson River and adjacent continental shelf (Pacheco 1984**)**. The highly hydrated sediments filling the depressions are of limited utility to colonizing benthic organisms.

In offshore mining operation sites, the character of the sediment which is exposed or subsequently accumulates at the extraction site is important in predicting the composition of the colonizing benthic community (ICES 1992). If the composition and topography of the extraction site resembles that which originally existed, then colonization of it by the same benthic fauna is likely (ICES 1992).

Changes in sediment composition

A review of studies conducted in Europe and Great Britain found that infilling and subsequent benthic recovery of borrow areas may take from 1-15 years, depending upon the tide and current strength, sediment characteristics, the stock of colonizing species and their immigration distance (ICES 1992). Typically the reestablishment of the community appears to follow a successional process similar to those on abandoned farmlands. Germano et al. (1994) described this process, reporting that pioneering species (i.e., Stage I colonizers) usually do not select any particular habitat but attempt to survive regardless of where they settle. These species are typically filter feeders relying on the availability of food in the overlying water rather than the seafloor on which they reside. Thus, their relationship to the substrate is somewhat tenuous, and their presence is often ephemeral. However, their presence tends to provide some stability to the seafloor, facilitating subsequent immigrations by other species that bioturbate the sediment seeking food and shelter. Their arrival induces further substrate consolidation and compaction. These colonizers are usually deemed to be Stage II community species. The habitat modification activities of Stage I and II species advance substrate stability and consolidation enough for it to support, both physically and nutritionally, the largest community members (i.e., Stage III). The benthic community instability caused by dredging gives rise to one of the principal justifications for retaining benthic disturbances: the disrupted site may become heavily populated by opportunistic (i.e., Stage I) colonizer species that flourish briefly and provide motile species with an abundance of food during late summer and fall periods (Kenny and Rees 1996). However, if environmental stresses are chronic, the expected climax community may never be attained (Germano et al. 1994).

If the borrow area fails to refill with sediment similar to that which was present prior to mining, the disturbed area may not possess the original physical and chemical conditions and recovery of the community structure may be restricted or fail to become reestablished. Dredge pits that have been excavated to depths much greater than the surrounding bottom often have very slow infill rates and can be a sink for sediments finer than those of the surrounding substrate (ICES 1992).

Changes in bottom topography and hydrology

The combination of rapid deposition, anomalous sediment character, and an uneven topography, as compared to the surrounding seafloor, limit recolonization opportunities for harvesting purposes (Wilk and Barr 1994). By altering bottom topography, aggregate mining can reduce localized current strength, resulting in lowered dissolved oxygen concentrations and increased accumulation of fine sediments inside borrow pits (ICES 1992). One potential benefit of some borrow pits is that they appear to provide refugia for pelagic species such as alewife (*Alosa pseudoharengus*) and scup (*Stenotomus chrysops*), as well as demersal species such as tautog (*Tautoga onitis*) and black sea bass (*Centropristis striata*) during seasonally fluctuating water temperatures (Pacheco 1984). However, it is doubtful these benefits outweigh the persistent adverse affects associated with borrow pits (Palermo et al. 1998; Burlas et al. 2001). Other consequences of aggregate mining may include alteration of wave and tidal current patterns which could affect coastal erosion (ICES 1992).

Siltation, sedimentation, and turbidity

Offshore mining can increase the suspended sediment load in the water column, increasing turbidity that can then adversely affect marine organisms, particularly less motile organisms such as shellfish, tunicates, and sponges. The duration of the turbidity plume in the water column depends upon the water temperature, salinity, current speed, and the size range of the suspended particles (ICES 1992). The distance the dredged material is transported from the excavation site will be dependent upon the current strength, storm resuspension, water salinity and temperature, and the grain size of the suspended material (ICES 1992).

The life stages of the affected taxa are an important factor affecting the type and extent of the adverse impacts (Wilber and Clarke 2001). As a general rule, the severity of sedimentation and turbidity effects tends to be greatest for early life stages and for adults of some highly sensitive species (Newcombe and Jensen 1996; Wilber and Clarke 2001). In particular, the eggs and larvae of nonsalmonid estuarine fishes exhibit some of the most sensitive responses to suspended sediment exposures of all the taxa and life history stages for which data are available (Wilber and Clarke 2001). Stewart and Arnold (1994) list the impacts on Atlantic herring from offshore mining to include the effects of the turbidity plume on demersal egg masses.

Impacts to water quality

The release of material into the water column during offshore mining operations can degrade water quality if the excavated material is high in organic content or clay. The effects of mixing on the water column are likely to include increased consumption of oxygen by decomposing organic matter and the release of nutrients (ICES 1992). However, mined aggregate material is typically low in organic content and clay, and any increase in the biological oxygen demand is thought to be minor and of limited spatial extent (ICES 1992).

Deep borrow pits can become anaerobic during certain times of the year. The dissolved oxygen concentration within these 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).

Release of contaminants

A number of factors (i.e., environmental, geochemical, and biological) influence the potential release and bioavailability of sediment contaminants. The toxicity of such releases, in general, is primarily dependent upon the contaminant involved, its concentration in the sediments and its chemical/geochemical state. Persistent organic pollutants (POPs), such as polyaromatic hydrocarbons (PAHs), pesticides, and polychlorinated biphenyl (PCBs), are sequestered in the total organic carbon (TOC) fraction of sediments (USEPA 2003a; USEPA 2003b; USEPA 2003c). Similarly, heavy metals are sequestered by acid volatile sulfides (AVS) and the TOC fraction of marine sediments (USEPA 2005a). For POPs like PAHs, the ratio of the concentrations of these contaminants relative to those of the fractions govern bioavailability and hence toxicity (USEPA 2003a). In the case of metals, bioavailability is governed by an excess of AVS concentrations relative to the metal concentrations as normalized by TOC (USEPA 2005a). Sand and gravel sediments typically contain low TOC and AVS concentrations, and where there is a prominent source of POPs and metals, such as in highly industrialized riverways, these coarser sediments could in fact release such contaminants when disturbed or oxidized. However, the coarse-grained sediments typically targeted for aggregate mining tend to be found in high-energy environments which are not depositional areas that can be sinks for fine-grained material containing POPs and metals. Since most offshore sand and gravel deposits do not have prominent nearby sources of POPs and metals, these deposits are generally low in contaminants (ICES 1992; Pearce 1994). Thus, the mining of offshore sand and gravel material typically do not release high levels of contaminants. In addition, because of their relatively large particle size, low surface area relative to total bulk, and low surface activity (i.e., few clay or organic materials to interact chemically), there is usually little chemical interaction in the water column (Pearce 1994). However, extraction of material in estuaries or deep channels, where fine material accumulates and is subject to anthropogenic pollution deposition, may be more likely to release harmful chemicals during dredging and excavation (Pearce 1994). Refer to the chapters on Coastal Development, Marine Transportation, and Chemical Effects: Water Discharge Facilities for additional information on the release of contaminants during dredging and excavation.

Sediment transport from site

Excavation at an offshore mining site that contains fine material can release suspended sediments into the water column during the excavation, as well as in the sorting or screening process. The distance the dredged material is transported from the excavation site will be dependent upon the current strength, storm resuspension, water salinity and temperature, and the grain size of the suspended material (ICES 1992). Some of the potential effects of redeposition of fines include smothering of demersal fish eggs on spawning grounds and the suffocation of filter-feeding benthos, such as shellfish and anemones (ICES 1992; Pearce 1994). Small-scale aggregate mining operations that are conducted in relatively shallow water and involving sandy, coarse-grained sediments often have relatively minimal physical and biological impacts on the surrounding seabed (Hitchcock and Bell 2004).

Noise impacts

Anthropogenic sources of ocean noise appear to have increased over the past decades, and have been primarily attributed to commercial shipping, offshore gas and oil exploration and drilling, and naval and other uses of sonar (Hildebrand 2004). Offshore mineral mining likely contributes to the overall range of anthropogenic ocean noise, but little information exists regarding specific effects on marine organisms and their habitats or the importance of offshore mining relative to other sources of anthropogenic noise. The dredging equipment noise generated in offshore mining may be similar to navigation channel dredging in nearshore habitats; however, because of the greater water depths involved in offshore mining, the noise may be propagated for greater distances than in confined nearshore areas (Hildebrand 2004). Reductions in Atlantic herring catches on the Finnish coast were hypothesized to be due to disturbance to the herring movement patterns by noise and activity associated with sand and gravel mining activities (Stewart and Arnold 1994). Refer to the chapters on Global Affects and Other Impacts and Marine Transportation for additional information on noise impacts.

Conservation measures and best management practices for offshore mineral mining

1. Avoid mining in areas containing sensitive or unique marine benthic habitats (e.g., spawning and feeding sites, surface deposits of cobble/gravel substrate).
2. Complete a comprehensive characterization of the borrow site and its resources prior to permit completion. Some of the components of a thorough assessment include:
    a. Determine the optimum dimensions of the borrow pit (i.e., small and deep areas or wide and shallow areas) in terms of minimizing the effects on resources.
    b. Prioritize the optimal locations of sand mining in terms of effects on resources.
    c. Assess the sand infill rates of borrow pits after completion.
    d. Assess the sediment migration patterns and rates as well as the side slope and adjacent natural seabed stability of the borrow pits after completion.
    e. Model and estimate the effect of massive and/or long-term sand mining on the surrounding seabed, shoreface (i.e., inner continental shelf), sand budgets, and resources.
    f. Assess the effect of removal (by dredging) of offshore sand banks/shoals on the surrounding natural seabed, adjacent shoreline, and the resources that use those habitats.
    g. Assess the effect of massive and/or long-term sand mining on the ecological structure of the seabed.
    h. Assess the effect of noise from mining operations on the feeding, reproduction, and migratory behavior of marine mammals and finfish.
3. Use site characterization and appropriate modeling to determine the areal extent and depth of extraction that affords expedited and/or complete recovery and recolonization times.
4. Employ sediment dispersion models to characterize sediment resuspension and dispersion during mining operations. Use model outputs to design mining operations, including "at sea" processing, to limit impacts of suspended sediment and turbidity on fishery resources and minimize the area affected.
5. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in offshore mining review processes.
6. Use seasonal restrictions when appropriate to avoid temporary impacts to habitat during species critical life history stages (e.g., spawning, and egg, embryo, and juvenile development). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements. Resource managers should incorporate adequate time for habitat recovery of affected functions and values to levels required by managed species.

Petroleum Extraction

Introduction

After some intense but unsuccessful petroleum exploration on the northeastern US continental shelf, the attention for commercial quantities of oil and gas have been directed elsewhere. Georges Bank and the continental shelf off New Jersey were thought to contain significant reserves of natural gas and several exploratory wells were drilled to locate and characterize those reserves in the late 1980s and early 1990s. At that time, few commercially viable reserves were found and the focus of petroleum exploration shifted to other regions. However, this could change in the future considering the escalating market prices and dwindling supplies of petroleum. Should renewed interest in offshore petroleum exploration and extraction in the northeast region occur, existing regulatory guidance on petroleum exploration and extraction, as well any recent research and development efforts, should be employed to ensure that marine resource impacts can be avoided, minimized, and compensated for these types of activity.

Petroleum extraction has impacts similar to mineral mining but usually with significantly less of an impact footprint (excluding spills). However, there is more risk and occurrence of adverse impacts associated with equipment operation, process related wastes and handling of byproducts (e.g., drill cuttings and spent drilling mud) which can disrupt and destroy pelagic and benthic habitats (Malins 1977; Wilk and Barr 1994). Potential releases of oil and petroleum byproducts into the marine environment may also occur as a result of production well blow-outs and spills.

Drilling muds are used to provide pressure and lubrication for the drill bit and to carry drill cuttings (crushed rock produced by the drill bit) back to the surface. Drilling muds and their additives are complex and variable mixtures of fluids, fine-grained solids, and chemicals (MMS 2005b). Some of the possible impacts associated with petroleum extraction include the dispersion of soluble and colloidal pollutants, as well as the alteration of turbidity levels and benthic substrates. Many of these impacts can be mitigated by on-site reprocessing and by transferring substances deemed inappropriate for unrestricted openwater disposal to landside disposal.

For more information on petroleum-related impacts and conservation recommendations for petroleum exploration, production, and transportation refer to the Energy-related Activities chapter of this report.

Offshore Dredged Material Disposal

Introduction

The disposal of dredged material in offshore waters involves environmental effects beyond those associated with the actual dredging operations. The US Army Corps of Engineers (USACE) disposes approximately 65% of its dredged material in open water, as opposed to "upland," or land disposal (Kurland et al. 1994). Although some adverse environmental effects can be avoided with land disposal, there are a number of drawbacks including securing large tracts of land, material handling problems, overflow and runoff of polluted water, saltwater intrusion into groundwater, and costs of transporting material to land disposal sites (Kurland et al. 1994).

Disposal of dredged material is regulated under the Clean Water Act (CWA) and the Marine Protection, Research, and Sanctuaries Act (MPRSA), also known as the Ocean Dumping Ban Act (33 U.S.C. § 1251 and 1401 et seq.). The differences in the two Acts are found in the necessity and type(s) of sediment testing required by each. Generally, ocean dumping only requires biological testing if it is determined that the sediments do not meet the testing exclusion criteria as specified under the MPRSA (i.e., are contaminated). While the CWA provides for biological testing, it does not require such tests to determine whether the sediment meets the 404b testing guidelines unless specified by the USACE or the US Environmental Protection Agency (US EPA). The US EPA and the USACE are currently involved in discussions intended to combine the testing and evaluation protocols described in regulations, and in the "Greenbook" (Ocean Dumping Ban Act) and "Inland" (CWA) testing manuals. Currently, the US EPA and USACE use a tiered approach under both Acts, based upon empirical data gathered from each evaluated dredging project for determining the appropriate management options for dredge spoils (i.e., unconfined open water disposal, open water disposal with capping [CWA only], no open water disposal, or confined area disposal in harbors). Under the CWA, sediment quality guidelines or benchmarks can be used in the lower tiers to determine compliance with 404b guidelines or the need for futher testing. Although not required under the MPRSA, regulators in practice often use sediment chemistry to help determine the contaminant and sampling requirements for biological tests.

Offshore disposal sites are identified and designated by the US EPA using a combination of the MPRSA and National Environmental Policy Act (NEPA) criteria. However, the permitted use of designated disposal sites under these laws is not usually associated with the designation of the sites. To be eligible to use an offshore (i.e., federal waters) disposal site for dredged materials, project proponents must demonstrate: (1) that there are no reasonable and practical alternative disposal options available and; (2) that the sediments are compatible with natural sediments at the disposal site and are not likely to disrupt or degrade natural habitats and/or biotic communities (USEPA 2005b). Dredge material disposed at sites managed under the MPRSA must meet Ocean Dumping Ban Act criteria, which do not permit disposal of contaminated dredged material (USEPA 2005b).

Burial/disturbance of benthic habitat

Studies using sidescan sonar and bottom video have been used to distinguish natural sediment character and evidence of past dumping of mud and boulders on sand bottom (Buchholtz ten Brink et al. 1996). These studies have indicated that not only have dumped materials disturbed and altered benthic habitats, but that in some cases (such as on Stellwagen Basin) the material dumped in the past was scattered far from the intended target areas (Buchholtz ten Brink et al. 1996). The discharge of dredged material disturbs benthic and pelagic communities during and after disposal. The duration and persistence of those impacts to the water column and seafloor are related to the grain size and specific gravity of the dredge spoil. Impacts to benthic communities are identified and assessed in the site designation documents (Battelle 2004; URI 2003), which may include benthic communities being buried and smothered and the physicochemical environment in which they reside being altered.

However, Rhoads and Germano (1982, 1986) and Germano et al. (1994) note that recolonization of benthic infauna at a disposal site following dumping often leads to increased occurrences of opportunistic species (Stage I), which are then heavily preyed upon by Stage II and III (e.g., target fisheries) species. According to these studies, this plethora of prey, resulting from the disturbance of the community structure, can at least temporarily increase the productivity at the disposal site. However, chronic disturbance from repeated disposal may prevent Stage III communities from establishing (Germano et al. 1994).

Conversion of substrate/habitat and changes in sediment composition

Dumping dredged materials results in varying degrees of change in the physical, chemical, and biological characteristics of the substrate. The discharges can adversely affect infauna, including benthic and epibenthic organisms at and adjacent to the disposal site by burying immobile organisms or forcing motile organisms to migrate from the area. Benthic infauna species that have greater burrowing capabilities may be better able to extricate themselves from the overburden of sediment. Seasonal constraints on dredging and disposal not withstanding, it is assumed that there is a cyclical and localized reduction in the populations of benthic organisms at a disposal site. Plants and benthic infauna present prior to a discharge are unlikely to recolonize if the composition of the deposited material is significantly different (NEFMC 1998). Altered sediment composition at the disposal site may reduce the availability of infaunal prey species, leading to reduced habitat quality (Wilber et al. 2005).

Siltation, sedimentation, and turbidity

Increased suspended sediment released during the discharge process and the associated increase in turbidity may hinder or disrupt activities in the pelagic zone (i.e., predator–prey relationships and photosynthesis rates). It has been estimated that less than 5% of the material in each disposal vessel is unaccounted for during and after the disposal activity (Bohlen et al. 1996), but the specific volume is influenced by both mechanical and sediment characteristics.

The discharge of dredged material usually results in elevated levels of fine-grained mineral particles, usually smaller than sand (i.e., silt/clay), and organic particles being introduced into the water column (i.e., suspended sediment plumes). The suspended particulates reduce light penetration, which affects the rate of photosynthesis and the primary productivity of an aquatic area. Typically, the suspended materials are dispersed and diluted to levels approaching ambient within 1-4 hours of the release (Bohlen et al. 1996). However, the turbidity plume resulting from a discharge can last much longer, particularly near the bottom, if the dredge material is composed of fine-grain material. In the plume field, living marine resources may experience either reduced or enhanced feeding ability as a result of the disruption of water clarity, depending upon the predator-prey relationships and the type(s) of avoidance/feeding methodologies used by the species. For instance, summer flounder (*Paralichthys dentatus*) and bluefish (*Pomatomus saltatrix*) are sight feeders and avoid areas with reduced water clarity resulting from suspended sediment such as might be found at a dredging or disposal site (Packer et al. 1999). Conversely, recent deposits of sediment at dumpsites have been reported to act as an attractant for other species of fish and crustaceans such as winter flounder (*Pseudopleuronectes americanus*) and American lobster (*Homarus americanus*) even though winnowing of fine-grained material from the excavation site or deposit mound was ongoing at the site (USACE 2001).

Generally, the severity of the effects of suspended sediments on aquatic organisms increases as a function of the 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. Mortality caused by suspended sediments for estuarine species have been reported from less than 1000 mg/L for 24 hours in highly sensitive species (e.g., Atlantic silversides [*Menidia menidia*], juvenile bluefish [*Pomatomus saltatrix*]) to greater than 10,000 mg/L for 24 hours in tolerant species (e.g., mummichog [*Fundulus heteroclitus*], striped killifish [*Fundulus majalis*], spot [*Leiostomus xanthurus*], oyster toadfish [*Opsanus tau*], hogchoker [*Trinectes maculates*]) (Wilber and Clark 2001). The egg and larval stages of marine and estuarine fish exhibit some of the most sensitive responses to suspended sediment exposures of all the taxa and life history stages studied (Wilber and Clark 2001). Impacts that have been identified for demersal eggs of fish from sedimentation and suspended sediments include delayed hatching and decreased hatching success (Wilber and Clark 2001; Berry et al. 2004). The development of larvae may be delayed or altered after exposure of elevated suspended sediments, and increased mortality rates in the larvae of some species, such as striped bass (*Morone saxatilis*) and American shad (*Alosa sapidissima*), have been reported with exposure of suspended sediment concentrations less than or equal to 500 mg/L for 3 to 4 days (Wilber and Clark 2001).

The effects of sedimentation on benthic organisms can include smothering and decreased gas exchange, toxicity from exposure to anaerobic sediments, reduced light intensity, and physical abrasion (Wilber et al. 2005). Mobile benthic species that require coarse substrates, such as gravel or cobble (e.g., American lobster) may be forced to seek alternate habitat that is less optimal or compete with other species or individuals for suitable habitat (Wilber et al. 2005). Messieh et al. (1981) investigated sedimentation impacts on Atlantic herring in laboratory experiments and found increased mortality in herring eggs, early hatching and shorter hatching lengths, and reduced feeding success in herring larvae leading to stunted growth and increased mortality.

Although there is generally a consensus among scientists and resource managers that elevated suspended sediments and sedimentation on benthic habitat caused by dredging and disposal of dredge spoils result in adverse impacts to marine organisms, the specific effects on biological communities need to be better quantified. Additional research is needed to investigate dose-response models at scales appropriate for dredging and disposal and for appropriate species and life history stages (Wilber et al. 2005).

Release of contaminants

Dredged material suspended in the water column can react with the dissolved oxygen in the water and result in localized depression of the oxygen level. However, research has indicated that reductions in dissolved oxygen levels during offshore sediment disposal is not appreciable or persistent in the general sediment classes found in the northeast region (USACE 1982; Fredette and French 2004; USEPA 2004).

In certain situations, trace levels of toxic metals and organics, pathogens, and viruses adsorbed or adhered to fine-grained particulates in the dredged material may become biologically available to organisms either in the water column or through food chain processes. Some of these pollutants and their concentrations are evaluated during project-specific sediment testing required under the MPRSA and CWA. Adverse chemical effects at the disposal site can be minimized through the sediment testing requirements under the MPRSA and CWA, since the discharge of potentially toxic materials are generally prohibited. Risk assessment approaches are used to further evaluate potential impacts using results from the MPRSA and CWA bioaccumulation and toxicity testing. In addition, monitoring is conducted to ensure that the biological and ecological functions and values are maintained within the site, notwithstanding the physical impacts associated with continued use of the site. However, some discharges of contaminated material may be permitted under CWA disposal regulations, if the sediments meet minimum testing criteria or the toxic affects can be managed by capping with clean material.

Fredette and French (2004) concluded that, after thirty-five years of monitoring and research, dredged material evaluated through preproject testing and deposited in properly located ocean disposal sites will remain where it is placed and have no unacceptable adverse effects on nearby marine resources. Furthermore, they concluded that the only discernible adverse impacts were near-field and short-term. These determinations were based on the magnitude of disposal activity relative to natural (e.g., storms) and other anthropogenic (e.g., outfalls) impacts (Rhoads

1994; Rhoads et al. 1995) and the low level of disposal-related impacts that have been documented (Fredette et al. 1993).

Changes in bottom topography, altered hydrological regimes, and altered current patterns

A concern often raised is the stability of dredge spoil sediments placed on the seafloor. Because ocean disposal sites are typically located in low current areas with water depths in excess of the active erosion zone, the material is generally contained within the disposal site. However, before 1985, dredged material sites were occasionally located in water depths insufficient to retain materials placed there (USEPA 1986). For example, the Mud Dump Site, located in the New York Bight Apex slope area off New York Harbor, contains water depths as shallow as 15 m and the site experienced extensive erosion by a nor'easter storm in October 1992 (USEPA 1997). Reclassified as a remediation site in 1997, the site is now known as the Historic Area Remediation Site (HARS). Erosion was reported at depths of 26 m, and the winnowed sediment included grain sizes up to small cobble. Fortunately, much of the sediment was relocated into deeper portions of the site westward of the erosion field (USEPA 1997). More comprehensive evaluation protocols have been put into place since 1985 to prevent dredged or fill material discharged at authorized sites from modifying current patterns and water circulation by obstructing the flow, changing the direction or velocity of water flow and circulation, or otherwise significantly altering the dimensions of a water body.

The USACE utilizes more than twenty selected or designated offshore dredged material disposal sites in the northeast region of the United States. Several of these sites have been used because they are dispersive in nature. These sites are used, normally, to put littoral material back into the nearshore drift pattern. The containment sites have an average size of 1.15 square nautical miles in size (USACE 2005b). By law and regulation, the significant adverse effects of dredged material disposal activities must be contained within the designated or selected disposal site and even those impacts must not degrade the area's overall ecological health. There is some dispersion of fine-grained sediments and contaminants outside the sites. Each site is required to have and be managed under a dredged material monitoring and management plan that assesses the health and well-being of the site and surrounding environment. Monitoring of disposal sites is a part of these plans, which is designed to ensure that any degradation of resources or alteration in seafloor characteristics are identified and would illicit actions by permitting agencies (USEPA 2004).

Release of nutrients/eutrophication

Nutrient overenrichment, or eutrophication, is one of the major causes of aquatic habitat decline associated with human activities (Deegan and Buchsbaum 2005). There are point sources of nutrients, such as sewage treatment outfalls, and nonpoint sources, such as urban storm water runoff, agricultural runoff, and atmospheric deposition, which have been discussed in other chapters of this report. Elevated levels of nutrients have undesirable effects, including: (1) increased incidence, extent, and persistence of blooms of noxious or toxic species of phytoplankton; (2) increased frequency, severity, spatial extent, and persistence of hypoxia; (3) alterations in the dominant phytoplankton species, which can reduce the nutritional and biochemical nature of primary productivity; and (4) increased turbidity levels of surface waters, leading to reductions in submerged aquatic vegetation (O'Reilly 1994).

Sediment particles can bind to some nutrients, and resuspension of sediments following dredge material disposal can cause a rapid release of nutrients to the water column (Lohrer and Wetz 2003). Ocean disposal of dredge material with high organic content can result in oxygen reduction (hypoxia) or even anaerobic conditions (anoxic) on the bottom and overlaying waters, particularly during periods when strong thermoclines are present (Kurland et al. 1994). Hypoxic and anoxic conditions can kill benthic organisms or even entire communities and lead to a proliferation of stress-tolerant species of reduced value to the ecosystem (Kurland et al. 1994). Generally, offshore waters are less sensitive to disposal of dredge material containing nutrients than inshore, enclosed water bodies.

Both the MPRSA and CWA regulations prohibit the discharge of dredge material containing high organic content and nutrient levels if the discharge results in adverse effects to the marine environment. However, prior to the stricter regulations instituted in the 1980s, the discharge of sewage sludge was permitted for decades in nearshore and offshore waters of many urbanized centers of the northeastern US coast (Barr and Wilk 1994).

Conservation measures and best management practices for dredge material disposal

1. Ensure that all options for disposal of dredged materials at sea are comprehensively assessed. The consideration of upland alternatives for dredged material disposal sites must be evaluated before offshore sites are considered.
2. Ensure that adequate sediment characterizations are completed and available for making informed decisions.
3. Ensure that adequate resource assessments are completed and available during project evaluation.
4. Employ sediment dispersion models to characterize sediment resuspension and dispersion during operations. Use model outputs to design disposal operations, including measures to avoid and minimize impacts from suspended sediment and turbidity on living marine resources. Sediment dispersion models should be field-verified to various sediment and hydraulic conditions to ensure they have been calibrated appropriately to predict sediment transport and dispersion.
5. Consider "beneficial uses" of dredged material, as appropriate.
6. Ensure that the site evaluation criteria developed for selection or designation of dredged material disposal sites have been invoked and evaluated, as appropriate.
7. Avoid dredged material disposal activities in areas containing sensitive or unique marine benthic habitats (e.g., spawning and feeding sites, surface deposits of cobble/gravel substrate).
8. Employ all practicable methods for limiting the loss of sediment from the activity. Consider closed or "environmental" buckets, when appropriate.
9. Ensure that disposal sites are being properly managed (e.g., disposal site marking buoys, inspectors, the use of sediment capping and dredge sequencing) and monitored (e.g., chemical and toxicity testing, benthic recovery) to minimize impacts associated with dredge material.
10. Use sequential dredging to avoid dredging activity during specific time periods in particularly environmentally sensitive areas of large navigation channel dredging projects. This can avoid turbidity and sedimentation, bottom disruption, and noise in sensitive areas used by fishery resources during spawning, migration, and egg development.
11. Require appropriate monitoring to avoid and minimize individual and cumulative impacts of the disposal operations.
12. Use seasonal restrictions when appropriate to avoid temporary impacts to habitat during critical life history stages (e.g., spawning, egg and embryo development, and juvenile growth). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements. Resource managers should incorporate

adequate time for habitat recovery of affected functions and values to levels required by managed species.

Fish Waste Disposal

Introduction

Fish waste or material resulting from industrial fish processing operations from either wild stocks or aquaculture consists of particles of flesh, skin, bones, entrails, shells, or process water (i.e., liquid “stickwater” or “gurry”). The organic components of fish waste have a high biological oxygen demand and, if not managed properly, can pose environmental and health problems. Generally, the solid wastes make up 30-40% of total production, depending on the species processed (IMO 2005a). Most fish wastes degrade rapidly in warm weather and can cause aesthetic problems and strong odors as a result of bacterial decomposition if not stored properly or disposed of quickly. Because these waste streams are generally required to be pretreated and fully processed on-site, disposed at a suitable upland site, or sent through municipal sewage treatment, at sea disposal is no longer widely employed in the northeastern United States. However, these materials are sometimes discharged at sea, when appropriate.

Permitting of at sea disposal should be coordinated with appropriate federal and state agencies. Processors should contact the US EPA to determine whether federal permits are necessary for the activity. In order to determine if a federal permit applies, the US EPA must determine if the material constitutes an environmental risk or is a traditional and acceptable "fish waste" disposal defined under Section 102(d) of the Ocean Dumping Ban Act, 33 U.S.C. Part 1412(d) and the regulations promulgated at 40 C.F.R. Part 220. Generally, permits are not required for the transportation or the ocean disposal of fish waste unless: 1) disposal is proposed in harbors or other protected and enclosed waters, and the location is deemed by the EPA as potentially endangering human health, the marine environment or ecological systems; or 2) the waste contains additives or disinfectants from the processing or treatment. In these cases, National Pollutant Discharge Elimination System (NPDES) permits may be required if chlorine or other similar chemicals are used. If an environmental or human health risk is determined, the applicant may be required to submit an assessment of the disposal area and potential impacts to marine resources and follow disposal guidelines consistent with the provisions of the London Convention 1972 (IMO 2005a). Permits required for ocean disposal of fish wastes define the discharge rate of the fluids, residual tissue, and hard part pieces by using a dispersion model. Inputs to the model include discharge flow rate, tissue dimensions, mixing rates, local current patterns, and the specific gravity of the solids (USEPA 2005c). The US EPA may also consult with applicable federal and state regulatory and resource agencies and regional fisheries councils, to identify any areas of concern with respect to the disposal area and activity. Persons wishing to dispose of fish wastes in the ocean may be required to submit specific dilution modeling in support of the proposed disposal and participate in monitoring to verify the results of the modeling (USEPA 2005c).

Bivalve shells, when brought ashore and processed, are not allowed to be returned to the ocean for the purpose of waste disposal. Reuse of the shells as “cultch” in oyster farming operations is a standard, traditional fishing practice in the northeastern United States and does not require permitting, but prior to disposal the shells may be required to meet water quality criteria, principally regarding residual tissue volume.

The guidelines established by the London Convention 1972 place emphasis on progressively reducing the need to use the sea for dumping of wastes. Implementation of these guidelines and the regulations promulgated by US EPA for the disposal of fish wastes includes consideration of potential waste management options that reduce or avoid fish waste to the disposal stream. For example, applications for disposal should consider reprocessing to fishmeal, composting, production of silage (i.e., food for domestic animals/aquaculture), use in biochemical industry products, use as fertilizer in land farming, and reduction of liquid wastes by evaporation (IMO 2005a).

Introduction of pathogens

Ocean disposal of fish wastes has the potential to introduce pathogens to the marine ecosystem that could infect fish and shellfish. In particular, aquaculture operations that raise nonnative species or those that provide food to animals derived from nonindigenous sources could introduce disease vectors to native species (IMO 2005a). However, the disposal guideline provisions implemented as part of the Ocean Dumping Ban Act is designed to ensure wide dispersion of the gurry and limited accumulation of soft parts waste on the sea floor. Models developed to predict the effects of authorized discharges of fish wastes were designed to avoid the accumulation of biodegradable materials on the seafloor and introduction of pathogens.

Release of nutrients/eutrophication

The organic components of fish wastes have a high biological oxygen demand (BOD) and if not managed properly could result in nutrient over-enrichment and reductions in the dissolved oxygen. In ocean disposal, these affects may be seen with mounding of wastes, subsequent increases in BOD and contamination with bacteria associated with partly degraded organic wastes (IMO 2005a). However, disposal guidelines require that dumpsite selection criteria maximizes waste dispersion and consumption of the wastes by marine organisms.

Release of biosolids

Generally, the solid wastes generated by fish waste disposal comprises approximately 30-40% of total production, depending upon the species processed (IMO 2005a). Biosolid waste at fish disposal sites could result in nutrient over-enrichment and reduced dissolved oxygen concentration. However, the disposal guideline provisions implemented as part of the Ocean Dumping Ban Act require wide dispersion of the gurry and limited accumulation of soft parts waste on the sea floor.

Alteration of benthic habitat

Ocean disposal of fish wastes that fail to meet permit conditions and guidelines have the potential to degrade fishery habitat by adversely affecting the productivity and ecological functions of the benthic community. Concentration and mounding of wastes can increase the BOD and reduce dissolved oxygen concentration of an area resulting in reductions in the ability to support small consumer organisms such zooplankton and amphipods. This can then affect species at higher trophic levels that depend upon these consumers for food. However, disposal guidelines require dump-site selection criteria that maximize waste dispersion and consumption of the wastes by marine organisms and disposal monitoring that ensures permit conditions are met (USEPA 2005c). In addition, guidelines and permit review must consider chemical contamination of the marine environment from the waste disposal. For example, the potential presence of chemicals used in aquaculture and fish wastes subjected to chemical treatment must be assessed prior to disposal (IMO 2005a).

Behavioral effects

The presence of biodegradable tissue in the water column has the potential to alter the behavior of organisms in various ways, such as causing an attractant source for scavengers. This could alter the diet of individuals and interfere with trophic-level energy dynamics and community structure. The discharge of process water and biosolid wastes should be monitored carefully to ensure conditions within state and federal permits are met.

Conservation measures and best management practices for disposal of fish wastes

1. Consider the practical availability of alternative methods of disposal to reuse, recycle, or treat the waste as a comparative risk assessment involving both ocean dumping and alternatives.
2. Perform site assessments of the proposed ocean disposal location prior to dumping, including the water depths, average velocities of tidal and nontidal currents, prevailing winds throughout the year, sediment and benthic habitat types, and nature of the sea floor (depositional versus dispersive). Information collected in the site assessment will be used in predictive models developed for the waste disposal activities. Existing uses of the site should be assessed, such as commercial and recreational fishing and whale watching vessels.
3. Use predictive models for plume dispersion and waste settlement based upon physical dynamics of the disposal area, nature of the fish waste, and the method of disposal. The models should be used to assess the probability of the waste plume reaching nearshore coastal waters or other protected areas, such as marine sanctuary waters. The models should also estimate the mass flux of nitrogen and organic carbon associated with the proposed discharges on a daily and annual basis, and how this input may affect phytoplankton production and benthic communities.
4. Dispose material at a steady rate while the vessel maintains headway speed (e.g., 3 nautical miles per hour) as opposed to dumping the entire load at once in a fixed location in order to provide better dilution of fish waste.
5. Grind organic materials to appropriate sizes (e.g., 0.5 inch) prior to discharge where they will be consumed or degraded in the water column dispersion field during and subsequent to their discharge. The intent should be to avoid water quality degradation and tissue deposition and accumulation on the seafloor.
6. Ensure that the waste will be rendered biologically inert during its residence time in the water column and avoid adverse effects on water quality, including reductions in dissolved oxygen concentrations and nutrient over-enrichment.
7. Require monitoring of the waste plume during and after discharge to verify model outputs and advance the knowledge regarding the practice of at-sea disposal of fish processing wastes.

Vessel Disposal

Introduction

When vessels are no longer needed, there are several options for their disposition, including reuse of the vessel or parts of the vessel, recycling or scrapping, creating artificial reefs, and disposal on land or sea (USEPA 2006). This section discusses the potential habitat and marine fisheries impacts associated with disposal at sea.

The disposal of vessels in the open ocean is regulated by the US EPA under section 102(a) of the MPRSA (Ocean Dumping Ban Act) and under 40 CFR § 229.3 of the US EPA regulations. In part, these regulations require that (1) vessels sink to the bottom rapidly and permanently and that marine navigation is not otherwise impaired by the sunk vessel; (2) all vessels shall be disposed of in depths of at least 1,000 fathoms (6,000 feet) and at least 50 nautical miles from land; and (3) before sinking, appropriate measures shall be taken to remove to the maximum extent practicable all materials which may degrade the marine environment, including emptying of all fuel tanks and lines so that they are essentially free of petroleum and removing from the hulls other pollutants and all readily detachable material capable of creating debris or contributing to chemical pollution.

The US EPA and US Department of Transportation Maritime Administration have developed national guidance, including criteria and best management practices for the disposal of ships at sea when the vessels are intended for creation or addition to artificial reefs (USEPA 2006). Vessels disposed of to create artificial reefs have historically been designed and intended to enhance fishery resources for recreational fishermen. However, in recent years artificial reefs have been constructed for a number of nonextractive purposes such as: (1) recreational SCUBA diving opportunities; (2) socioeconomic benefits to local coastal communities; (3) increase habitat to reduce user pressure on nearby natural reefs; (4) reduce user conflicts (e.g., diving in heavily fished areas), and; (5) provide mitigation or restoration to habitat loss for commercial activities (e.g., beach nourishment, dredging, pipeline routes) (NOAA 2007). Some vessels may be sunk to provide a combination of these purposes. Vessels prepared for use as artificial reefs should: (1) be "environmentally sound" and free from hazardous and potentially polluting materials; (2) have had resource assessments for the disposal locations conducted to avoid adverse impacts to existing benthic habitats; and (3) have had stability analyses for the sinking and the ship's ultimate location conducted to ensure there is minimal expectation of adverse impacts on adjacent benthic habitats. Several guidance documents have been developed for the planning and preparation of vessels as artificial reef material, including the National Artificial Reef Plan (NOAA 2007), Coastal Artificial Reef Planning Guide (ASMFC and GSMFC 1998), the Guidelines for Marine Artificial Reef Materials (ASMFC and GSMFC 2004), and the National Guidance: Best Management Practices for Preparing Vessels Intended to Create Artificial Reefs (USEPA 2006). These documents should be consulted to ensure that conflicts with existing uses of the potential disposal site/artificial reef site are addressed and that materials onboard the vessel do not adversely impact the marine environment. Section 203 of the National Fishing Enhancement Act of 1984 (Title II of P.L. 98-623, Appendix C) established that artificial reefs in waters covered under the Act shall "be sited and constructed, and subsequently monitored and managed in a manner which will: (1) enhance fishery resources to the maximum extent practicable; (2) facilitate access and utilization by US recreational and commercial fishermen; (3) minimize conflicts among competing uses of waters covered under this title and the resources in such waters; (4) minimize environmental risks and risks to personal health and property; and (5) be consistent with generally accepted principles of international law and shall not create any unreasonable obstruction to navigation."

The appropriate siting is vital to the overall success of an artificial reef. Considerations and options for site placement and function in the environmental setting should be carefully weighed to ensure program success. Since placement of a reef involves displacement and disturbance of the existing habitat, and building the reef presumably accrues some benefits that could not exist in the absence of the reef, documentation of these effects should be brought out in the initial steps to justify artificial reef site selection. Placement of a vessel to create an artificial reef should: (1) enhance and conserve targeted fishery resources to the maximum extent practicable; (2) minimize conflicts among competing uses of water and water resources; (3) minimize the potential for environmental risks related to site location; (4) be consistent with international law and national fishing law and not create an obstruction to navigation; (5) be based on scientific information; and (6) conform to any federal, state, or local requirements or policies for artificial reefs (USEPA 2006).

The Coastal Artificial Reef Planning Guide (ASMFC and GSMFC 1998) state that when an artificial reef has been constructed, another important phase of reef management begins: monitoring and maintenance. Monitoring provides an assessment of the predicted performance of reefs and assures that reefs meet the general standards established in the Section 203 of the National Fishing Enhancement Act as listed above. It also ensures compliance with the conditions of any authorizing permits. Artificial reef monitoring should be linked with performance objectives, which ensures that NOAA National Marine Fisheries Service responsibilities to protect, restore, and manage living marine resources, and to avoid and minimize any adverse effects on these resources are fulfilled.

Release of contaminants

Ships disposed of at sea, including those intended to create artificial reefs, are often military and commercial vessels which typically contain various materials that, if released into the marine environment, could have adverse effects on the marine environment. Some of the materials of concern include fuels and oil, asbestos, polychlorinated biphenyl (PCB), paint, debris (e.g., vessel debris, floatables, introduced material), and other materials of environmental concern (e.g., mercury, refrigerants) (USEPA 2006). Depending upon the nature of the contaminant and the concentration and duration of the release of contaminant(s) adverse effects to marine organisms may be acute or chronic and either lethal or sublethal. Some contaminants, such as PCB and mercury, can be persistent and bioaccumulate in the tissues of organisms resulting in more serious impacts in higher trophic level organisms. The Ocean Dumping Ban Act and the various guidance documents available for offshore disposal of vessels prohibit materials containing contaminants which may impact the marine environment. The guidance documents provide detailed best management practices regarding recommended measures to remove and abate contaminants contained within and as part of a vessel.

Release of debris

Debris, including solids and floatables, are materials that could break free from a vessel during transportation to the disposal site, and during and after sinking. The release of debris can adversely affect the ecological and aesthetic value of the marine environment. Debris released from vessels is generally categorized into vessel debris (material that was once part of the vessel) and clean-up debris (material that was not part of the vessel but was brought on board the vessel during preparation for disposal).

Some debris released from vessels is not highly degradable and can be persistent in the marine environment for long periods of time, increasing the threat it poses to the environment. Some of the impacts associated with debris include: (1) entanglement and/or ingestion, leading to injury, infection, or death of marine animals that may be attracted to or fail perceive the debris in the water; (2) alteration of the benthic floral and faunal habitat structure, leading to injury or mortality or indirect impact to other species linked in the benthic food web and; (3) elevation of the risk of spills and other environmental impacts caused by impacts with other vessels (e.g., hull damage, damage to cooling or propulsion systems) (USEPA 2006). The Ocean Dumping Ban Act and the various guidance documents available for offshore disposal of vessels require all debris to be removed from vessels prior to sinking. The guidance documents provide detailed best management practices regarding recommended measures to remove vessel and clean-up debris.

Conversion of substrate/habitat and changes in community structure

Vessels that are sunk for the purpose of discarding obsolete or decommissioned ships, as well as those sunk to create an artificial reef, can convert bottom habitat type and alter the ecological balance of marine communities inhabiting the area. For example, placement of vessels over sand bottom can change niche space and predator/prey interactions for species or life history stages utilizing that habitat type. Large structures such as ships tend to attract adult fish and larger predators, which may increase predation rates on smaller and juvenile fish or displace smaller fish and juveniles to other areas (USEPA 2006). Large, anthropogenic structures, such as oil and gas platforms in the Gulf of Mexico, have been shown to affect the distribution of larval and juvenile fish (Lindquist et al. 2005). In addition, large structures tend to provide proportionally less shelter for demersal fishes and invertebrates than smaller, lower profile structures, while the surfaces of steel hull vessels are less ideal for colonization by epibenthos than are natural surfaces like rock (ASMFC and GSMFC 2004). Certain types of habitat and areas may be more susceptible to physical and chemical impacts from the placement of vessels, particularly those vessels sunk as artificial reefs. Generally, vessels sunk for disposal only are located in deeper water (> 6,000 feet) and very far offshore (> 50 nautical miles from land) and may have less impacts on sensitive benthic habitats. However, vessels sunk as artificial reefs are usually located in nearshore coastal waters that also support or are frequented by marine resources that may be adversely impacted by the placement of the structure. Artificial reefs should not be sited in sensitive areas that contain coral reefs or other reef communities, submerged aquatic vegetation, or habitats known to be utilized by endangered or threatened species (USEPA 2006). The Ocean Dumping Ban Act prohibits vessel disposal in areas that may adversely effect the marine environment.

Changes in bathymetry and hydrodynamics

The location of a vessel on the ocean bottom will change the bathymetry and can potentially alter the current flow of the disposal area. A proposed disposal site should be assessed as to the effects the vessel disposal and subsequent bathymetry change may have on the hydrodynamics and geomorphology of the immediate and adjacent habitats. For example, even small vessels placed on the bottom can alter currents and create turbulence around the vessel that may scour existing soft substrates and adversely affect adjacent habitats and communities. In addition, the high vertical profile may cause some vessels to be prone to movement and structural damage from ocean currents and wave surge during storm events. For example, during Hurricane Andrew, a category 5 storm, in south Florida during 1992, nearly all steel-hulled vessels sunk as artificial reefs in the area of the storm's path sustained structural damage, and a number moved 100-700 m because of the storm surge (ASMFC and GSMFC 2004). The movement of vessels after disposal can impact adjacent habitats and relocate the vessels to areas that could alter the ecological balance of marine communities in the area. In addition, reductions in navigational clearance, either as a result of the vessel being sunk in the wrong location and in an area too shallow or because later movement of the vessel from storm surge or currents may increase the potential danger to vessel navigation (e.g., hull damage, damage to cooling or propulsion systems) which may cause further damage from oil/fuel spills or groundings (ASMFC and GSMFC 2004). To minimize the risk of alterations to the bathymetry and hydrodynamics of the disposal area and vessel movement, the Ocean Dumping Ban Act and the various guidance documents available for offshore disposal of vessels require a number of evaluations prior to dumping activities, including: (1) stability analyses; (2) assessments of the seabed, including topography and geological characteristics and; (3) assessment of mean direction and velocity of currents and storm-wave induced bottom currents (ASMFC and GSMFC 2004; IMO 2005b).

Deployment impacts

Some risks to the marine environment exist during the deployment (i.e., sinking) of vessels for disposal or as an artificial reef. Some potential impacts that may occur during deployment include the release of contaminants accidentally left onboard the vessel, damage to adjacent benthic habitats from anchors and cables used to maintain the vessel position as it sinks, impacts to benthic habitats from a vessel accidentally sinking in an unintended location while being towed or from movement of the ship after deployment (ASMFC and GSMFC 2004). However, careful planning during the assessment stages and adherence to operational protocols can avoid impacts during deployment.

Conservation measures and best management practices for disposal of vessels

1. Require that a vessel disposal site assessment adequately characterize the physical and biological environment of the site. In addition to identifying the habitat types and species utilizing the area and targeted for enhancement, ecological investigations should include community settlement and recruitment and predator/prey dynamics and anticipated changes in competition and niche space as a result of the vessel disposal (USEPA 2006).
2. Identify the locations of any sensitive marine habitats in the area. Potential vessel disposal sites should generally not be located near any of the following marine resources: coral reefs; significant beds of aquatic vegetation or macroalgae; oyster reefs; scallop, mussel, or clam beds; existing live bottom (i.e., marine areas supporting sponges, sea fans, corals, or other sessile invertebrates generally associated with rock outcrops); and habitats of endangered or threatened species (federal and state listed) (USEPA 2006).
3. Conduct vessel stability analysis to ensure the vessel is retained in the intended location, including characterization of anticipated weather conditions, tidal dynamics, mean direction and velocity of surface and bottom drifts and storm-wave induced currents, and general wind and wave characteristics (IMO 2005b).
4. Ensure that a thorough inventory and assessment of all potential contaminants on the vessel are completed and that all preplacement cleaning and inspections are completed thoroughly and effectively.
5. Avoid the use of explosives to the extent possible in sinking vessels under 150 feet in length where alternate methods (e.g., opening seacocks, flooding with pumps, etc.) are feasible (ASMFC and GSMFC 2004).
6. Monitor the disposal operation and the placement site for adherence to permit compliance and performance objectives.
7. Ensure that physical and biological monitoring plans for vessels disposed of as artificial reefs are developed as appropriate and that monitoring and reporting requirements are met throughout the designed timeframe.

References for Offshore Dredging and Disposal

[ASMFC], [GSMFC] Atlantic and Gulf States Marine Fisheries Commissions. 1998. Coastal artificial reef planning guide. Washington (DC): Joint Artificial Reef Technical Committee of the Atlantic and Gulf States Marine Fisheries Commissions. 45 p.

[ASMFC], [GSMFC] Atlantic and Gulf States Marine Fisheries Commissions. 2004. Guidelines for marine artificial reef materials, 2nd ed. Washington (DC): Subcommittees of the Atlantic and Gulf States Marine Fisheries Commissions. Number 121. 198 p.

Barr BW, Wilk SJ. 1994. Sewage sludge and industrial waste dumping. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 41-4.

Battelle. 2004. Designating an ocean disposal site for dredged material. [Internet]. Columbus (OH): Battelle Memorial Institute; [cited 2007 Dec 21]. Available from: http://www.battelle.org/Environment/publications/EnvUpdates/summer2004/article1.stm.

Berry WJ, Hinchey EK, Rubinstein NI, Klein-MacPhee G. 2004. Winter flounder, *Pseudopleuronectes americanus*, hatching success as a function of burial depth in the laboratory. In: Ninth flatfish biology conference-poster presentation; 2004 Dec 1-2; Westbrook, CT. Woods Hole (MA): Northeast Fisheries Science Center Reference Document 04-13.

Bohlen WF, Howard-Strobel MM, Cohen DR, Morton ET. 1996. An investigation of the dispersion of sediments resuspended by dredging operations in New Haven Harbor. Waltham (MA): New England Division, US Army Corps of Engineers. DAMOS Contribution #112. 12 p + appendices.

Breitburg DL. 1988. Effects of turbidity on prey consumption by striped bass larvae. Transactions of the American Fisheries Society 117(1):72-7.

Buchholtz ten Brink MR, Manheim FT, Bothner MH. 1996. Contaminants in the Gulf of Maine: what's there and should we worry? In: Dow D, Braasch E, editors. The health of the Gulf of Maine ecosystem: cumulative impacts of multiple stressors. Hanover (NH): Dartmouth College, Regional Association for Research on the Gulf of Maine (RARGOM) Report 96-1. p 91-5.

Burlas M, Ray GL, Clarke D. 2001. The New York District's biological monitoring program for the Atlantic Coast of New Jersey, Asbury Park to Manasquan Section beach erosion control project. [Internet]. Final Report. Vicksburg (MS): US Army Corps of Engineers, New York and US Army Engineer Research and Development Center, Waterways Experiment Station. [cited 2007 Dec 21]. Available from: http://www.nan.usace.army.mil/business/prjlinks/coastal/asbury/index.htm.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New

England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Drucker BS, Waskes W, Byrnes MR. 2004. The US Minerals Management Service outer continental shelf sand and gravel program: environmental studies to assess the potential effects of offshore dredging operations in federal waters. Journal of Coastal Research 20(1):1-5.

Fredette TJ, French GT. 2004. Understanding the physical and environmental consequences of dredged material disposal: history in New England and current perspectives. Marine Pollution Bulletin 49(1-2):93-102.

Fredette TJ, Kullberg PG, Carey DA, Morton RW, Germano JD. 1993. Twenty-five years of dredged material disposal site monitoring in Long Island Sound: a long-term perspective. In: Van Patten MS, editor. Proceedings of the Long Island Sound Research Conference; 1992 Oct 23-24; New Haven, CT. Groton (CT): Connecticut Sea Grant Program, Publication No. CT-SG-93-03, p 153-61.

Germano JD, Rhoads DC, Lunz JD. 1994. An integrated, tiered approach to monitoring and management of dredged material sites in the New England region. Waltham (MA): US Army Corps of Engineers, New England Division. DAMOS Contribution No. 87, Report No. SAIC-90/7575&234. 55 p + appendices.

Hildebrand J. 2004. Sources of anthropogenic sound in the marine environment. [Internet]. In: Vos E, Reeves RR, editors. Report of an international workshop: policy on sound and marine mammals; 2004 Sep 28-30; London, England. Bethesda (MD): Marine Mammal Commission. [cited 2008 Jul 21]. 129 p. Available from: http://www.mmc.gov/sound/internationalwrkshp/pdf/hildebrand.pdf.

Hitchcock DR, Bell S. 2004. Physical impacts of marine aggregate dredging on seabed resources in coastal deposits. Journal of Coastal Research 20(1):101-114.

[ICES] International Council for the Exploration of the Sea. 1992. Report of the ICES working group on the effects of extraction of marine sediments on fisheries. Copenhagen (Denmark): ICES Cooperative Research Report # 182.

[IMO] International Maritime Organization. 2005a. Specific guidelines for assessment of fish wastes, or material resulting from industrial fish processing operations. [Internet]. London (England): IMO, Office for the London Convention 1972. [cited 2007 Dec 21]. 12 p. Available from: http://www.imo.org/includes/blastDataOnly.asp/data_id%3D17022/3-FishWaste.pdf.

[IMO] International Maritime Organization. 2005b. Specific guidelines for assessment of vessels. [Internet]. London (England): IMO, Office for the London Convention 1972. [cited 2007 Dec 21]. 13 p. Available from: http://www.imo.org/includes/blastDataOnly.asp/data_id%3D17023/4-Vessels.pdf.

Johnson SW, Stanley DR, Moles DA. 1998a. Effects of submarine mine tailings disposal on juvenile yellowfin sole (*Pleuronectes asper*): a laboratory study. Marine Pollution Bulletin 36(4):278-87.

Johnson SW, Stone RP, Love DC. 1998b. Avoidance behavior of ovigerous Tanner crabs (*Chionoecetes bairdi*) exposed to mine tailings: a laboratory study. Alaska Fishery Research Bulletin 5:39-45.

Kenny AJ, Rees HL. 1996. The effects of marine gravel extraction on the macrobenthos: results 2 years post-dredging. Marine Pollution Bulletin 32(8-9):615-22.

Kurland JM, Ludwig FM, Gorski SW, Mantazaris C. 1994. Dredging and dredged-material disposal. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 44-7.

Lindquist DC, Shaw RF, Hernandez FJ Jr. 2005. Distribution patterns of larval and juvenile fishes at offshore petroleum platforms in the north-central Gulf of Mexico. Estuarine, Coastal and Shelf Science 62(4):655-65.

Lohrer AM, Wetz JJ. 2003. Dredging-induced nutrient release from sediments to the water column in a southeastern saltmarsh tidal creek. Marine Pollution Bulletin 46(9):1156-63.

Malins DC, editor. 1977. Effects of petroleum on arctic and subarctic marine environments and organisms. New York (NY): Academic Press.

Messieh SN, Wildish DJ, Peterson RH. 1981. Possible impact from dredging and spoil disposal on the Miramichi Bay herring fishery. St. Andrews (New Brunswick): Canadian Technical Report of Fisheries and Aquatic Sciences. No. 1008. iv + 33 p.

[MMS] Minerals Management Service. 2005a. Marine Minerals Program, offshore energy and minerals management. [Internet]. Herndon (VA): MMS, [updated 2007 Jun 28; cited 2007 Dec 21]. Available from: http://www.mms.gov/sandandgravel/.

[MMS] Minerals Management Service. 2005b. Offshore discharges from oil and gas development operations – FAQ. Gulf of Mexico Region, Offshore Information. [Internet]. New Orleans (LA): MMS. [updated 2001 June 6; cited 2007 Dec 21]. Available from: http://www.gomr.mms.gov/homepg/offshore/egom/factshee.html.

[NOAA] National Oceanic and Atmospheric Administration. 2007. National Artificial Reef Plan (as amended): guidelines for siting, construction, development, and assessment of artificial reefs. Washington (DC): US Department of Commerce, National Oceanic and Atmospheric Administration. 51 p.

Newcombe CP, Jensen JO. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4):693-727.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol. 1.

O'Reilly JE. 1994. Nutrient loading and eutrophication. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 25-30.

Pacheco A. 1984. Seasonal occurrence of finfish and larger invertebrates at three sites in Lower New York Harbor, 1981-1982. Final report. Sandy Hook (NJ): NOAA/NMFS. Special report for USACE, New York District. 53 p.

Packer DB, Griesbach SJ, Berrien PL, Zetlin CA, Johnson DL, Morse WW. 1999. Essential fish habitat source document: summer flounder, *Paralichthys dentatus*, life history and habitat characteristics. [Internet]. Woods Hole (MA): NOAA, NMFS NER, NEFSC. NOAA Technical Memorandum NMFS-NE-151. [cited 2008 Jul 22]. 88 p. Available from: http://www.nefsc.noaa.gov/nefsc/publications/tm/tm151/tm151.pdf.

Palermo MR, Clausner JE, Rollings MP, Williams GL, Myers TE, Fredette TJ, Randall RE. 1998. Guidance for subaqueous dredged material capping. Vicksburg (MS): US Army Corps of Engineers, WES, Dredging Operations and Environmental Research (DOER). Technical Report DOER-1. 302 p.

Pearce JB. 1994. Mining of seabed aggregates. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum. NMFS-NE-106. p 48-50.

Rhoads DC. 1994. Analysis of the contribution of dredged material to sediment and contaminant fluxes in Long Island Sound. Special Technical Report. Waltham (MA): US Army Corps of Engineers, New England Division, Regulatory Division. DAMOS Contribution No. 88. 32 p.

Rhoads DC, Germano JD. 1982. Characterization of organism-sediment relations using sediment profile imaging: an efficient method of remote ecological monitoring of the seafloor (REMOTS ™ System). Marine Ecology Progress Series 8:115-28.

Rhoads DC, Germano JD. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia 142(1):291-308.

Rhoads DC, Kullberg PG, Fredette TJ. 1995. Dredged material disposal within an estuary: is it a significant source of sediment and contaminant fluxes? The case of Long Island Sound. In:

Knott MA, editor. Proceedings of the Conference for Ports '95, Volume 1; 1995 Mar 13-15; Tampa, FL. Reston (VA): American Society of Civil Engineers. p 428-39.

Scarrat DJ. 1987. Fisheries' interests and ocean mining. Marine Mining 6:141-7.

Stevenson DK, Scott ML. 2005. Essential fish habitat source document: Atlantic herring, *Clupea harengus*, life history and habitat characteristics (2nd ed). [Internet]. Woods Hole (MA): NOAA, NMFS, Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-NE-192. [cited 2008 Jul 22]. 94 p. Available from: http://www.nefsc.noaa.gov/nefsc/habitat/efh.

Stewart PL, Arnold SH. 1994. Environmental requirements Atlantic herring (*Clupea harengus harengus*) in eastern Canada and its response to human impacts. Dartmouth (Nova Scotia): Canadian Technical Report of Fisheries and Aquatic Sciences 2003: ix + 37p.

[URI] University of Rhode Island Coastal Institute. 2003. Designation of a long-term offshore disposal site for dredged material. [Internet]. [cited 2007 Dec 21]. Available from: http://www.ci.uri.edu/Projects/DD/.

[USACE] US Army Corps of Engineers. 1982. Final Supplemental Environmental Impact Statement for the designation of a disposal site for dredged material in Western Long Island Sound WLIS III. Waltham (MA): USACE, New England Division.

[USACE] US Army Corps of Engineers. 2001. Monitoring at the New London Disposal Site 1992-1998 Vol. I. [Internet]. Concord (MA): USACE, New England District. DAMOS Contribution No. 128, SAIC No. 515. [cited 2008 Jul 22]. 157 p + appendices. Available from: http://www.nae.usace.army.mil/damos/pdf/128.pdf.

[USACE] US Army Corps of Engineers. 2005a. HARS – Historic Area Remediation Site. [Internet]. [cited 2007 Dec 21]. Available from: http://www.nan.usace.army.mil/business/prjlinks/dmmp/benefic/hars.htm.

[USACE] US Army Corps of Engineers. 2005b. Managed dredged material disposal sites in the North Atlantic Division. [Internet]. [updated 2007 Apr 12; cited 2007 Dec 21]. Available from: http://www.nae.usace.army.mil/damos/maps.asp.

[USEPA] US Environmental Protection Agency. 1986. Ocean dumping site designation delegation handbook for dredged material. Washington (DC): US EPA Office of Marine and Estuarine Protection.

[USEPA] US Environmental Protection Agency. 1997. Supplemental to the Environmental Impact Statement on the New York dredged material disposal site designation for the designation of the Historic Area Remediation Site (HARS) in the New York Bight. New York (NY): US EPA.

[USEPA] US Environmental Protection Agency. 2003a. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: PAH mixtures. [Internet]. Washington (DC): US EPA Office of Research and

Development. Report No. EPA-600-R-02-013. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/PAHESB.pdf.

[USEPA] US Environmental Protection Agency. 2003b. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: dieldrin. [Internet]. Washington (DC): US EPA Office of Research and Development. Report No. EPA-600-R-02-010. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/dieldrin.pdf.

[USEPA] US Environmental Protection Agency. 2003c. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: endrin. [Internet]. Washington (DC): US EPA Office of Research and Development. Report No. EPA-600-R-02-009. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/endrin.pdf.

[USEPA] US Environmental Protection Agency. 2004. Rhode Island region long-term dredged material disposal site evaluation project. Boston (MA): US EPA, New England Region.

[USEPA] US Environmental Protection Agency. 2005a. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: metal mixtures (cadmium, copper, lead, nickel, silver, and zinc). [Internet]. Washington (DC): US EPA Office of Research and Development. Report No. EPA-600-R-02-011. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/metalsESB_022405.pdf.

[USEPA] US Environmental Protection Agency. 2005b. Ocean dumping in the Southeast. [Internet]. [updated 2007 Nov 21; cited 2007 Dec 21]. Atlanta (GA): US EPA, Region 4. Available from: http://www.epa.gov/region4/water/oceans/.

[USEPA] US Environmental Protection Agency. 2005c. Disposal of fish waste. [Internet]. [updated 2007 Nov 1; cited 2007 Dec 21]. New York (NY): US EPA, Region 2. Available from: http://www.epa.gov/region2/water/oceans/fwaste.htm.

[USEPA] US Environmental Protection Agency. 2006. National guidance: best management practices for preparing vessels intended to create artificial reefs. [Internet]. Washington (DC): US EPA Office of Wetlands, Oceans, and Watersheds. EPA Report 842-B-06-002. [cited 2008 Jul 24]. 77 p. Available from: http://www.epa.gov/owow/oceans/habitat/artificialreefs/documents/0605finalreefguidance.pdf.

van Dalfsen JA, Essink K, Toxvig Madsen H, Birklund J, Romero J, Manzanera M. 2000. Differential responses of macrozoobenthos to marine sand extraction in the North Sea and the Western Mediterranean. Copenhagen (Denmark): ICES Journal of Marine Science 57:1439-45.

Wilber D, Brostoff W, Clarke D, Ray G. 2005. Sedimentation: potential biological effects of dredging operations in estuarine and marine environments. [Internet]. DOER Technical Notes Collection. Vicksburg (MS): US Army Engineer Research and Development Center.

ERDC TN-DOER-E20. [cited 2008 Jul 22]. 14 p. Available from: http://el.erdc.usace.army.mil/dots/doer/pdf/doere20.pdf.

Wilber DH, Clarke DG. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21(4):855-75.

Wilber DH, Clarke DG, Ray GL, Burlas M. 2003. Response of surf zone fish to beach nourishment operations on the northern coast of New Jersey, USA. Marine Ecology Progress Series 250:231-46.

Wilk SJ, Barr BW. 1994. Multiple-use issues in estuarine and coastal habitat loss. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 51-3.

CHAPTER SEVEN: CHEMICAL EFFECTS—WATER DISCHARGE FACILITIES

Introduction

Disposal of various waste materials into rivers, estuaries, and marine waters is not a modern phenomenon; this practice has been used as a preferred disposal option virtually since the beginning of human civilization (Ludwig and Gould 1988; Islam and Tanaka 2004). Nevertheless, when the full spectrum of emissions from land-based activities is taken into account, the use of coastal waters as a repository for anthropogenic waste has not previously been practiced on as large or intense a global scale as in recent decades (Williams 1996). In the United States, growing human population densities in coastal communities have manifested a demonstrably adverse effect on aquatic resources. The scientific literature is replete with evidence of inorganic and organic pollutant accumulation in coastal waters from anthropogenic effluents (e.g., Ragsdale and Thorhaug 1980; Tessier et al. 1984; Phelps et al. 1985; Long E et al. 1995; Pastor et al. 1996; Smith et al. 1996; Chapman and Wang 2001; Hare et al. 2001; O'Connor 2002; Robinet and Fenteun 2002; Wurl and Obbard 2004). The federal Clean Water Act (CWA), enacted in 1972 to address many of these issues, eliminated certain types of disposal activities and otherwise induced improvements to the nation's surface water quality. Nonetheless, "despite reductions in pollution from municipal and industrial point sources more than one-third of the river miles, lake acres, and estuary square miles suffer [*sic*] some degree of impairment" (Ribaudo et al. 1999). To the extent that it may alter natural processes and natural resource communities, unabated degradation of the aquatic environment caused by a wide spectrum of human activities poses consequences for fishery resources and their habitats.

Contaminants enter our waterways through two generic vectors: point and nonpoint sources. Pollutants of nonpoint source origins tend to enter aquatic systems as relatively diffuse contaminant streams primarily from atmospheric and terrestrial sources (see Coastal Development chapter of this report for discussions on nonpoint source pollution). In contrast, point source pollutants generally are introduced via some type of pipe, culvert, or similar outfall structure. These discharge facilities typically are associated with domestic or industrial activities, or in conjunction with collected runoff from roadways and other developed portions of the coastal landscape. Waste streams from sewage treatment facilities and watershed runoff in many urbanized portions of the northeastern United States are first intermingled and then subsequently released into aquatic habitats via combined sewer overflows (CSOs). Such point discharges collectively introduce a cocktail of inorganic and organic contaminants into aquatic habitats, where they may become bioavailable to living marine resources.

While all pollutants can become toxic at high enough levels, there are a number of compounds that are toxic even at relatively low levels. The US Environmental Protection Agency (US EPA) has identified and designated more than 126 analytes as "priority pollutants." According to the US EPA, "priority pollutants" of particular concern for aquatic systems include: (1) dichlorodiphenyl trichloroethane (DDT) and its metabolites; (2) chlorinated pesticides other than DDT (e.g., chlordane and dieldrin); (3) polychlorinated biphenyl (PCB) congeners; (4) metals (e.g., cadmium, copper, chromium, lead, mercury); (5) polycyclic aromatic hydrocarbons (PAHs); (6) dissolved gases (e.g., chlorine and ammonium); (7) anions (e.g., cyanides, fluorides, and sulfides); and (8) acids and alkalis (Kennish 1998; USEPA 2003a). While acute exposure to these substances produce adverse effects of aquatic biota and habitats, chronic exposure to low concentrations probably is a more significant issue for fish population structure and may result in multiple substances acting in “an additive, synergistic or antagonistic manner” that may render impacts relatively difficult to discern (Thurberg and Gould 2005).

Determining the eventual fate and effect of naturally occurring and synthetic contaminants in coastal environments and biota is a highly dynamic proposition that requires interdisciplinary evaluation. It is essential that all processes sensitive to pollutants be identified and that investigators realize that the resulting adverse effects may be manifested at the biochemical level in organisms (Luoma 1996) in a manner particular to the species or life stage exposed. Pollutant exposure can inhibit: (1) basic detoxification mechanisms, like production of metallothioneins or antioxidant enzymes; (2) the ability to resist diseases; (3) the ability of individuals or populations to counteract pollutant-induced metabolic stress; (4) reproductive processes including gamete development and embryonic viability; (5) growth and successful development through early life stages; (6) normal processes including feeding rate, respiration, osmoregulation; and (7) overall Darwinian fitness (Capuzzo and Sassner 1977; Widdows et al. 1990; Nelson et al. 1991; Stiles et al. 1991; Luoma 1996; Thurberg and Gould 2005).

The nature and extent of a pollutant's dispersal in our waterways are collectively dependent on a variety of factors including site-specific ecological conditions, the physical state in which the contaminant is introduced into the aquatic environment, and the inherent chemical properties of the substance in question. Soluble or miscible substances typically enter waterways in an aqueous phase and eventually become adsorbed onto organic and inorganic particles (Wu et al. 2005); however, contaminants may enter aquatic systems as either particle-borne suspensions or as solutes (Bishop 1984; Turner and Millward 2002). Dilution and settling out from such effluent streams initially are dictated by physical factors (e.g., the presence of significant currents or perhaps a strong thermocline or pycnocline) which predominantly influence the spatial extent of contaminant dispersal. In particular, turbulent mixing, or diffusion, disperses contaminant patches in coastal waters resulting in larger, comparatively diluted contaminant distributions further away from the initial point source (Bishop 1984). Biological activity and geochemical processes subsequently intercede and typically result in contaminant partitioning between the aqueous and particulate phases (Turner and Millward 2002).

While physical dispersion, biological activity, and other ecological factors clearly have important roles regarding the distribution of contaminants in aquatic habitats, contaminant partitioning is largely governed by certain ambient environmental conditions, notably salinity, pH, and the physical nature of local sediments (Turekian 1978; McElroy et al. 1989; Turner and Millward 2002; Leppard and Droppo 2003; Wu et al. 2005). Highly reactive suspended particles typically serve as important carriers of aquatic contaminants and largely are responsible for their bioavailability, transport, and ecological fate as they become dispersed in receiving waters (Turner and Millward 2002). In addition, hyporheic (i.e., the saturated zone under a river or stream, comprising substrate with the interstices filled with water) exchange between overlying water and groundwater can alter salinity, dissolved oxygen concentration, and other water chemistry aspects in ways that can influence the affinity of local sediment types for particular contaminants or otherwise affect contaminant behavior (Ren and Packman 2002).

Amendments to the CWA include important provisions to address acute or chronic water pollution emanating from discharge pipes and outfalls under the National Pollutant Discharge Elimination System (NPDES) program. Until the late 1980s, the NPDES program traditionally focused efforts on controlling industrial and municipal sewage discharges but has since expanded its purview to include storm water management (USEPA 1996). While the NPDES program has led to ecological improvements in waters of the United States, point sources continue to introduce pollutants into the aquatic environment, albeit at reduced levels. Nonetheless, studies demonstrate that particle-associated contaminants collected in coastal depositional areas are preserved in chronological strata or horizons (Huntley et al. 1995; Chillrud et al. 2003). Consequently, historically deposited contaminants may be encountered when installing new outfalls or coastal infrastructure, especially near urbanized areas. Regardless of whether these pollutants were deposited recently or decades ago, dredging incidental to construction and related activities that enhance their potential biological availability can have adverse ecological implications.

The environmental dynamics of point source wastes are complex and involve a variety of physical, chemical, and biological processes simultaneously acting on the introduced suite of contaminants and their surrounding habitat. Because of the many competing variables involved, it is difficult to predict the ultimate fate and effects of anthropogenic wastes with great precision; however, local habitat characteristics in combination with the relative solubility, degree of hydrophobicity (i.e., tending to repel and not absorb water), and chemical reactivity of the introduced substances are important determining factors at the most basic level of analysis.

To minimize redundancy, all recommended conservation measures and best management practices for sewage discharge facilities, industrial discharge facilities, and combined sewer overflows have been included at the end of this chapter.

Sewage Discharge Facilities

Introduction

Sewage treatment plants introduce a host of contaminants into our waterways primarily through discharge of fluid effluents comprising a mixture of processed "black water" (sewage) and "gray water" (all other domestic and industrial wastewater). Such municipal effluents begin as a complex mixture of human waste, suspended solids, debris, and a variety of chemicals collectively derived from domestic and industrial sources. These contaminants include an array of suspended and dissolved substances, representing both inorganic and organic chemical species (Grady et al. 1998; Epstein 2002). These substances potentially include the full spectrum of EPA priority pollutants mentioned previously and many other contaminants of anthropogenic origin. However, the five constituents that are usually the most important in determining the type of treatment that will be required are: (1) organic content (usually measured as volatile solids); (2) nutrients; (3) pathogens; (4) metals; and (5) toxic organic chemicals (USEPA 1984).

Coastal communities rely on municipal wastewater treatment to contend with potential human health issues related to sewage and also to protect surface and groundwater quality. Municipal processing facilities typically receive raw wastewater from both domestic and industrial sources, and are designed to produce a liquid effluent of suitable quality that can be returned to natural surface waters without endangering humans or producing adverse aquatic effects (Grady et al. 1998; Epstein 2002). As it is currently practiced in the United States, wastewater treatment entails subjecting domestic and industrial effluents to a series of physical, chemical, or even biological processes designed to address or manipulate different aspects of contaminant mitigation. For both logistical and economic reasons, not all municipalities expend the same level of effort removing contaminants from their wastewater before returning it to a receiving aquatic habitat. The following discussion summarizes the different levels that municipal wastewater treatment and resulting water quality benefits derived from them.

Primary treatment, also known as "screen and grit," is only marginally effective at addressing sewage contaminants and simply entails bulk removal of "settleable" solids from the wastewater by sedimentation and filtration. Sometimes total suspended solids are further reduced in the initial effluent treatment phase by implementing another level of primary treatment, which entails using chemicals to induce coagulation and flocculation of smaller particles (Parnell 2003).

The resulting bio-solids must be disposed, and their final disposition could entail composting with subsequent use in agricultural applications, placement in a landfill, disposal at sea, or even incineration (Werther and Ogada 1999). Removal and appropriate disposal of sewage present in a solid phase are important steps, if elementary, in addressing human health and aesthetic issues surrounding sewage management because doing so removes visible substances that otherwise would accumulate in the aquatic environment at or near the discharge point. Unfortunately, primary treatment of municipal wastewater alone often fails to meet overall environmental goals of supporting important water-dependent uses like fishery resource production and recreational uses featuring primary contact with the water. As a consequence, coastal communities in the northeastern region process their wastewater through one or more additional treatment levels beyond bulk solids removal to address the environmental challenges of their sewage effluents more effectively.

Following bulk sludge removal, sewage treatment plants typically pass the highly organically-enriched water emerging from primary treatment through a second process that is intended to address biological oxygen demand (BOD), an indirect measure of the concentration of biologically degradable material present in organic wastes that reflects the amount of oxygen necessary to break down those substances in a set time interval. Such secondary treatment, which is required for all municipal wastewater treatment in the United States, involves removal of much of the remaining organic material by introducing aerobic microorganisms under oxygen-enriched conditions (Parnell 2003). The resulting microbial action breaks organic substrates into progressively simpler compounds, with the final waste components predominantly released as carbon dioxide. The bacteria subsequently are removed by chlorination before the secondarily-treated effluent is released into local surface waters or the secondarily treated wastewater is directed to another part of the sewage treatment plant for additional processing. Where practiced, such effluent-polishing or advanced treatment measures use any of several techniques to remove inorganic nitrogenous or phosphorous salts to reduce the final effluent's potential to cause excessive nutrient enrichment of the receiving waters (Epstein 2002; Parnell 2003).

Because of the large expense of tertiary sewage treatment, the public sector does not implement it as a uniform municipal wastewater treatment policy. Consequently, while secondary treatment is the standard operating procedure for municipal wastewater treatment in the northeastern United States, natural resource managers cannot assume that advanced, tertiary treatment is available to meet desired environmental goals. Recent point source management policy decisions by Boston, MA, area communities are a case in point. Rather than implementing more costly advanced treatment during system upgrades, these communities chose to address local municipal wastewater challenges by implementing primary and secondary treatment combined with source reduction of certain contaminants and offshore diversion of outfalls to encourage enhanced effluent dilution (Moore et al. 2005). Despite the added expense of implementing them, both secondary and advanced treatment processes are important potential habitat protection measures, particularly because they mitigate oxygen depletion events, eutrophication, and related phenomena that can result in adverse ecological conditions.

Release of nutrients and eutrophication

Particularly under lesser levels of treatment, municipal sewage facilities discharge large volumes of nutrient-enriched effluent. While some level of readily available nutrients are essential to sustain healthy aquatic habitats and ecological productivity, excess concentrations result in eutrophication of coastal habitats. Elevated nitrogen and phosphorous concentrations in municipal wastewater effluents can cause pervasive ecological responses including: exaggeration of phytoplankton and macroalgal populations; initiation of harmful algal blooms (Anderson et al. 2002); adverse affects on the physiology, growth, and survival of certain ecologically important aquatic plants (Touchette and Burkholder 2000); reduction of water transparency with accompanying adverse effects to submerged and emergent vascular plants or other disruptions to the normal ecological balance among vascular plants and algae (Levinton 1982; Cloern 2001); hypoxic or anoxic events that may cause significant fish and invertebrate mortalities; disturbances to normal denitrification processes; and concomitant decrease in local populations of fishery resources and forage species (USEPA 1994). Sewage outfalls also may become an attraction nuisance in that they may at least initially attract fish around the point of discharge until hypoxia, toxin production, and algal bloom development render the aquatic area less productive (Islam and Tanaka 2004). Collectively, adverse chemical effects may be especially significant to aquatic resources in temperate regions because strong thermoclines and persistent ice cover restrict vertical mixing and exacerbate deteriorating habitat conditions at depth.

For additional information on the mechanisms involved in denitrification of organic and inorganic compounds, Korom's (1992) review of denitrification in natural aquifers is a concise and informative compilation of heterotrophic and autotrophic denitrifiers.

Release of contaminants

Municipal treatment facilities discharge large volumes of effluent into the aquatic environment. The waste stream typically contains a complex mixture of domestic and industrial wastes that contain predominantly natural and synthetic organic substances, metals, and trace elements, as well as pathogens (Islam and Tanaka 2004). Similarly, introductions of certain pharmaceuticals via municipal wastewater discharges have become causes for concern because of their potential to act as endocrine disruptors in fish and other aquatic resources. Residence time of the different contaminant classes in aquatic environments is an important habitat management consideration. Some of these substances, such as volatile organic compounds, may have a relatively short residence time in the system and other, more persistent substances, such as synthetic organometallic compounds, may linger for decades after becoming associated with the substrate or concentrated in local biota. Such pollution has been associated with mortality, malformation, abnormal chromosome division, and higher frequencies of mitotic abnormality in adult fish from polluted areas compared with those from less polluted regions of the northwest Atlantic Ocean (Longwell et al. 1992).

Increased concentrations of the various contaminant classes associated with municipal wastewater can be highly ecologically significant. For instance, exposure to contaminants within these categories have been correlated with deleterious effects on aquatic life including larval deformities in haddock (*Melanogrammus aeglefinus*) (Bodammer 1981), reduced hatching success and increased larval mortality in winter flounder (*Pseudopleuronectes americanus*) (e.g., Klein-MacPhee et al. 1984; Nelson et al. 1991), skeletal deformities in Atlantic cod (*Gadus morhua*) (Lang and Dethlefsen 1987), inhibited gamete production and maturation in sea scallops (*Placopecten magellanicus*) (Gould et al. 1988), and reproductive impairment in Atlantic cod (Thurberg and Gould 2005).

Laboratory experiments with pesticides have shown a positive relationship between malformation and survival of embryos and larvae of Atlantic cod and concentration of DDT and its breakdown product dichlorodiphenyl dichloroethylene (DDE) (Dethlefsen 1976). The proportion of fin erosion in winter flounder collected on contaminated sediments was found to be greater in fish sampled with higher concentrations of PCB in muscle, liver, and brain tissues than in fish collected in reference sites (Sherwood 1982). Studies conducted in the harbor of New Haven, CT, found high occurrences of liver lesions, blood cell abnormalities, liver DNA damage, and liver neoplasms among winter flounder with high concentrations of organic compounds, metals, and PCB in their gonads (Gronlund et al. 1991). Such pollution also has been associated with mortality, malformation, abnormal chromosome division, and higher frequencies of mitotic abnormality in adult fish from polluted areas compared with those from less polluted regions of the northwest Atlantic Ocean (Longwell et al. 1992). Observed effects of fish exposed to PAH include decrease in growth, cardiac disfunction, lesions and tumors of the skin and liver, cataracts, damage to immune systems, estrogenic effects, bioaccumulation, bioconcentration, trophic transfer, and biochemical changes (Logan 2007).

For almost a century, sewage sludge (the solids extracted from raw wastewater during sewage treatment) was disposed of at sea. In the northeastern United States, a number of designated offshore sewage sludge dumpsites existed, including one in Boston Harbor, MA, and sites in the New York Bight and the Mid-Atlantic Bight (Barr and Wilk 1994). Not surprisingly, sediments sampled in the vicinity of sewage sludge dumpsites have contained higher levels of contaminants (e.g., PCB, PAH, chlorinated pesticides, and metals) than in control sites (Barr and Wilk 1994). Sewage sludge has been demonstrated to have adverse effects on aquatic organisms. For example, early life stages of Atlantic herring (*Clupea harengus*) have shown a series of developmental abnormalities, including premature hatching accompanied by reduced viability of emerging fry; poor larval survival; smothering or incapacitation of larvae by particle flocs; and fin damage (Urho 1989; Costello and Gamble 1992). The Ocean Dumping Ban Act of 1988 prohibited sewage sludge and industrial wastes from being dumped at sea after December 31, 1991. This law is an amendment to the Marine Protection, Research, and Sanctuaries Act of 1972, which regulates the dumping of wastes into ocean waters.

In addition to these diverse contaminant classes, wastewater facilities also discharge a host of synthetic hormones or other substances that could disrupt normal endocrine function in aquatic vertebrates, as well as introduce zoonotic viruses, bacteria, and fungi that may be present in raw human sewage. These chemicals act as "environmental hormones" that may mimic the function of the sex hormones (Thurberg and Gould 2005). Adverse effects include reduced or altered reproductive functions, which could result in population-level impacts. Metals, PAHs, and other contaminants have been implicated in disrupting endocrine secretions of marine organisms (Brodeur et al. 1997; Thurberg and Gould 2005). However, the long-term effect of endocrine-disrupting substances on aquatic life is not well understood and demands serious attention by the scientific and resource policy communities. Refer to the Endocrine Disruptors subsection of this chapter for a broader discussion on this topic. Metals such as mercury are also capable of moving upward through trophic levels and can accumulate in fish (i.e., bioaccumulation) at levels which may cause health problems in human consumers.

While modern sewage treatment facilities undeniably reduce the noxious materials present in raw wastewater and some substances typical of processed effluents have their own inherent toxic effects, it also is important to recognize that secondary and advanced treatment can alter the chemistry of ordinarily benign materials in ways that initiate or enhance their toxicity. In particular, normally nonhazardous organic compounds present in wastewater potentially can be rendered toxic when raw municipal effluent is chlorinated in the sewage treatment process (NRC 1980; Epstein 2002). Other contaminants may become toxic to humans or many different aquatic resource taxa when these substances are methylated (addition of a $-CH_4$ group) or otherwise after having been chemically transformed into a harmful, biologically available molecular form.

The behavior and effects of trace chemicals in aquatic systems largely depend on the speciation and physical state of the pollutants in question. A detailed description concerning contaminant partitioning and bioavailability is beyond the scope of this technical discussion.

However, Gustafsson and Gschwend (1997) offer an excellent review of the matter in terms of how dissolved, colloidal and settling particle phases affect trace chemical fates and cycling in aquatic environments. While the observations provided by these Massachusetts Institute of Technology researchers pertain specifically to cycling of compounds in natural waters, the generic properties they discuss also would apply in the context of substances in treated wastewater since they are subject to the same physical and chemical forces. In addition, Tchobanoglous et al. (2002) may be consulted for an authoritative technical review of the environmental engineering aspects of wastewater treatment.

Exposure to potentially mutagenic or teratogenic pollutants and the resulting declines in viability at any life stage reduce the likelihood of maturation and eventual recruitment to adulthood or a targeted fishery. Literature on the aqueous and sedimentary geochemistry and physiological effects of contaminants on aquatic biota should be consulted to determine the fate of persistent compounds in local sediments and associated pore-water and the extent of acute or chronic toxic effects on affected aquatic biota (Varanasi 1989; Allen 1996; Langmuir 1996; Stumm and Morgan 1996; Tessier and Turner 1996; Paquin et al. 2003).

Alteration of water alkalinity

Municipal sewage effluent that does not meet water quality standards can alter the alkalinity of riverine receiving waters. However, freshwater and low-salinity waters with low buffering capacity are more susceptible to acidification than are marine waters. Acidification of riverine habitats has been linked to the disruption of reproduction, development, and growth of anadromous fish (USFWS and NMFS 1999; Moring 2005). For example, osmoregulatory problems in Atlantic salmon (*Salmo salar*) smolts have been related to habitats with low pH (Staurnes et al. 1996). In estuarine waters, low pH has been shown to cause cellular changes in the muscle tissues of Atlantic herring which may lead to a reduction in swimming ability (Bahgat et al. 1989). However, all municipal sewage facilities are required to obtain water quality permits through the US EPA's NPDES program and must meet established pH standards for receiving waters. Acid precipitation from atmospheric sources is of concern in the northeastern United States. Refer to the Global Effects and Other Impacts chapter for more information regarding acid precipitation.

Impacts to submerged aquatic vegetation

Submerged aquatic vegetation (SAV) requires relatively clear water in order to allow adequate light transmittance for metabolism and growth. Sewage effluent containing high concentrations of nutrients can lead to severely eutrophic conditions. The resulting depression of dissolved oxygen and diminished light transmittance through the water may result in local reduction or even extirpation of SAV beds that are present before habitat conditions become too degraded to support them (Goldsborough 1997). Examples of large scale SAV declines have been seen throughout the eastern coastal states, most notably in Chesapeake Bay, MD/VA, where overall abundance has been reduced by 90% during the 1960s and 1970s (Goldsborough 1997). Although a modest recovery of the historic SAV distribution has been seen in Chesapeake Bay over the past few decades, reduced light penetration in the water column from nutrient enrichment and sedimentation continues to impede substantial restoration. Primary sources of nutrients into Chesapeake Bay include fertilizers from farms, sewage treatment plant effluent, and acid rain (Goldsborough 1997). Short and Burdick (1996) correlated eelgrass losses in Waquoit Bay, MA, with anthropogenic nutrient loading primarily as a result of increased number of septic systems from housing developments in the watershed.

Eutrophication can alter the physical structure of SAV by decreasing the shoot density and blade stature, decreasing the size and depths of beds, and stimulating excessive growth of macroalgae (Short et al. 1993). An epidemic of an eelgrass wasting disease wiped out most eelgrass beds along the east coast during the 1930s, and although some of the historic distribution of eelgrass has recovered, eutrophication may increase the susceptibility of eelgrass to this disease (Deegan and Buchsbaum 2005).

Reduced dissolved oxygen

The decline and loss of fish populations and habitats because of low dissolved oxygen concentrations is "one of the most severe problems associated with eutrophication in coastal waters" (Deegan and Buchsbaum 2005). The effect of chronic, diurnally fluctuating levels of dissolved oxygen has been shown to reduce the growth of young-of-the-year winter flounder (Bejda et al. 1992). High nutrient loads into aquatic habitats can cause hypoxic or anoxic conditions, resulting in fish kills in rivers and estuaries (USEPA 2003b; Deegan and Buchsbaum 2005) and potentially altering long-term community dynamics (NRC 2000; Castro et al. 2003). Highly eutrophic conditions have been reported in a number of estuarine and coastal systems in the northeastern United States, including Boston Harbor, Long Island Sound, NY/CT, and Chesapeake Bay (Bricker et al. 1999). For the southern portions of the northeast coast (i.e., Narragansett Bay, RI, to Chesapeake Bay), O'Reilly (1994) described chronic hypoxia (low dissolved oxygen) as a result of coastal eutrophication in several systems. This author reported episodic, low dissolved oxygen conditions in some of the northern portions of the northeast coast, such as in Boston Bay/Charles River and the freshwater portion of the Merrimack River, MA/NH (O'Reilly 1994). Areas particularly vulnerable to hypoxia are those that have restricted water circulation, such as coastal ponds, subtidal basins, and salt marsh creeks (Deegan and Buchsbaum 2005). While any system can become overwhelmed by unabated nutrient inputs or nutrient enrichment, the effects of these generic types of pollution when experienced in temperate regions may be especially significant in the summer. This is primarily a result of stratification of the water column and higher water temperatures and metabolic rates during summer months (Deegan and Buchsbaum 2005).

Siltation, sedimentation, and turbidity

Municipal sewage outfalls, especially those that release untreated effluent from storm drains, can release suspended sediments into the water column and the adjacent benthic habitat. Increased suspended particles within aquatic habitats can cause elevated turbidity levels, reduced light transmittance, and increased sedimentation of benthic habitat which may lead to the loss of SAV, shellfish beds, and other productive fishery habitats. Other affects from elevated suspended particles include respiration disruption of fishes, reduction in filtering efficiencies and respiration of invertebrates, disruption of ichthyoplankton development, reduction of growth and survival of filter feeders, and decreased foraging efficiency of sight-feeders (Messieh et al. 1991; Barr 1993).

Introduction of pathogens

Pathogens are generally a concern to human health because of consumption of contaminated shellfish and finfish and exposure at beaches and swimming areas (USEPA 2005). Microorganisms entering aquatic habitats in sewage effluents do pose some level of biological risk since they have been shown to infect marine mammals (Oliveri 1982; Bossart et al. 1990; Islam and Tanaka 2004). The degree to which anthropogenically-derived microbes may affect fish, shellfish, and other aquatic taxa remains an important research topic; however, some recently published observations concerning groundfish populations near the Boston sewage outfall into Massachusetts Bay are suggesting that appropriate management practices may address at least part of this risk (Moore et al. 2005). See also the chapters on Coastal Development and Introduced/Nuisance Species and Aquaculture for more information on the introduction of pathogens.

Introduction of harmful algal blooms

Sewage treatment facilities releasing effluent with a high BOD that may enter estuarine and coastal habitats have been associated with harmful algal bloom events, which can deplete the oxygen in the water during bacterial degradation of algal tissue and result in hypoxic or anoxic "dead zones" and large-scale fish kills (Deegan and Buchsbaum 2005). There is evidence that nutrient overenrichment has led to increased incidence, extent, and persistence of nuisance and/or 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 plankton algae (O'Reilly 1994).

Algal blooms may also contain species of phytoplankton such as dinoflagellates that produce toxins. Toxic algal blooms, such as red tides, can decimate large numbers of fish, contaminate shellfish beds, and cause health problems in humans. Shellfish sequester toxins from the algae and become dangerous to consume. Toxic algal blooms could increase in the future because many coastal and estuarine areas are currently moderately to severely eutrophic (Goldburg and Triplett 1997). Heavily developed watersheds tend to have reduced stormwater storage capacity, and the high flow velocity and pulse of contaminants from freshwater systems can have long-term, cumulative impacts to estuarine and marine ecosystems. Some naturally occurring microorganisms, 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 will require additional research, nutrient enrichment of coastal waters is suspected to play a role (Buck et al. 1997). See also the chapter on Introduced/Nuisance Species and Aquaculture for more information on harmful algal blooms.

Impacts to benthic habitat

As discussed above, treated sewage effluent containing high concentrations of nutrients can lead to severely eutrophic conditions that can reduce or eliminate SAV beds (Goldsborough 1997). In addition, municipal sewage outfalls can release suspended sediments into the water column and the adjacent benthic habitat. Increased suspended particles within aquatic habitat can cause elevated turbidity levels, reduced light transmittance, which may lead to the reduction or loss of SAV, shellfish beds and other productive benthic habitats.

Changes in species composition

Treated sewage effluent can contain, at various concentrations, nutrients, toxic chemicals, and pathogens that can affect the health, survival, and reproduction of aquatic organisms. These effects may lead to alterations in the composition of species inhabiting coastal aquatic habitats and can result in community and trophic level changes (Kennish 1998). For example, highly eutrophic water bodies have been found to contain exaggerated phytoplankton and macroalgal populations that can lead to harmful algal blooms (Anderson et al. 2002). Sewage treatment facilities may initially attract fish around the point of discharge until hypoxia, toxin production, and algal bloom development render the aquatic area less productive (Islam and Tanaka 2004). Reduced light penetration in the water column from nutrient enrichment and sedimentation has been shown to contribute to the loss of eelgrass beds in coastal estuaries in southern Massachusetts, Long Island Sound, and the Chesapeake Bay (Goldsborough 1997; Deegan and Buchsbaum 2005).

Contaminant bioaccumulation and biomagnification

Sewage discharges can contain metals and other substances known to be toxic to marine organisms. Not surprisingly, the bays and estuaries of highly industrialized urban areas in northeastern US coastal areas, such as Boston Harbor, Portsmouth Harbor, NH/ME, Newark Bay,NJ, western Long Island Sound, and New York Harbor, have shown relatively high metal burdens in sampled sediments (Larsen 1992; Kennish 1998; USEPA 2004a). While industrial outfalls are responsible for metal contamination in some areas, sewage has been identified as one of the primary sources. For example, although lead contamination in coastal sediments can originate from a variety of sources, sewage is believed to be the primary source of silver contamination (Buchholtz ten Brink et al. 1996). Metals may move upward through trophic levels and accumulate in fish and some invertebrates (bioaccumulation) at levels which can eventually cause health problems in human consumers (Kennish 1998; NEFMC 1998). Other chemicals are known to bioaccumulate and biomagnify in the ecosystem, including pesticides (e.g., DDT) and PCB congeners (Kennish 1998). The National Coastal Condition Report (USEPA 2004a) reported that after metals, PCB congeners and DDT metabolites were responsible for most of the contaminant criteria exceedances in northeast coast samples. For example, sediment samples collected by NOAA's National Status and Trends (NS&T) Program found in some samples very high concentrations of chlorinated hydrocarbons such as PCBs, pesticides, and dioxins from the lower Passaic River, NJ, and Newark Bay in the Hudson-Raritan estuary (Long ER et al. 1995). Other locations in this estuary containing moderately to highly toxic samples in the NS&T Program included Arthur Kill, NY/NJ, and East River, NY.

Release of pharmaceuticals

Concerns have been emerging over the past few years regarding the continual exposure of aquatic organisms to the complex spectrum of active ingredients in pharmaceuticals and personal care products (PPCP), which can persist in treated effluent from sewage facilities. PPCPs comprise thousands of chemical substances, including prescription and over-the-counter therapeutic drugs, veterinary drugs, fragrances, lotions, and cosmetics (Daughton and Ternes 1999; USEPA 2007). The concentrations of PPCP in the aquatic environment are generally detected in the range of parts per thousand to parts per billion and may not pose an acute risk. However, aquatic organisms may be adversely affected because they can have continual and multigenerational exposures, exposures at high concentrations from untreated water, and they may have low dose effects (Daughton and Ternes 1999; USEPA 2007). Some of these PPCPs include steroid compounds, which may act as endocrine disruptors by mimicking the functions of sex hormones (refer to the subsection below for more information on endocrine disruptors). The effects of antibiotics and antimicrobial drugs on aquatic organisms are also of concern. Although population level effects on aquatic organisms from PPCPs are inconclusive at this time, the growing evidence on this topic suggests further investigation is warranted.

Endocrine disruptors

Another recent topic of concern involves a group of chemicals, called "endocrine disruptors," which interfere with the endocrine system of aquatic organisms. Growing concerns have mounted in response to the effects of endocrine-disrupting chemicals on humans, fish, and wildlife (Kavlock et al. 1996; Kavlock and Ankley 1996). These chemicals act as "environmental hormones" that may mimic the function of the sex hormones androgen and estrogen (Thurberg and Gould 2005). Adverse effects include reduced or altered reproductive functions, which could result in population-level impacts. Several studies have implicated endocrine-disrupting chemicals with the presence of elevated levels of vitellogellin in male fish, a yolk precursor protein that is normally only found in mature female fish (Thurberg and Gould 2005). Some of the chemicals shown to be estrogenic include PCB congeners, dieldrin, DDT, phthalates, and alkylphenols (Thurberg and Gould 2005), which have had or still have applications in agriculture and may be present in irrigation water and storm water runoff. Metals have also been implicated in disrupting endocrine secretions of marine organisms, potentially disrupting natural biotic processes (Brodeur et al. 1997).

In summary, the chemical implications of sewage treatment plant effluents vary as a function of the effort taken to remove organic and inorganic contaminants collected by the wastewater treatment plant. Further complicating matters, while secondary treatment is the minimal acceptable standard treatment process in the northeastern United States, inadequately treated or even raw wastewater containing human sewage and attendant debris routinely passes into the aquatic environment from municipal processing plant outfalls when the flow and/or storage demands exceed design specifications. Such releases are commonly experienced when older sewer systems are inundated, particularly in conjunction with storm events. Accordingly, the types of treatment processes implemented, how effectively the wastewater treatment infrastructure is operating, and the salinity of the receiving waters (to the extent that it influences contaminant chemistry) are critical variables when considering the chemical implications of releasing treated wastewater into the aquatic environment.

Maintenance activities associated with sewage discharge facilities

Maintenance activities associated with sewage treatment plants typically involve periodic application of chemicals to treat piping for colonization of biofouling organisms. Efforts to control fouling communities can produce larger field or even chronic disturbances that could adversely affect the aquatic environment. Under some circumstances, chemical treatments are not necessary and fouling communities may be removed mechanically using hot water under pressure. When this type of procedure is implemented, most of the direct impacts are physical. Although the use of pressurized, hot freshwater to mechanically remove fouling organisms may temporarily alter salinity and solute loads, some localized indirect thermodynamic effects that alter ambient chemistry could also occur in the dispersal plume until ambient temperature is restored. In addition, differences in the chemical composition of the source and receiving waters would be expected to have at least a minimal effect, particularly when chlorinated water is used to facilitate the removal of fouling organisms and when there is a significant difference in salinity between cleaning and receiving waters. Perhaps more typically, colonization of fouling communities is controlled through periodic use of antifouling paints, coatings, or other treatments. When conducted inappropriately, periodic applications of these substances can have chronic and potentially harmful effects in the aquatic environment.

Fortunately, application of biocides in aquatic systems is regulated under the CWA, which includes provisions to protect fishes and many invertebrate species to the extent practicable. Since local salinity ranges and diffusion rates at the outfall are important considerations in terms of eventual dispersion and relative toxicity of outfall maintenance materials, these and similar site-specific considerations often dictate which products may be used safely at a given project site. It is vital that only products designed and federally approved for use in and near aquatic habitats are deliberately allowed to enter US waterways under any circumstances.

In general, the most deleterious effects of sewage outfall maintenance probably revolve around fouling community control measures. That is because the underlying intent of such practices is to remove a large variety of plant, animal, and even bacterial populations from inhabiting the area surrounding the outfall. Biocide applications control undesirable organisms by chemical or biological means (Knight and Cooke 2002). Whether removed chemically or mechanically, the loss of these organisms at least initially may result in other forms of local ecological disturbance, such as reduced productivity and diminished prey and cover (Meffe and Carroll 1997). While outfall maintenance events individually result in an acute chemical impact to the environment and biota, it is important also to consider the cumulative effects of repeated applications over a project's maintenance cycle. Especially when undertaken regularly, the maintenance of outfall structures can create a chronic cycle of disturbance on resident biota, particularly sessile organisms.

Individual biocides and other contaminants released during outfall maintenance operations may have direct effects on local aquatic biota or they may act in an additive, synergistic, or antagonistic manner in concert with ambient physical and chemical habitat conditions. Such exposure to organic and inorganic pollutants may result in a spectrum of lethal and sublethal effects that may be discerned at every level of biological organization (Thurberg and Gould 2005). Wide distribution of contaminants, such as biocides and related outfall maintenance substances, can be facilitated through bioaccumulation in motile aquatic organisms that are capable of dispersing between riverine, estuarine, and marine habitats (Mearns et al. 1991). The pollutant-induced effects these substances engender are not limited to biochemical or physiological responses, as they may also disrupt a variety of complex behaviors which may be essential for maintaining fitness and survival (Atchison et al. 1987; Blaxter and Hallers-Tjabbes 1992; Kasumyan 2001; Scott and Sloman 2004).

In addition to measures to control fouling organisms in wastewater treatment facilities, maintenance activities also involve repairs and enhancements of structures associated with the facilities' infrastructure. Because they typically are undertaken on a relatively small scale, physical repairs of existing infrastructure usually produce impacts of lesser intensity and on a more limited spatial scale than those created during initial installation. In contrast, application of antifouling coatings or related treatments not only discourages settlement by aquatic organisms on the treated surface, but also releases biocide into the aquatic environment (Richardson 1997; Terlizzi et al. 2001). Depending on the individual case, such releases can range from very limited to extensive plumes, as measured by the volume of material emitted, and the distance broadcast away from the point source the substance may be detected in the water column.

Collectively, such releases degrade local water quality. Fortunately, chemical effects of sewage outfall maintenance in lotic coastal systems generally would be expected to dissipate relatively quickly because of dispersion by river flow or tidal action. For health and aesthetic reasons, municipal sewage outfalls should not be sited in quiescent waters. In addition, government-established protocols for biological control agents approved for applications in subaqueous discharges generally are applied in isolation within a capped pipe and are subsequently released after sufficient time has passed for the biocide properties to have abated, or more rarely after the bulk of the treating solution is siphoned off and dealt with offsite. Typically, such biocide solutions are designed to decompose into relatively benign constituent forms within hours and, when used properly, are thought not to pose a significant risk to nontarget organisms (Diderich 2002).

As is the case for initial outfall installation impacts, a variety of chemical and biological factors determine the extent to which the polluting substance affects the water column, sediments, and biota and the distance it migrates from the point source. Among them, salinity and carbonate alkalinity (i.e., carbonic acid and bicarbonate ion content) are especially important because of their respective roles in mediating chemical reactions in solution and in conferring the buffering capacity provided by marine and estuarine waters. Carbonate alkalinity, or water hardness, is an especially important property in riverine systems because the ambient carbonate concentrations regulate acid-base chemistry and other water quality parameters, which are thought to be important factors in the recovery of depleted salmonid populations in Maine (Johnson and Kahl 2005). While salmonids are particularly sensitive to degraded water quality, poor water quality is known to affect a wide variety of aquatic organisms (Tessier et al. 1984; Scott and Sloman 2004; Moore et al. 2005; Thurberg and Gould 2005).

Construction impacts associated with sewage discharge facilities

The construction of municipal wastewater outfalls can have chemical effects that result from a number of activities, including releasing suspended sediments and associated pore-water in the construction zone; releasing drill mud or cuttings from a directional drilling operation; discharging substances from mechanized equipment (e.g., incidental discharges of hydrocarbons or hydraulic fluid); and introducing leachate from fresh and curing concrete, antifouling paints, and other construction materials. Contaminants initially reside in aquatic systems in either a dissolved phase in the water column or in a particulate phase when they have adsorbed onto sediments or other solids. Pollutants present in biologically-available forms subsequently become assimilated by aquatic biota and become biomagnified as they are taken up in successive trophic strata (Levinton 1982; Sigel and Sigel 2001).

While plume and sedimentation effects incidental to outfall construction do not always result in a readily observable ecological response, they commonly produce a range of direct and indirect effects to living aquatic resources and their habitats. Not all of the ecological implications of sediment resuspension and transport result in adverse effects to aquatic organisms (Blaber and Blaber 1980). These effects vary a great deal depending on which life history stages are affected (Wilber and Clarke 2001). As a general rule, however, the severity of adverse chemical effects tends to be greatest for early life stages and for adults of some highly sensitive species (Newcombe and Jensen 1996). In particular, predictive models of dose-response relationships corroborate that the eggs and larvae of nonsalmonid estuarine fishes exhibit some of the most sensitive responses to suspended sediment exposures of all the taxa and life history stages for which data are available (Wilber and Clarke 2001). Mitigative measures that limit the nature and extent of chemical impacts arising from outfall installation typically can and should be undertaken to avoid and minimize adverse construction effects.

From the standpoint of water quality, most chemical effects associated with outfall construction should be relatively acute and transitory. Adverse water quality impacts arising from outfall installation generally arise as a consequence of: (1) substances that have adsorbed onto resuspended particles; (2) pollutants that have dissolved or leached into the water column; or (3) contaminants that have been released directly by construction equipment. These pollutants may include substances that lead to nutrient enrichment; they may be chemically reduced; they may exhibit acidic or caustic properties; they may contain organometallic complexes or a variety of other natural or synthetic compounds; they may be hydrophobic or hydrophilic; or they otherwise may exhibit a diverse spectrum of chemical properties that affect their relative toxicity and dispersal in the water column.

While various physical, chemical, and biological factors come into play, the area into which such water quality impacts extend is largely dependent upon the length of time particles and solutes are held in the water column and the distance they are transported from the construction site. Grain size and ambient sediment structure characteristics have an important bearing on dispersal. As benthic material is disturbed during outfall installation and site preparation, resuspended particulate matter would settle predominantly in the immediate project vicinity. Remaining waterborne fractions subsequently would be transported over a distance and direction that are related to the grain size of disturbed sediments, the velocity of local water currents, and local wave action (Neumann and Pierson 1966). Contaminants mobilized in and subsequently deposited by the dispersal plume generated by construction activities are subject to complex biogeochemical processes that ultimately dictate their fate and ecological effects. For example, hydrogen sulfide released with pore-water from disturbed sediments depletes dissolved oxygen and results in locally hypoxic or anoxic conditions in the water column until the area engulfed within the dispersal plume becomes reoxygenated.

While important, it is essential to recognize that local sediment characteristics alone do not determine contaminant introduction or resuspension during outfall installation. The type of construction equipment used to build an outfall structure also has an important influence on the dispersion of disturbed bottom material. For traditional clamshell dredging, Tavolaro (1984) estimates a 2% loss of material through sediment resuspension at the dredge site. It is reasonable to conclude that similar losses would accrue when clamshells are used to install outfall pipes for sewage treatment facilities. In the same way, dredging methods that purposely fluidize sediments to facilitate their removal (e.g., hydraulic dredges, water jets) could result in even greater dispersion of resuspended sediment, especially when local waters are not quiescent or in situations where unfiltered return flow to the waterway is permitted. Since fine depositional sediments tend to have greater contaminant loads than do coarser sediments typical of higher energy areas, the chemical consequences of resuspending fine sediments during outfall installation are potentially greater since they are more likely to be associated with pollutants.

Likewise, water quality implications of outfall construction are not limited to sediment resuspension or releasing pore-water that contains hydrogen sulfide. Secondary vectors of chemical contamination during outfall installation include substances introduced into aquatic habitats by construction equipment and materials. Mechanized construction equipment may inadvertently or incidentally release a broad spectrum of chemicals, fuels, and lubricants into the waterway. Similarly, until the building material has completely cured or has leached out soluble contaminant fractions, subaqueous applications of wet concrete or grout, treated timber products, paints, and other construction materials would all potentially introduce pollutants into the surrounding water.

The chemical implications of constructing municipal outfalls to local substrates ultimately depend on whether (and to what extent) contaminants are released, become associated with, and accumulate in, sediments and surrounding pore-water. While sediment particles naturally exhibit cycles of exchange between the water column and bottom substrate materials (Turner and Millward 2002), dredging or outfall installation can be expected to disturb much deeper sediment horizons in a short period of time than would be expected from storms or in all but the most highly erosion prone coastal areas. As construction equipment disrupts sediment horizons at the project site, some fraction of the benthic substrate becomes resuspended into the water column (Tavolaro 1984).

Outfall construction for sewage treatment facilities can create measurable adverse impacts within the disturbed footprint, including the disruption of ambient sediment stratigraphy, cohesiveness, and geochemistry. These effects have geochemical consequences that may be particularly significant when construction activities are located in depositional or nutrient-enriched areas and where local sediments tend to be fine-grained and contain at least moderate levels of pollution. Regardless of the nature and concentration of substances adsorbed onto the sediment or sequestered in the pore-water, salinity may significantly affect local aqueous conditions, sedimentary geochemistry, and resulting ecological effects.

While it is critical to consider the impacts of outfall construction on physical habitat features, implications for resident and transitory biota also should be taken into account. Excavation and relocation of sediments, which may be performed incidental to outfall installation, would produce a sediment plume and create sedimentation effects that could result in detrimental effects on aquatic resources present in the affected area (Newcombe and Jensen 1996; Wilber and Clarke 2001; Berry et al. 2003; Wilber et al. 2005). Direct and indirect impacts related to the removal of benthic material can elicit a variety of responses from aquatic biota (Wilber and Clarke 2001) which have been addressed elsewhere in this report.

While many potential construction impacts clearly are physical in nature, the chemical effects are complex and may have important implications for biota present in the affected area. In addition to the physicochemical considerations already discussed above, the life history and ecological strategies characteristic of different species also are important considerations in assessing the potential chemical impacts of outfall installation. For instance, while highly motile adult and fish in juvenile life stages of most species could flee when construction is ongoing, those in egg and larval stages and nonmotile benthic organisms could not escape contaminant exposure. While some species like the sessile life stages of eastern oyster (*Crassostrea virginica*) have adapted to withstand some acute habitat disturbances (Galtsoff 1964; Levinton 1982), most benthic and slow-moving species would not be able to escape contaminant exposure and instead would exhibit adaptive physiological and biochemical responses to counter any pollutants present.

Contaminants released during outfall installation activities may have direct effects on local aquatic biota or they may act in an additive, synergistic, or antagonistic manner in concert with ambient physical and chemical habitat conditions. Such exposure to organic and inorganic pollutants may result in a spectrum of lethal and sublethal effects that can be discerned at the organismal, tissue, cellular, and subcellular levels of biological organization (Thurberg and Gould 2005). Wide distribution of contaminants can be facilitated through bioaccumulation in motile aquatic organisms that are capable of dispersing between riverine, estuarine, and marine habitats (Mearns et al. 1991).

Importantly, pollutant-induced effects are not limited to biochemical or physiological responses. Environmental pollutants such as metals, pesticides, and other organic compounds also have been shown to disrupt a variety of complex fish behaviors, some of which may be essential for maintaining fitness and survival (Atchison et al. 1987; Blaxter and Hallers-Tjabbes 1992; Kasumyan 2001; Scott and Sloman 2004). In particular, Kasumyan (2001) provided an excellent review of how chemical pollutants interfere with normal fish foraging behavior and chemoreception physiology, while Scott and Sloman (2004) have focused on the ways metals and organic pollutants have been shown to induce behavioral and physiological effects on fresh water and marine fishes.

Industrial Discharge Facilities

Introduction

Industrial wastewater facilities face many of the same engineering and environmental challenges as municipal sewage treatment plants. Industrial discharge facilities produce a wide variety of trace elements and organic and inorganic compounds. In the industrialized portions of the northeastern United States, such facilities include a variety of chemical plants, refineries, paper mills, defense factories, energy generating facilities, electroplating firms, mining operations, and many other high intensity industrial uses that generate large volumes of wastewater. In many situations, the sanitary and industrial process streams are intermingled and processed at the industrial facility's own treatment plant, requiring that the eventual effluent is treated to address water quality concerns from a fairly broad spectrum of contaminants. While the procedures involved are similar to those implemented at municipal treatment facilities, the specific levels and methods of wastewater treatment at industrial treatment plants vary considerably. While a detailed description of industrial wastewater engineering is well beyond the scope of this report, readers interested in specific technical information may consult portions of Tchobanoglous et al. (2002) or Perry (1997) for more information.

Like sewage plant outfalls, industrial discharge structures are point sources for a variety of environmental contaminants, particularly metals and other trace elements; nutrients; and persistent organic compounds such as pesticides and organochlorines. These substances tend to adhere to solid particles within the waste stream, become adsorbed onto finer sediment fractions once dispersed into coastal waters, and subsequently accumulate in depositional areas. Together with microbial action, local salinity and other properties of the riverine, estuarine, or marine receiving waters may alter the chemistry of these contaminant-particle complexes in ways that render them more toxic than their parent compounds. Upon entering the food web, such contaminants tend to accumulate in benthic organisms at higher concentrations than in surrounding waters (Stein et al. 1995) and may result in various physiological, biochemical, or behavioral effects (Scott and Sloman 2004; Thurberg and Gould 2005).

Release of metals

Industrial discharge structures can release large volumes of effluent containing a variety of potentially harmful substances into the aquatic environment. Metals and other trace elements are common byproducts of industrial processes and as a consequence are anticipated to be components of typical industrial waste streams that may enter the aquatic environment (Kennish 1998). Metals may be grouped into transitional metals and metalloids. Transitional metals, such as copper, cobalt, iron, and manganese, are essential for metabolic function of organisms at low concentrations but may be toxic at high concentrations. Metalloids, such as arsenic, cadmium, lead, mercury, and tin, are generally not required for metabolic function and may be toxic even at low concentrations (Kennish 1998). Metals are known to produce skeletal deformities and various developmental abnormalities in marine fish (Bodammer 1981; Klein-MacPhee et al. 1984; Lang and Dethlefsen 1987). The early life history stages of fish can be quite susceptible to the toxic impacts associated with metals (Gould et al. 1994).

Release of organic compounds

A variety of synthetic organic compounds are released by industrial facilities, find their way into aquatic environments and can be taken up by resident biota. These compounds are some of the most persistent, ubiquitous, and toxic pollutants known to occur in marine ecosystems (Kennish 1998). Organochlorines, such as DDT, chlordane, and PCBs, are some of the most highly toxic, persistent, and well documented and studied synthetic organic compounds. Others include dioxins and dibenzofurans that are associated with pulp and paper mills and wood treatment plants and have been shown to be carcinogenic and capable of interfering with the development of early development stages of organisms (Kennish 1998). Longwell et al. (1992) determined that dozens of different organic contaminants were present in ripe winter flounder eggs. Such accumulation can reduce egg quality and disrupt ontogenic development in ways that significantly depress survival of young (Islam and Tanaka 2004). Organic contaminants, such as PCBs, have been shown to induce external lesions (Stork 1983) and fin erosion (Sherwood 1982) and reduce reproductive success (Nelson et al. 1991) in marine fishes. In addition, suspicion is mounting that exposure to even very low levels of such persistent xenobiotic (i.e., foreign) compounds may disrupt normal endocrine function and lead to reproductive dysfunction such as reduced fertility, hatch rate, and offspring viability in a variety of vertebrates.

Release of petroleum products

Oil, characterized as petroleum and any derivatives, consists of thousands of chemical compounds and can be a major stressor on inshore fish habitats (Kennish 1998). Industrial wastewater, as well as combined wastewater from municipal and storm water drains, contributes to the release of oil into coastal waters. Petroleum hydrocarbons can adsorb readily to particulate matter in the water column and accumulate in bottom sediments, where they may be taken up by benthic organisms (Kennish 1998). Petroleum products consist of thousands of chemical compounds that can be toxic to marine life including PAHs and water-soluble compounds, such as benzene, toluene, and xylene, which can be particularly damaging to marine biota because of their extreme toxicity, rapid uptake, and persistence in the environment (Kennish 1998). PAHs can be toxic to meroplankton, ichthyoplankton, and other pelagic life stages exposed to them in the water column (Kennish 1998). 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). Oil has been demonstrated 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. In general, the early life stages (eggs and larvae) are most sensitive, juveniles are less sensitive, and adults least so (Rice et al. 2000). Refer to the chapters on Coastal Development, Energy-related Activities, and Marine Transportation for additional information on impacts associated with petroleum products and PAH.

Alteration of water alkalinity

A major point of departure when comparing municipal sanitary treatment outfall and industrial plant effluents concerns the ability of some industrial discharges to affect carbonate alkalinity, or buffering capacity, of receiving waters. Both riverine and estuarine strata are particularly susceptible to point source acidification because their low buffering capacity can be quickly overwhelmed by acid discharges; however, even marine habitats can be significantly and adversely affected when continual influx of acidified liquid wastewater outstrips the natural buffering capability of seawater. In riverine systems, it has been postulated that locally reduced pH may be linked to impaired Atlantic salmon recovery (Johnson and Kahl 2005) and osmoregulatory problems (NRC 2004). Oulasvirta (1990) reported periodic massive mortalities of Atlantic herring eggs from effluent containing sulfuric acid and various other metals released at a titanium-dioxide plant in the Gulf of Bothnia, Finland. Low pH in estuarine waters may lead to cellular changes in muscle tissues, which could reduce swimming ability in herring (Bahgat et al. 1989). A variety of industrial operations, ranging from mining and metal production to certain industrial manufacturing activities, is known to release acid effluents that may have adverse effects on fish, shellfish, and their habitat. Collectively, such detrimental impacts can hinder the survival and sustainability of fishery resources and their prey. Point source pollution from industrial sources is currently regulated by the states or the US EPA through the NPDES permit program, which generally does not allow discharges of low pH water into estuaries and coastal waters of the United States.

Release of nutrients and other organic wastes

Industrial facilities that process animal or plant by-products can release effluent with high BOD which may have deleterious affects to receiving waters. Wood processing facilities, paper and pulp mills, and animal tissue rendering plants can release nutrients, reduced sulfur and organic compounds, and other contaminants through wastewater outfall pipes. For example, wood processing plants and pulp mills release effluents with tannins and lignin products containing high organic loads and BOD into aquatic habitats (USFWS and NMFS 1999). The release of these contaminants in mill effluent can reduce dissolved oxygen in the receiving waters. In addition, paper and pulp mills can release a number of toxic chemicals used in the process of bleaching pulp for printing and paper products. The bleaching process may use chlorine, sulfur derivatives, dioxins, furans, resin acids, and other chemicals that are known to be toxic to aquatic organisms (Mercer et al. 1997). These chemicals have been implicated in various abnormalities in fish, including skin and organ tissue lesions, fin necrosis, gill hyperplasia, elevated detoxifying enzymes, impaired liver functions, skeletal deformities, increased incidence of parasites, disruption of the immune system, presence of tumors, and impaired growth and reproduction (Barker et al. 1994; Mercer et al. 1997). Because of concern about the release of dioxins and other contaminants, considerable improvements in the bleaching process have reduced or eliminated the use of elemental chlorine. According to the US EPA, all pulp and nearly all paper mills in the United States have chemical recovery systems in place and primary and secondary wastewater treatment systems installed to remove particulates and BOD (USEPA 2002). Approximately 96% of all bleached pulp production uses chlorine-free bleaching technologies (USEPA 2002).

Construction impacts of industrial discharge facilities

The chemical impacts associated with constructing an industrial discharge are similar to those described for sewage treatment outfalls. Generally, such discharges predominantly entail suspending sediments and releasing pore-water in the construction zone, releasing drill mud or cuttings from horizontal directional drilling equipment, incidental discharges of fuels, lubricants and other substances from mechanized construction equipment, and leachates from construction materials. Since the substances encountered and circumstances of exposure would be the same regardless of the type of outfall being installed, the Construction Impacts Associated with Sewage Discharge Facilities subsection of this chapter should be reviewed for details regarding the impacts to the water column, sediment, and aquatic biota from the construction of industrial discharge facilities.

Maintenance impacts of industrial discharge facilities

The chemical impacts of maintaining industrial discharge facilities are similar to those described for sewage treatment facilities. Generally, the impacts of performing structural repairs are expected to be similar to those experienced during initial outfall installation, but on a lesser scope and magnitude. Impacts associated with the removal and treatment of fouling communities would be similar to those described for the maintenance activities of sewage treatment facilities. The reader should review the previous subsection on Maintenance Activities Associated with Sewage Discharge Facilities for details on the implications of outfall maintenance on the water column, sediment, and aquatic biota.

Combined Sewer Overflow (CSO)

The discussion of point source discharges would be incomplete without mention of CSOs, which are ubiquitous in urban and even suburban areas in New England and the Mid-Atlantic region. For a variety of reasons, many of these municipalities operate wastewater collection systems composed of “separate” and “combined” sewers. “Separate” sewers tend to be newer or replacement installations that have distinct piping components for stormwater and sanitary sewers.

Under storm or other high runoff conditions, the separate sewer system allows excess volumes of storm water to bypass sewage treatment facilities and discharge directly into the receiving water body constraining all sanitary waste to processing at the wastewater treatment plant. This prevents the excess volume of watershed runoff from overwhelming the operating capacity of the treatment facilities. Older systems tend to be "combined" sewer systems that commingle watershed runoff and sanitary waste streams.

Typical CSOs do not discharge effluent under dry conditions but may permit unprocessed sewage under high runoff events to enter the receiving waters completely or partially untreated. This occurs when large volumes of storm water and sewage overwhelm the treatment plant and untreated sewage is discharged prematurely. Some CSO discharges violate state and/or federal water quality standards, and each municipality must develop a plan to control and eliminate these CSOs. There is no precise estimate on the number of CSOs that exist or on how much untreated sewage is discharged from them each year. However, 828 separate NPDES permits were issued by the US EPA in 2004. There were a total 9,348 authorized discharges from CSOs nationally in 2004, with approximately one half located in the northeastern United States and the remaining half in the Great Lakes region (USEPA 2002; USEPA 2004b).

The chemical implications of CSOs are that they are potential sources of very large amounts of untreated nutrients and contaminating chemicals that degrade both the aesthetic and ecological conditions of affected habitats. In addition to the adverse effects mentioned for the other outfall types, CSOs can be important point sources for pesticides, herbicides, fertilizers, and other substances commonly applied to terrestrial habitats, ranging from rural farmland and suburban yards or golf courses to highly urbanized centers. In addition, they are sources of terrestrial particulates and may be a secondary source of atmospherically-deposited pollutants that have settled anywhere in the local watershed. While impacts associated with nonpoint sources are discussed elsewhere in this report, the sanitary sewer component of CSO effluents can be construed as an extension of the preceding discussions for municipal and industrial outfalls. The net effect of permitting untreated domestic wastewater to enter the receiving waterway is to diminish the effectiveness of wastewater treatment elsewhere. In so doing, CSOs contribute to increased pollution levels and related natural resource impairments. It is not possible to measure the resulting habitat damage and accompanying aquatic resource degradation in isolation from nonpoint pollution. However, it is important that resource managers consider that CSO discharges can and will occur and account for the added pollutant loads they generate when setting permissible local effluent limits or establishing priorities for replacing outmoded urban infrastructure.

Construction and maintenance impacts of CSOs

The chemical impacts associated with construction and maintenance activities in CSOs are similar to those described for sewage treatment and industrial discharge facilities. Generally, discharges associated with construction activities may include releasing contaminants associated with suspended sediments, releasing pore-water and drill mud or cuttings from directional drilling, discharges of fuels, lubricants, and other substances from construction equipment. Maintenance activities may include the removal and treatment of fouling communities and releases of contaminants similar to those described above. The reader should refer to the Construction Impacts Associated with Sewage Discharge Facilities and the Maintenance Activities Associated with Sewage Discharge Facilities subsections of this chapter for additional information on this topic.

Conservation measures and best management practices for sewage and industrial discharge facilities and CSOs (adapted from Hanson et al. 2003)

1. Locate discharge points in coastal waters well away from shellfish beds, submerged aquatic vegetation, reefs, fish spawning grounds, and similar fragile and productive habitats.
2. Determine benthic productivity by sampling prior to any construction activity related to installation of new or modified facilities. Implement all appropriate best management practices to maintain habitat quality during construction including any seasonal restrictions, use of cofferdams, working in the dry at low tide, etc., as is necessary and practicable.
3. Use seasonal restrictions during construction and maintenance operations to avoid impacts to habitat during species' critical life history stages (e.g., spawning and egg development periods), when appropriate. Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements.
4. Develop appropriate modeling studies for plume effects and other parameters of concern in cooperation with the involved resource agencies before finalizing outfall design. Any appropriate recommendations that involve agencies and developed as a consequence of the study results should be incorporated in the construction plans and operation plan for these facilities as enforceable permit conditions.
5. Institute all appropriate source control measures and/or elevate the treatment level to reduce the polluting substances in all effluents to the extent practicable. Ensure that discharge facilities obtain and adhere to NPDES program permits, as appropriate.
6. Ensure that maximum permissible discharges are appropriate for the given project setting and specify any and all operation procedures, performance standards, or best management practices that must be observed to address all reasonably foreseeable contingencies over the life of the project. Consider implementing an adaptive management plan that includes representatives from appropriate agencies to participate in future consultations for administering the management plan. Management plans should include monitoring protocols designed to measure discharge and potential impacts to sensitive resources and habitats.
7. Use best available technologies to treat discharges to the maximal effective and practicable extent, including measures that reduce discharges of biocides and other toxic substances.
8. Take precautions to mitigate the ecological damage arising from outfall maintenance activities. Facility maintenance plans should include measures such as: (a) ensuring biocides selected for a particular application are specifically designed for their intended use; (b) applying no more than the minimal effective dose, and; (c) closely following instructions for use in aquatic applications and ultimate disposal.
9. Use land treatment and upland disposal or storage for any sludge or other remaining wastes after wastewater processing is concluded. Use of vegetated wetlands as biofilters and pollutant assimilators for large-scale discharges should be limited only to circumstances where other less damaging alternatives are not available and the overall environmental suitability of such an action has been demonstrated.
10. Avoid locating pipelines and treatment facilities in wetlands and streams. Discharges should not be sited near eroding waterfronts or where receiving waters cannot reasonably assimilate the amount of anticipated discharge.
11. Ensure that the design capacity for all facilities will address present and reasonably foreseeable needs and that the best available technologies are implemented.
12. Encourage communities to reduce the volume of pollutants entering CSOs and reduce the number of CSO overflows during storm water runoff producing events. The US EPA provides recommended best management practices for communities (USEPA 1999), including: (a) reduce

and manage solid wastes streams; (b) encourage waste reduction and recycling; (c) reduce commercial and industrial pollution; (d) implement regular program of street cleaning; (e) maintain catch basins; (f) conserve water; (g) reduce unnecessary fertilizer and pesticide applications and; (h) control sediment and erosion.

References for Chemical Effects: Water Discharge Facilities

Allen H, editor. 1996. Metal contaminated aquatic sediments. Boca Raton (FL): CRC Press. 292 p.

Anderson DM, Glibert PM, Burkholder JM. 2002. Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. Estuaries 25(4b):704-26.

Atchison G, Henry M, Sandheinrich M. 1987. Effects of metals on fish behavior: a review. Environmental Biology of Fishes 18:11-25.

Bahgat FJ, King PE, Shackley SE. 1989. Ultrastructural changes in the muscle tissue of *Clupea harengus* L. larvae induced by acid pH. Journal of Fish Biology 34:25-30.

Barker DE, Khan RA, Hooper R. 1994. Bioindicators of stress in winter flounder, *Pleuronectes americanus*, captured adjacent to a pulp and paper mill in St. George's Bay, Newfoundland. Canadian Journal of Fisheries and Aquatic Sciences 51(10):2203-9.

Barr BW. 1993. Environmental impacts of small boat navigation: vessel/sediment interactions and management implications. In: Magoon OT, editor. Coastal Zone '93: proceedings of the eighth Symposium on Coastal and Ocean Management; 1993 Jul 19-23; New Orleans, LA. American Shore and Beach Preservation Association. p 1756-70.

Barr BW, Wilk SJ. 1994. Sewage sludge and industrial waste dumping. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 41-4.

Bejda AJ, Phelan BA, Studholme AL. 1992. The effects of dissolved oxygen on growth of young-of-the-year winter flounder. Environmental Biology of Fishes 34:321-7.

Berry W, Rubenstein N, Melzian B, Hill B. 2003. The biological effects of suspended and bedded sediments (SABS) in aquatic systems: a review. Narragansett (RI): Internal report to US EPA, Office of Research and Development. 58 p.

Bishop JM. 1984. Applied oceanography. McCormick ME, Bhattacharyya R, editors. New York (NY): John Wiley & Sons. 252 p.

Blaber S, Blaber T. 1980. Factors affecting the distribution of juvenile estuarine and inshore fish. Journal of Fish Biology 17:143-62.

Blaxter J, Hallers-Tjabbes C. 1992. The effect of pollutants on sensory systems and behaviour of aquatic animals. Netherlands Journal of Aquatic Ecology 26(1):43-58.

Bodammer JE. 1981. The cytopathological effects of copper on the olfactory organs of larval fish (*Pseudopleuronectes americanus* and *Melanogrammus aeglefinus*). Copenhagen (Denmark): ICES CM-1981/ E: 46.

Bossart G, Brawner T, Cabal C, Kuhns M, Eimstad E, Caron J, Trimm M, Bradley P. 1990. Hepatitis B-like infection in a Pacific whitesided dolphin (*Lagenorhynchus obliquidens*).

American Veterinary Medical Association 196:127-30.

Bricker SB, Clement CG, Pirhalla DE, Orlando SP, Farrow DRG. 1999. National estuarine eutrophication assessment: Effects of nutrient enrichment in the Nation's estuaries. Silver Spring (MD): NOAA, NOS, Special Projects Office and the National Centers for Coastal Ocean Science. 71 p.

Brodeur JC, Sherwood G, Rasmussen JB, Hontela A. 1997. Impaired cortisol secretion in yellow perch (*Perca flavescens*) from lakes contaminated by heavy metals: *in vivo* and *in vitro* assessment. Canadian Journal of Fisheries and Aquatic Sciences 54 (12):2752-8.

Buchholtz ten Brink MR, Manheim FT, Bothner MH. 1996. Contaminants in the Gulf of Maine: what's there and should we worry? In: Dow D, Braasch E, editors. The health of the Gulf of Maine ecosystem: cumulative impacts of multiple stressors. Hanover (NH): Dartmouth College, Regional Association for Research on the Gulf of Maine (RARGOM) Report 96-1. p 91-5.

Buck EH, Copeland C, Zinn JA, Vogt DU. 1997. *Pfiesteria* and related harmful blooms: natural resource and human health concerns. [Internet]. Washington (DC): National Council for Science and the Environment. Congressional Research Service Report for Congress 97-1047 ENR. [cited 2008 Jul 9]. Available from: http://www.cnie.org/NLE/CRSreports/marine/mar-23.cfm.

Capuzzo JM, Sassner JJ Jr. 1977. The effect of chromium on filtration rates and metabolic activity of *Mytilus edulis* L. and *Mya arenaria* L. In: Vernberg FJ, and others, editors. Physiological responses of marine biota to pollutants. San Diego (CA): Academic Press. p 225-37.

Castro MS, Driscoll CT, Jorden TE, Reay WG, Boynton WR. 2003. Sources of nitrogen to estuaries in the United States. Estuaries 26(3):803-14.

Chapman P, Wang F. 2001. Assessing sediment contamination in estuaries. Bulletin of Environmental Contamination and Toxicology 20:3-22.

Chillrud S, Hemming S, Shuster E, Simpson H, Bopp R, Ross J, Pederson D, Chaky D, Tolley L, Estabrooks F. 2003. Stable lead isotopes, contaminant metals and radionuclides in upper Hudson River sediment cores: implications for improved time stratigraphy and transport processes. Chemical Geology 199(1-2):53-70.

Cloern J. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210:223-52.

Costello MJ, Gamble JC. 1992. Effects of sewage sludge on marine fish embryos and larvae. Marine Environmental Research 33(1):49-74.

Daughton CG, Ternes TA. 1999. Pharmaceuticals and personal care products in the environment: agents of subtle change? [Internet]. Environmental Health Perspectives 107(S6): 907-38. [cited 2008 Aug 5]. Available from: http://www.epa.gov/ppcp/pdf/errata.pdf.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Dethlefsen V. 1976. The influence of DDT and DDE on the embryogenesis and the mortality of larvae of cod (*Gadus morhua* L.). Meereforshung 25(3-4):115-48.

Diderich R. 2002. Environmental safety and risk assessment. In: Knight D, Cooke M, editors. The biocides business. Wemheim (Federal Republic of Germany): Wiley-VCH. p 167-95.

Epstein E. 2002. Land applications of sewage sludge and biosolids. Boca Raton (FL): CRC Press. 216 p.

Galtsoff P. 1964. The American oyster *Crassostrea virginica* Gmelin. Fisheries Bulletin 64:1-480.

Goldburg R, Triplet T. 1997. Murky waters: environmental effects of aquaculture in the United States. Washington (DC): Environmental Defense Fund.

Goldsborough WJ. 1997. Human impacts on SAV-a Chesapeake Bay case study. In: Stephan CD, Bigford TE, editors. Atlantic coastal submerged aquatic vegetation: a review of its ecological role, anthropogenic impacts, state regulation, and value to Atlantic coastal fisheries. Washington (DC): ASMFC Habitat Management Series #1. 68 p + appendices.

Gould E, Clark PE, Thurberg FP. 1994. Pollutant effects on dermersal fishes. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 30-41.

Gould E, Thompson RJ, Buckley LJ, Rusanowsky D, Sennefelder GR. 1988. Uptake and effects of copper and cadmium in the gonad of the scallop *Placopecten magellanicus*: concurrent metal exposure. Marine Biology 97(2): 217-223.

Grady CPL, Daigger GT, Lim HC. 1998. Biological wastewater treatment. New York (NY): Marcel Dekker. 1076 p.

Gronlund W, Chan S, McCain B, Clark R, Myers M, Stein J, Brown D, Landahl J, Krahn M, Varanasi U. 1991. Multidisciplinary assessment of pollution at three sites in Long Island Sound. Estuaries 14(3):299-305.

Gustafsson Ö, Gschwend PM. 1997. Aquatic colloids: concepts, definitions, and current challenges. Limnology and Oceanography 42(3):519-28.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Hare L, Tessier A, Warren L. 2001. Cadmium accumulation by invertebrates living at the sediment-water interface. Environmental Toxicology and Chemistry 20(4):880-9.

Huntley S, Wenning R, Bonnevie N, Paustenbach D. 1995. Geochronology and sedimentology of the lower Passaic River, New Jersey. Estuaries 18(2):351-61.

Islam MdS, Tanaka M. 2004. Impacts of pollution on coastal and marine ecosystems including coastal and marine fisheries and approach for management: a review and synthesis. Marine Pollution Bulletin 48(7-8):624-49.

Johnson K, Kahl S. 2005. A systematic survey of water chemistry for Downeast area rivers (Final Report). [Internet]. Orono (ME): Senator George J. Mitchell Center for Environmental and Watershed Research, University of Maine. [cited 2007 Dec 21]. Available from: http://www.umaine.edu/waterresearch/research/nfwf_salmon_water_chemistry.htm.

Kasumyan A. 2001. Effects of chemical pollutants on foraging behavior and sensitivity of fish to food stimuli. Journal of Ichthyology 41(1):76-87.

Kavlock RJ, Ankley GT. 1996. A perspective on the risk assessment process for endocrine-disruptive effects on wildlife and human health. Risk Analysis 16(6):731-9.

Kavlock RJ, Daston GP, DeRosa C, Fenner-Crisp P, Gray LE, Kaattari S, Lucier G, Luster M, Mac MJ, Maczka C, and others. 1996. Research needs for the risk assessment of health and environmental effects of endocrine disruptors: a report of the US EPA-sponsored workshop. Environmental Health Perspectives 104 (Suppl 4):715-40.

Kennish MJ. 1998. Pollution impacts on marine biotic communities. Boca Raton (FL): CRC Press.

Klein-MacPhee G, Cardin JA, Berry WJ. 1984. Effects of silver on eggs and larvae of the winter flounder. Transactions of the American Fisheries Society 113(2):247-51.

Knight D, Cooke M, editors. 2002. The biocides business: regulation, safety and application. New York (NY): Wiley-Interscience. 404 p.

Korom S. 1992. Natural denitrification of the saturated zone: a review. Water Resources Research WRERAQ 28(6):1657-68.

Lang T, Dethlefsen V. 1987. Cadmium in skeletally deformed and normally developed Baltic cod (*Gadus morhua* L.). Copenhagen (Denmark): ICES CM-1987/ E: 30:18 p.

Langmuir D. 1996. Aqueous environmental geochemistry. New York (NY): Prentice-Hall. 600 p.

Larsen PF. 1992. An overview of the environmental quality of the Gulf of Maine. In: The Gulf of Maine. Silver Spring (MD): NOAA Coastal Ocean Program Synthesis Series No. 1. p 71-95.

Leppard GG, Droppo IG. 2003. The need and means to characterize sediment structure and behaviour prior to the selection and implementation of remediation plans. Hydrobiologia 494:313-7.

Levinton J. 1982. Marine ecology. New York (NY): Prentice-Hall. 526 p.

Lin Q, Mendelssohn IA. 1996. A comparative investigation of the effects of south Louisiana crude oil on the vegetation of fresh, brackish, and salt marshes. Marine Pollution Bulletin 32(2):202-9.

Logan DT. 2007. Perspective on ecotoxicology of PAHs to fish. Human and Ecological Risk Assessment 13(2):302-316.

Long E, MacDonald D, Smith S, Calder F. 1995. Incidence of adverse biological effects within ranges of chemical concentrations in marine and estuarine sediments. Environmental Management 19(1):81-97.

Long ER, Wolfe DA, Scott KJ, Thursby GB, Stern EA, Peven C, Schwartz T. 1995. Magnitude and extent of sediment toxicity in the Hudson-Raritan Estuary. Silver Spring (MD): US Department of Commerce. NOAA/National Ocean Service/Office of Ocean Resources Conservation and Assessment. NOAA Technical Memorandum NOS ORCA 88. 134 p + append.

Longwell AC, Chang S, Hebert A, Hughes J, Perry D. 1992. Pollution and development abnormalities of Atlantic fishes. Environmental Biology of Fishes 35:1-21.

Ludwig M, Gould E. 1988. Contaminant input, fate and biological effects. In: Pacheco AL, editor. Woods Hole (MA): US Department of Commerce. NOAA/NMFS/Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-F/NEC 56. p 305-322.

Luoma SN. 1996. The developing framework of marine ecotoxicology: pollutants as a variable in marine ecosystems. Journal of Experimental Marine Biology and Ecology 200:29-55.

McElroy A, Farrington J, Teal J. 1989. Bioavailability of polycyclic aromatic hydrocarbons in the aquatic environment. In: Varanasi U, editor. Metabolism of polycyclic aromatic hydrocarbons in the aquatic environment. Boca Raton (FL): CRC Press. p 2-39.

Mearns A, Matta M, Shigenaka G, MacDonald D, Buchman M, Harris H, Golas H, Laurenstein G. 1991. Contaminant trends in the southern California Bight: inventory and assessment. Silver Spring (MD): US Dept. Commerce, NOAA/NOS. NOAA Technical Memorandum NOS ORCA 62. 413 p.

Meffe G, Carroll C. 1997. Principles of conservation biology, 2nd ed. New York (NY): Sinauer Associates. 729 p.

Mercer IRG, Barker DE, Khan RA. 1997. Stress-related changes in cunner, *Tautogolabrus adspersus*, living near a paper mill. Bulletin of Environmental Contamination and Toxicology 58(3):442-7.

Messieh SN, Rowell TW, Peer DL, Cranford PJ. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. Continental Shelf Research 11(8-10):1237-63.

Moore M, Lefkovitz L, Hall M, Hillman R, Mitchell D, Burnett J. 2005. Reduction in organic contaminant exposure and resultant hepatic hydropic vacuolation in winter flounder (*Pseudopleuronectes americanus*) following improved effluent quality and relocation of the Boston sewage outfall into Massachusetts Bay, USA: 1987-2003. Marine Pollution Bulletin 50:156-66.

Moring J. 2005. Recent trends in anadromous fishes. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 25-42.

[NRC] National Research Council. 1980. The chemistry of disinfectants in water: reactions and products. In: Drinking water and health, Vol. 2. Washington (DC): National Academy Press. p 139-250.

[NRC] National Research Council. 2000. Clean coastal waters: understanding and reducing the effects of nutrient pollution. Washington (DC): National Academy Press. 405 p.

[NRC] National Research Council. 2004. Atlantic salmon in Maine. Washington (DC): National Academy Press. 275 p.

Nelson D, Miller J, Rusanowsky D, Greig R, Sennefelder G, Mercaldo-Allen R, Kuropat C, Gould E, Thurberg F, Calabrese A. 1991. Comparative reproductive success of winter flounder in Long Island Sound: a three-year study (biology, biochemistry, and chemistry). Estuaries 14(3):318-31.

Neumann G, Pierson WJ Jr. 1966. Principles of physical oceanography. Englewood Cliffs (NJ): Prentice-Hall International. 545 p.

Newcombe CP, Jensen JO. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4):693-727.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol. 1.

O'Connor T. 2002. National distribution of chemical concentrations in mussels and oysters in the USA. Marine Environmental Research 53:117-43.

Olivieri V. 1982. Bacterial indicators of pollution. In: Pipes W, editor. Bacterial indicators of pollution. Boca Raton (FL): CRC Press. p 21-41.

O'Reilly JE. 1994. Nutrient loading and eutrophication. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 25-30.

Oulasvirta P. 1990. Effects of acid-iron effluent from a titanium dioxide factory on herring eggs in the Gulf of Bothnia. Finnish Fisheries Research 11:7-15.

Paquin P, Ditoro D, Farley K, Santore R, Wu K-B, editors. 2003. Metals in aquatic systems: a review of exposure, bioaccumulation, and toxicity models. Pensacola (FL): Society of Environmental Toxicology and Chemistry (SETAC). 160 p.

Parnell P. 2003. The effects of sewage discharge on water quality and phytoplankton of Hawaiin coastal waters. Marine Environmental Research 55:293-311.

Pastor D, Boix J, Fernandez V, Albaiges J. 1996. Bioaccumulation of organochlorinated contaminants in three estuarine fish species (*Mullus barbatus, Mugil cephalus* and *Dicentrarcus labrax*). Marine Pollution Bulletin 32(3):257-62.

Perry RDWG. 1997. Perry's chemical engineer's handbook. New York (NY): McGraw-Hill Professional. 2640 p.

Phelps H, Wright D, Mihursky J. 1985. Factors affecting trace metal accumulation by estuarine oysters *Crassostrea virginica*. Marine Ecology Progress Series 22:187-97.

Ragsdale H, Thorhaug A. 1980. Trace metal cycling in the U.S. coastal zone: a synthesis. American Journal of Botany 67(7):1102-12.

Ren J, Packman A. 2002. Effects of background water composition on stream-subsurface exchange of submicron colloids. Journal of Environmental Engineering 128(7):624-34.

Ribaudo M, Horan R, Smith M. 1999. Economics of water quality protection from non-point sources: theory and practice. Washington (DC): US Department of Agriculture, Resource Economics Division, Economic Research Service. Report NR 782. 113 p.

Rice SD, Short JW, Heintz RA, Carls MG, Moles A. 2000. Life-history consequences of oil pollution in fish natal habitat. In: Catania P, editor. Energy 2000: the beginning of a new millennium. Lancaster (PA): Technomic Publishers. p 1210-5.

Richardson H. 1997. Handbook of copper compounds and applications. New York (NY): Marcel Dekker. 432 p.

Robinet T, Feunteun E. 2002. Sublethal effects of exposure to chemical compounds: a cause for decline in Atlantic eels? Ecotoxicology 11:265-77.

Scott GR, Sloman KA. 2004. The effects of environmental pollutants on complex fish behaviour: integrating behavioural and physiological indicators of toxicity. Aquatic Toxicology 68:369-92.

Sherwood M. 1982. Fin erosion, liver condition, and trace contaminant exposure in fishes from three coastal regions. In: Mayer GG, editor. Ecological stress and the New York Bight: science and management. Columbia (SC): Estuarine Research Federation. p 359-77.

Short FT, Burdick DM. 1996. Quantifying, eelgrass habitat loss in relation to housing development and nitrogen loading in Waquoit Bay, Massachusetts. Estuaries 19(3):730-9.

Short FT, Burdick DM, Wolf JS, Jones GE. 1993. Eelgrass in estuarine research reserves along the East Coast, U.S.A. Part I: declines from pollution and disease and Part II: management of eelgrass meadows. Silver Spring (MD): NOAA-Coastal Ocean Program Publication. 107 p.

Shumway SE, Kraeuter JN, editors. 2000. Molluscan shellfish research and management: charting a course for the future. Final Proceedings from the Workshop; 2000 Jan; Charleston, SC. Washington (DC): Department of Commerce. 156 p.

Sigel H, Sigel A, editors. 2001. Metal ions in biological systems: concepts on metal ion toxicity. New York (NY): Marcel Dekker. 416 p.

Smith S, MacDonald D, Keenleyside K, Ingersoll C, Field L. 1996. A preliminary evaluation of sediment quality assessment values for freshwater ecosystems. Journal of Great Lakes Research 22(3):624-38.

Staurnes M, Hansen LP, Fugelli K, Haraldstad O. 1996. Short-term exposure to acid water impairs osmoregulation, seawater tolerance, and subsequent marine survival of smolts of Atlantic salmon (*Salmo salar* L.). Canadian Journal of Fisheries and Aquatic Sciences 53(8):1695-704.

Stein JE, Hom T, Collier T, Brown DR, Varanasi U. 1995. Contaminant exposure and biochemical effects in outmigrant juvenile chinook salmon from urban estuaries of Puget Sound, WA. Environmental Toxicology and Chemistry 14(6):1019-29.

Stiles S, Choromanski J, Nelson D, Miller J, Grieg R., Sennefelder G. 1991. Early reproductive success of the hard clam (*Mercenaria mercenaria*) from five sites in Long Island Sound. Estuaries 14(3):332-42.

Stork T. 1983. PCB levels correlated with the ulcus-syndromes in cod (*Gadus morhua* L.). Copenhagen (Denmark): International Council for the Exploration of the Sea. ICES CM-1983/ E: 50:11 p.

Stumm W, Morgan J. 1996. Aquatic chemistry: chemical equilibria and rates in natural waters. New York (NY): John Wiley & Sons. 1005 p.

Tavolaro J. 1984. A sediment budget study of clamshell dredging and ocean disposal activities in the New York Bight. Environmental Geology and Water Science 6(3):133-40.

Tchobanoglous G, Burton F, Stensel H. 2002. Wastewater engineering: treatment and reuse. New York (NY): McGraw-Hill Professional. 1848 p.

Terlizzi A, Fraschetti S, Gianguzza P, Faimali M, Boero F. 2001. Environmental impact of antifouling technologies: state of the art and perspectives. Aquatic Conservation: Marine and Freshwater Ecosystems 11:311-7.

Tessier A, Campbell P, Auclair J, Bisson M. 1984. Relationship between the partitioning of trace metals in sediments and their accumulation in the tissues of the freshwater mollusc *Elliptio complanata* in a mining area. Canadian Journal of Fisheries and Aquatic Sciences 41:1463-72.

Tessier A, Turner D, editors. 1996. Metal speciation and bioavailability in aquatic systems. New York (NY): John Wiley and Sons. 696 p.

Thurberg FP, Gould E. 2005. Pollutant effects upon cod, haddock, pollock, and flounder of the inshore fisheries of Massachusetts and Cape Cod Bays. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 43-66.

Touchette BW, Burkholder JM. 2000. Review of nitrogen and phosphorus metabolism in seagrasses. Journal of Experimental Marine Biology and Ecology 250:133-67.

Turekian K. 1978. The fate of estuaries. Washington (DC): US Environmental Protection Agency. EPA-600/9-78-038. p 27-38.

Turner A, Millward GE. 2002. Suspended particles: their role in estuarine biogeochemical cycles. Estuarine, Coastal and Shelf Science 55:857-83.

Urho L. 1989. Fin damage in larval and adult fishes in a polluted inlet in the Baltic [abstract]. In: Blaxter JHS, Gamble JC, Westernhagen HV, editors. The early life history of fish. Third ICES Symposium; 1988 Oct 3-5; Bergen, Norway. Copenhagen (Denmark): Rapports et Proces-Verbaux Des Reunions 191. International Council for the Exploration of the Sea. p 493-4.

[USEPA] US Environmental Protection Agency. 1984. Use and disposal of municipal wastewater sludge. Washington (DC): US EPA. 625/10-84-003.

[USEPA] US Environmental Protection Agency. 1994. The Long Island Sound Study: the comprehensive conservation and management plan. Stony Brook (NY): University of New York; Long Island Sound Office of the US EPA. 168 p.

[USEPA] US Environmental Protection Agency. 1996. Overview of the storm water management program. Washington (DC): US EPA Office of Water. EPA 833-R-96-008. 42 p.

[USEPA] US Environmental Protection Agency. 1999. Combined sewer overflow management fact sheet: pollution prevention. Washington (DC): US EPA Office of Water. EPA 832-F-99-038. 9 p.

[USEPA] US Environmental Protection Agency. 2002. Profile of the pulp and paper industry, 2nd ed. [Internet]. Washington (DC): US EPA Office of Compliance Sector Notebook Project. EPA-310-R-02-002. [cited 2008 Jul 24]. 135 p. Available from: http://www.epa.gov/compliance/resources/publications/assistance/sectors/notebooks/pulp.html.

[USEPA] US Environmental Protection Agency. 2003a. Guide for industrial waste management. Washington (DC): US EPA Office of Solid Waste. EPA-530-R-03-001.

[USEPA] US Environmental Protection Agency. 2003b. National management measures for the control of non-point pollution from agriculture. [Internet]. Washington (DC): US EPA Office of Wetlands, Oceans, and Watersheds. EPA-841-B-03-004. [cited 2008 Jul 9]. Available from: http://www.epa.gov/owow/nps/agmm/index.html.

[USEPA] US Environmental Protection Agency. 2004a. National coastal condition report II. [Internet]. Washington (DC): US EPA Office of Research and Development/Office of Water. EPA-620/R-03/002. [cited 2008 Jul 9]. Available from: http://www.epa.gov/owow/oceans/nccr2.

[USEPA] US Environmental Protection Agency. 2004b. Report to Congress on the impacts and control of CSOs and SSOs. Washington (DC): US EPA Office of Water. EPA 833-R-04-001.

[USEPA] US Environmental Protection Agency. 2005. National management measures to control nonpoint source pollution from urban areas. Washington (DC): US EPA Office of Water. EPA-841-B-05-004. 518 p.

[USEPA] US Environmental Protection Agency. 2007. Pharmaceuticals and personal care products (PPCP): frequent questions. [Internet]. Washington (DC): US EPA Office of Research and Development. [updated 2007 Dec 14; cited 2008 Aug 5]. Available from: http://www.epa.gov/ppcp/faq.html.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1999. Status review of anadromous Atlantic salmon in the United States. Hadley (MA): USFWS. 131 p.

Varanasi U, editor. 1989. Metabolism of polycyclic aromatic hydrocarbons in the aquatic environment. Boca Raton (FL): CRC Press. 341 p.

Werther J, Ogada T. 1999. Sewage sludge combustion. Progress in Energy and Combustion Science 25(1):55-116.

Widdows J, Burns KA, Menon NR, Page D, Soria S. 1990. Measurement of physiological energentics (scope for growth) and chemical contaminants in mussel (*Arca zebra*) transplanted along a contamination gradient in Bermuda. Journal of Experimental Marine Biology and Ecology 138:99-117.

Wilber D, Brostoff W, Clarke D, Ray G. 2005. Sedimentation: potential biological effects of dredging operations in estuarine and marine environments. [Internet]. DOER Technical Notes Collection. Vicksburg (MS): US Army Engineer Research and Development Center. ERDC TN-DOER-E20. [cited 2008 Jul 22]. 14 p. Available from: http://el.erdc.usace.army.mil/dots/doer/pdf/doere20.pdf.

Wilber DH, Clarke DG. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21(4):855-75.

Williams C. 1996. Combatting marine pollution from land-based activities: Australian initiatives. Ocean & Coastal Management 33(1-3):87-112.

Wu Y, Falconer R, Lin B. 2005. Modeling trace metal concentration distributions in estuarine waters. Estuarine, Coastal and Shelf Science 64:699-709.

Wurl O, Obbard J. 2004. A review of pollutants in the sea-surface microlayer (SML): a unique habitat for marine organisms. Marine Pollution Bulletin 48(11-12):1016-30.

CHAPTER EIGHT: PHYSICAL EFFECTS—WATER INTAKE AND DISCHARGE FACILITIES

Introduction

Water intake and discharge facilities are typically municipal or industrial operations that use water for some processing purpose and/or release effluent water into the aquatic environment. Increased water diversion is associated with human population growth and development (Gregory and Bisson 1997). Some examples of facilities that use and discharge water include fossil-fuel and nuclear power plants, sewage treatment facilities, industrial manufacturing facilities, and domestic and agricultural water supply facilities. The construction and operation of water intake and discharge facilities can have a wide range of physical effects on the aquatic environment including changes in the substrate and sediments, water quality and quantity, habitat quality, and hydrology. Most facilities that use water depend upon freshwater or water with very low salinity for their needs. Reductions in the quality and quantity of freshwater to bays and estuaries have led to serious damage to estuaries in the northeast US region and worldwide (Deegan and Buchsbaum 2005). This chapter discusses the physical impacts associated with water discharge and intake facilities. Refer to the chapter on Chemical Affects: Water Discharge Facilities for information on chemical impacts.

Intake Facilities

Introduction

Water intake facilities can be located in riverine, estuarine, and marine environments and can include domestic water supply facilities, irrigation systems for agriculture, power plants, and industrial process users. Nearly half of US water withdrawals are attributed to thermoelectric power facilities, and about one-third are used for agriculture irrigation (Markham 2006). In freshwater riverine systems, water withdrawal for commercial and domestic water use supports the needs of homes, farms, and industries that require a constant supply of water. Freshwater is diverted directly from lakes, streams, and rivers by means of pumping facilities or is stored in impoundments or reservoirs. Water withdrawn from estuarine and marine environments may be used to cool coastal power generating stations, as a source of water for agricultural purposes, and more recently, as a source of domestic water through desalinization facilities. In the case of power plants and desalinization plants, the subsequent discharge of water with temperatures higher than ambient levels can also occur.

Water intake structures can interfere or disrupt ecosystem functions in the source waters, as well as downstream water bodies such as estuaries and bays. The volume and the timing of freshwater delivery to estuaries have been substantially altered by the production of hydropower, domestic and industrial use, and agriculture (Deegan and Buchsbaum 2005). Long-term water withdrawal may adversely affect fish and shellfish populations by adding another source of mortality to the early life-stage, which affects recruitment and year-class strength (Travnichek et al. 1993). Water intake structures can result in adverse impacts to aquatic resources in a number of ways, including: (1) entrainment and impingement of fishes and invertebrates; (2) alteration of natural flow rates and hydroperiod; (3) degradation of shoreline and riparian habitats; and (4) alteration of aquatic community structure and diversity.

Entrainment and impingement

Entrainment is the voluntary or involuntary movement of aquatic organisms from the parent water body into a surface diversion or through, under, or around screens and results in the loss of the organisms from the population. Impingement is the involuntary contact and entrapment of aquatic organisms on the surface of intake screens caused when the approach velocity exceeds the swimming capability of the organism (WDFW 1998). Most water-intake facilities have the potential to cause entrainment and impingement of some aquatic species when they are located in areas that support those organisms. Facilities that are known to entrain and impinge marine animals include power plants, domestic and agricultural water supplies, industrial manufacturing facilities, ballast water intakes, and hydraulic dredges. Some of these types of facilities need very large volumes and intake rates of water. For example, conventional 1,000-megawatt fossil fuel and nuclear power plants require cooling water rates of approximately 50 and 75 m^3/s, respectively (Hanson et al. 1977). Water diversion projects have been identified as a source of fish mortality and injury, and egg and larval stages of aquatic organisms tend to be the most susceptible (Moazzam and Rizvi 1980; NOAA 1994; Richkus and McLean 2000). Entrainment can subject these life stages to adverse conditions such as increased heat, antifouling chemicals, physical abrasion, rapid pressure changes, and other detrimental effects. Although some temperate species of fish are able to tolerate exposure to extreme temperatures for short durations (Brawn 1960; Barker et al. 1981), fish and invertebrates entrained into industrial and municipal water intake structures experience nearly 100% mortality from the combined stresses associated with altered temperatures, toxic effects of chemical exposure, and mechanical and pressure-related injuries (Enright 1977; Hanson et al. 1977; Moazzam and Rizvi 1980; Barker et al. 1981; Richkus and McLean 2000).

Both entrainment and impingement of fish and invertebrates in power plant and other water intake structures have immediate as well as future impacts to the riverine, estuarine, and marine ecosystems. Not only is fish and invertebrate biomass removed from the aquatic system, but the biomass that would have been produced in the future would not become available to predators (Rago 1984). Water intake structures, such as power plants and industrial facilities, are a source of mortality for managed-fishery species and play a role as one of the factors driving changes in species abundance over time (Richkus and McLean 2000).

Various physical impacts to fish traversing low-head, tidal turbines in the Bay of Fundy, Canada, were reported by Dadswell and Rulifson (1994) and included mechanical strikes with turbine blades, shear damage, and pressure- and cavitation-related injuries/mortality. They found 21-46% mortality rates for experimentally tagged American shad (*Alosa sapidissima*) passing through the turbine. NOAA (1994) reported fish diverted into power turbines experience up to 40% mortality, as well as injury, disorientation, and delay of migration. An entrainment and impingment study for a once-through cooling system of an 848-megawatt electric generating plant on the East River (NY) concluded the reduction in biomass of spawners from an unfished stock in the Long Island Sound and New York-New Jersey estuary to be extremely small (i.e., 0.01% for Atlantic menhaden [*Brevoortia tyrannus*] and 0.09% for winter flounder [*Pseudopleuronectes americanus*]) compared to fishing mortality (Heimbuch et al. 2007).

Organisms that are too large to pass through in-plant screening devices become stuck or impinged against the screening device or remain in the forebay sections of the system until they are removed by other means (Hanson et al. 1977; Langford et al. 1978; Helvey 1985; Helvey and Dorn 1987; Moazzam and Rizvi 1980). They are unable to escape because the water flow either pushes them against the screen or prevents them from exiting the intake tunnel. This can cause injuries such as bruising or descaling, as well as direct mortality. The extent of physical damage to organisms is directly related to the duration of impingement, techniques for handling impinged fish, and the intake water velocity (Hanson et al. 1977). Similar to entrainment, the withdrawal of water can entrap particular species, especially when visual acuity is reduced (Helvey 1985) or when the ambient water temperature and the metabolism of individuals are low (Grimes 1975). This condition reduces the suitability of the source waters to provide normal habitat functions necessary for subadult and adult life stages of managed living marine resources and their prey. Increased predation can also occur. Intakes can stress or disorient fish through nonlethal impingement or entrainment in the facility and by creating conditions favoring predators such as larger fish and birds (Hanson et al. 1977; NOAA 1994).

Ballast water and vessel operations intake

Vessels take in and release water in order to maintain proper ballast and stability, which is affected by the variable weight of passengers and cargo and sea conditions. In addition, water is used for cooling engines and other systems. While the discharge of ballast water can cause significant impacts on the aquatic environment, particularly through the introduction of invasive species as discussed below, the intake of water for ballast and vessel cooling can also cause entrainment and impingement impacts on aquatic organisms.

Depending upon the size of the vessel, millions of gallons of water and its associated aquatic life, particularly eggs and larvae, can be transferred to the ballast tanks of a ship at a rate of tens of thousands of gallons per minute. For example, large ships, such as those constructed to transport liquefied natural gas (LNG), need to take on ballast water to stabilize the ship during offloading of the LNG. A 200,000-m^3 capacity LNG carrier would withdraw approximately 19.8 million gallons of water over a 10-hour period at an intake rate of 2 million gallons per hour (FERC 2005). The use of water for ballast and vessel cooling at these volumes and rates has the potential to entrain and impinge large numbers of fish eggs and larvae. For example, a proposed offshore LNG degasification facility using a closed-loop system near Gloucester, MA, would have estimated annual mortality of eggs and larvae from vessel ballast and cooling water for Atlantic mackerel (*Scomber scombrus*), pollock (*Pollachius virens*), yellowtail flounder (*Limanda ferruginea*), and Atlantic cod (*Gadus morhua*) of 8.5 million, 7.8 million, 411,000, and 569,000, respectively (USCG 2006). Refer to the chapters on Energy-related Activities and Marine Transportation for additional information on vessel entrainment and impingement impacts.

Alteration of hydrological regimes/flow restrictions

Water withdrawals for industrial or municipal water needs can have a number of physical effects to riverine systems, including altering stream velocity, channel depth and width, turbidity, sediment and nutrient transport characteristics, dissolved oxygen concentrations, and seasonal and diel temperature patterns (Christie et al. 1993; Fajen and Layzer 1993). These physical changes can have ecological impacts, such as a reduction of riparian vegetation that affects the availability of fish habitat and prey (Christie et al. 1993; Fajen and Layzer 1993; Spence et al. 1996). Alteration of freshwater flows is one of the most prevalent problems facing coastal regions and has had profound effects on riverine, estuarine, and marine fisheries (Deegan and Buchsbaum 2005). For example, water in the Ipswich River in Massachusetts has been reduced to 10% of historic natural flows because of increased water withdrawals, such as irrigation water during the growing season, power plant cooling water, and potable water for a growing human population (Bowling and Mackin 2003). Approximately one-half of the 45-mile long Ipswich River was reported to have gone completely dry in 1995, 1997, 1999, and 2002, and nearly one-half of the native fish populations have either been extirpated or severely reduced in size (Bowling and Mackin 2003). Many estuarine and diadromous species, such as American eel (*Anguilla rostrata*), striped bass (*Morone* *saxatilis*), white perch (*Morone americana*), Atlantic herring (*Clupea harengus*), blue crab (*Callinectes sapidus*), American lobster (*Homarus americanus*), Atlantic menhaden (*Brevoortia tyrannus*), cunner (*Tautogolabrus adspersus*), Atlantic tomcod (*Microgadus tomcod*), and rainbow smelt (*Osmerus mordax*), depend upon the development of a counter current flow set up by freshwater discharge to enter estuaries as larvae or early juveniles; reductions in the timing and volume of freshwater entering estuaries can reduce this counter current flow and disrupt larval transport (Deegan and Buchsbaum 2005).

Increased need for dredging

The alteration of the hydrological regimes and reductions in flow in riverine and estuarine systems caused by water intake structures can result in the build-up of sediments and increase the need to dredge around the intake facilities in order to prevent the sediments from negatively affecting the operations of the facility. Dredging can cause direct mortality of the benthic organisms within the area to be dredged, result in turbidity plumes of suspended particulates that can reduce light penetration, interfere with respiration and the ability of site-feeders to capture prey, impede the migration of anadromous fishes, and affect the growth and reproduction of filter feeding organisms. For more detailed discussion on the impacts of dredging, refer to the chapters on Marine Transportation and Offshore Dredging and Disposal Activities.

Habitat impacts

The operation of water intake facilities can have a broad range of adverse effects on fishery habitats, including the conversion and loss of habitat and the alteration of the community structure resulting from changes in the hydrological regimes, salinities, and flow patterns. Large withdrawals of freshwater from riverine systems above the tidal water influence can cause an upstream "relocation" of the salt wedge, altering an area's suitability for some freshwater species and possibly altering benthic community structure. In addition, reductions in the volume of freshwater entering estuaries can alter vertical and longitudinal habitat structure and disrupt larval transport (Deegan and Buchsbaum 2005). Water withdrawals during certain times of the year, such as the use of irrigation water during the growing season of crops, power plant cooling water used during high energy-demand periods, or for domestic water usage during dry, summer months can severely impact the ecological health of riverine systems. For example, the water withdrawal from the Ipswich River in Massachusetts increases by two-fold or more during summer months when natural river flows are lowest (Bowling and Mackin 2003). This has led to one-half of the river going completely dry in some years and has caused fish kills and habitat degradation (Bowling and Mackin 2003).

Construction-related impacts

Impacts to aquatic habitats can result from construction-related activities (e.g., dewatering, dredging) as well as routine operation and maintenance activities for water intake facilities. Generally, these impacts are similar in nature to both water intake and discharge structures and facilities. There is a broad range of impacts associated with these activities depending on the specific design and needs of the system. For example, dredging activities associated with construction of pipelines, bulkheads and seawalls, and buildings for a facility can cause turbidity and sedimentation in nearby waters, degraded water quality, noise, and substrate alterations. Filling of the aquatic habitat may also be needed for the construction of the facilities. Excavation of sediments in subtidal and intertidal habitats during construction may have at least short-term impacts, but the recovery of the aquatic habitat for spawning and egg deposition is uncertain

(Williams and Thom 2001). Many of these impacts can be reduced or eliminated through the use of various techniques, procedures, or technologies such as careful siting of the facility, timing restrictions on in-water work, and the use of directional drilling for the installation of pipelines. Some impacts may not be fully eliminated except by eliminating the activity itself.

Turbidity plume and sedimentation effects incidental to facility construction commonly produce a range of direct and indirect effects to living aquatic resources and their habitats. However, not all of the ecological implications of sediment resuspension and transport result in adverse effects to aquatic organisms (Blaber and Blaber 1980). The life history and ecological strategies characteristic of different species also are important considerations in assessing potential physical impacts from facility installation. For instance, while highly motile adult and juvenile life stages of most fishes could flee when construction is ongoing, egg and larval stages as well as nonmotile benthic organisms will likely not be able to avoid impacts. As a general rule, the severity of adverse effects tends to be greatest for early life stages and for adults of some highly sensitive species (Newcombe and Jensen 1996). The eggs and larvae of nonsalmonid estuarine fishes exhibit some of the most sensitive responses to suspended sediment exposures of all the taxa and life history stages for which data are available (Wilber and Clarke 2001). Reductions in the hatching success of white perch and striped bass eggs were reported at suspended sediment concentrations of 1,000 mg/L, and the survival of striped bass and yellow perch (*Perca flavescens*) larvae were reduced at concentrations greater than 500 mg/L and for American shad larvae at concentrations greater than 100 mg/L (Auld and Schubel 1978). Nelson and Wheeler (1997) found reduced hatching success for winter flounder eggs exposed to suspended sediment concentrations as low as 75 mg/L. While some species like the sessile life stages of eastern oyster (*Crassostrea virginica*) have adapted to withstand some acute habitat disturbances such as sedimentation and turbidity (Galtsoff 1964; Levinton 1982), most benthic and slow-moving species would not be able to escape exposure and instead would exhibit adaptive physiological and biochemical responses to counter adverse effects to water quality.

The area affected by water quality impacts from the construction of a water intake facility is largely dependent on the nature of the resuspended sediments, the duration the sediments are held in the water column, and the factors contributing to the transport of the sediments from the site. As benthic material is disturbed during facility installation and site preparation, resuspended particulate matter settles predominantly in the immediate vicinity of the project. Remaining waterborne fractions subsequently would be transported from the site and dispersed according to the grain size of disturbed sediments, the velocity of local water currents, and local wave action (Neumann and Pierson 1966).

The construction of water intake facilities can create adverse impacts within the immediate vicinity of the construction, including disrupting ambient sediment stratigraphy, cohesiveness, and geochemistry. These effects have geochemical consequences that may be particularly significant when construction activities are located in depositional or nutrient-enriched areas and where local sediments tend to be fine-grained. While important, it is essential to recognize that local sediment composition is not the only factor which affects resuspension during water intake facility installation. The type of construction equipment used to build an intake structure also has an important influence on the dispersion of dredge material. For traditional clamshell dredging, Tavolaro (1984) estimates a 2% loss of material through sediment resuspension at the dredge site. Dredge equipment that fluidizes sediments to facilitate their removal (e.g., hydraulic dredges or water jets) could result in a greater dispersion of resuspended sediment, especially when local waters are not quiescent or in situations where unfiltered return flow to the waterway is permitted. While sediment particles naturally exhibit cycles of exchange between the water column and materials composing the bottom substrate (Turner and Millward 2002), mechanized equipment used to remove sediments can reasonably be expected to disturb much deeper sediment horizons in a short period of time than would be expected from storms or in all but the most highly erosion prone coastal areas.

Additional discussions of the effects of dredging, dredged material disposal, and coastal development can be found in the Marine Transportation, Coastal Development, and Offshore Dredging and Disposal chapters.

Conservation measures and best management practices for water intake facilities (adapted from Hanson et al. 2003)

1. Locate facilities that rely on surface waters for cooling or ballast in areas other than estuaries, inlets, heads of submarine canyons, rock reefs, or small coastal embayments where important fishery species or their prey concentrate for spawning and migration.
2. Design and operate facilities to create flow conditions that provide for passage, water quality, proper timing of life history stages, and properly functioning channel, floodplain, riparian, and estuarine conditions.
3. Establish adequate instream flow conditions for anadromous fish.
4. Design intake structures to minimize entrainment or impingement. Velocity caps that produce horizontal intake/discharge currents should be employed, and intake velocities across the intake screen should generally not exceed 0.5 ft/s.
5. Use closed-loop cooling systems in facilities requiring water whenever practicable, especially in areas that would impinge and entrain large numbers of fish and invertebrates.
6. Screen water diversions on fish-bearing streams, as needed. In general, 2 mm wedge wire screens are recommended on intake facilities in areas that support anadromous fishes.
7. Incorporate juvenile and adult fish passage facilities on all water diversion projects (e.g., fish bypass systems).
8. Assess existing and potential aquatic vegetation, the volume and depth of the water body, the amount and timing of freshwater inflow, the presence of upland rearing and spawning habitat, and the relative salinity of the water body.
9. Assess the hydrology of the regulated land's tolerance for increased water exchange. The assessment should account for active management of the water intake facility to allow increased water exchange during critical periods.
10. Install intake pipes and facilities during low flow periods and tidal stage; incorporate appropriate erosion and sediment control best management practices, and have an equipment spill and containment plan and appropriate materials onsite.
11. Monitor facility operations to assess impacts on water temperatures, dissolved oxygen, and other applicable parameters. Adaptive management should be designed to minimize impacts.

Discharge Facilities

Introduction

Although there are a number of potential impacts to aquatic resources from point-source discharges, it is important to be aware that not all point-source discharge results in adverse impacts to aquatic organisms or their habitats. Most point-source discharges are regulated by the US Environmental Protection Agency (US EPA) under the National Pollutant Discharge Elimination System (NPDES), and the effects on receiving waters are generally considered under this permitting program. As authorized by the Clean Water Act, the NPDES permit program controls water pollution by regulating point sources that discharge pollutants into waters of the United States.

Industrial, municipal, and other facilities must obtain permits if their discharges go directly into surface waters. In most cases, the NPDES permit program is administered by authorized state agencies.

Point source discharges may modify habitat by creating adverse impacts to sensitive areas such as freshwater, estuarine, and marine wetlands; emergent marshes; and submerged aquatic vegetation beds and shellfish beds. Extreme discharge velocities of effluent may also cause scouring at the discharge point as well as entrain particulates and thereby create turbidity plumes.

Habitat conversion and exclusion

The discharge of effluent from point sources can cause numerous habitat impacts resulting from the changes in sediments, salinities, temperatures, and current patterns. These can include the conversion and loss of habitat as the salinities of estuarine areas decrease from the inflow of large quantities of freshwater or as areas become more saline through the discharge of effluent from desalinization plants. Temperature changes, increased turbidity, and the release of contaminants can also result in the reduced use of an area by marine and estuarine species and their prey and impede the migration of some diadromous fishes. Outfall pipes and their discharges may alter the structure of the habitats that serve as juvenile development habitat, such as eelgrass beds (Williams and Thom 2001). Power plants, for example, release large volumes of water at higher than ambient temperatures, and the area surrounding the discharge pipes may not support a healthy, productive community because of physical and chemical alterations of the habitat (Wilbur and Pentony 1999).

The accumulation of sediments at an outfall may alter the composition and abundance of infaunal or epibenthic invertebrate communities (Ferraro et al. 1991). These accumulated sediments can smother sessile organisms or force mobile animals to migrate from the area. If sediment characteristics are changed drastically at the discharge location, the benthic community composition may be altered permanently. This can lead to reductions in the biological productivity of the habitat at the discharge site for some aquatic resources as their prey species and important habitat types, such as aquatic vegetation, are no longer present. Outfall pipes can act as groins and interrupt sand transport, cause scour around the structures, and convert native sand habitat to larger course sediment or bedrock (Williams and Thom 2001). This can affect the spawning success of diadromous and estuarine species, many of which serve as prey species for other commercially or recreationally important species.

Alteration of sediment composition

As discussed above, outfall pipes and their discharges may alter the composition of sediments that serve as juvenile development habitat through scouring or deposition of dissimilar sediments (Williams and Thom 2001). Outfalls that typically release water at high velocities may scour sediments in the vicinity of the outfall and convert the substrate to course sediments or bedrock. Conversely, outfalls that release water at lower velocities that contain fine grained, silt-laden sediments may accumulate sediments near the outfall and increase the need to dredge to remove sediment buildup (Williams and Thom 2001). This can lead to a change in the community composition because many benthic organisms are sensitive to grain size. The chronic accumulation of sediments can also bury benthic organisms that serve as prey and limit an area's suitability as forage habitat.

Substrate and sediment scouring

The discharge of effluent from point sources can result in a variety of benthic habitat and water quality impacts relating to scouring of substrate and sediments at the discharge point.

Changes to the substrate from scouring may impact benthic invertebrate and shellfish community, as well as submerged aquatic vegetation, such as eelgrass (Williams and Thom 2001).

Turbidity and sedimentation effects

Turbidity plumes of suspended particulates caused by the discharge of effluent, the scouring of the substrate at the discharge point, and even the repeated maintenance dredging of the discharge area can reduce light penetration and lower the rate of photosynthesis and the primary productivity of an aquatic area while elevated turbidity persists. Fish and invertebrates in the immediate area may suffer a wide range of adverse effects, including avoidance and abandonment of the area, reduced feeding ability and growth, impaired respiration, a reduction in egg hatching success, and resistance to disease if high levels of suspended particulates persist (Newcombe and MacDonald 1991; Newcombe and Jensen 1996; Wilber and Clarke 2001). Auld and Schubel (1978) reported reduced egg hatching success in white perch and striped bass at suspended sediment concentrations of 1,000 mg/L. They also found reduced survival of striped bass and yellow perch larvae at concentrations greater than 500 mg/L and for American shad at concentrations greater than 100 mg per liter (Auld and Schubel 1978). Short-term effects associated with an increase in suspended particles may include high turbidity, reduced light, and sedimentation, which may lead to the loss of benthic structure and disrupt overall productivity if elevated levels persist (USFWS and NMFS 1999; Newcombe and Jensen 1996). Other problems associated with suspended solids include reduced water transport rates and filtering efficiency of fishes and invertebrates and decreased foraging efficiency of sight feeders (Messieh et al. 1991; Wilber and Clarke 2001). Breitburg (1988) found the predation rates of striped bass larvae on copepods decreased by 40% when exposed to high turbidity conditions in the laboratory. In riverine habitats, Atlantic salmon (*Salmo salar*) fry and parr find refuge within interstitial spaces provided by gravel and cobble that can be potentially clogged by sediments, subsequently decreasing survivorship (USFWS and NMFS 1999).

Increased need for dredging

The release of sediment from water discharge facilities, as well as increased turbidity and sedimentation resulting from high velocity outfall structures, can lead to a build-up of sediments. Over time this may increase the need to dredge around the discharge facility in order to prevent the sediments from negatively affecting the operations of the facility or interfering with vessel navigation. Dredging can cause direct mortality of the benthic organisms within the area to be dredged, as well as create turbidity plumes of suspended particulates that can reduce light penetration, interfere with respiration and the ability of site-feeders to capture prey, impede the migration of anadromous fishes, and affect the growth and reproduction of filter feeding organisms (Wilber and Clarke 2001). For more detailed discussion on the impacts of dredging, refer to the chapters on Marine Transportation and Offshore Dredging and Disposal Activities.

Reduced dissolved oxygen

The contents of the suspended material can react with the dissolved oxygen in the water and result in oxygen depletion, which can impact submerged aquatic vegetation and benthos in the vicinity. Reduced dissolved oxygen (DO) can cause direct mortality of aquatic organisms or result in subacute effects such as reduced growth and reproductive success. Bejda et al. (1992) found that the growth of juvenile winter flounder was significantly reduced when 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.

Alteration of temperature regimes

Sources of thermal pollution from water discharge facilities include industrial and power plants. Temperature changes resulting from the release of cooling water from power plants can cause unfavorable conditions for some species while attracting others. 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 2003). Increased water temperatures in the upper strata of the water column can result in water column stratification, which inhibits the diffusion of oxygen into deeper water leading to reduced (hypoxic) or depleted (anoxic) dissolved oxygen concentrations in estuaries (Kennedy et al. 2002). Because warmer water holds less oxygen than colder water does, increased water temperatures reduce the DO 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, thermal 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.

Elevated water temperature can alter the normal migration patterns of some species or result in thermal stress and mortality in individuals should the discharges cease during colder months of the year. Thermal effluents in inshore habitat can cause severe problems by directly altering the benthic community or killing marine organisms, especially larval fish. Temperature influences biochemical processes of the environment and the behavior (e.g., migration) and physiology (e.g., metabolism) of marine organisms (Blaxter 1969). Investigations to determine the thermal tolerances of larvae of Atlantic herring, smooth flounder (*Pleuronectes putnami*), and rainbow smelt suggests that these species can tolerate elevated temperatures for short durations which are near the upper limits of cooling systems of most normally operating nuclear power plants (Barker et al. 1981). However, a number of factors affected the survival of larvae, including the salinity the individuals were acclimated to and the age of the larvae.

Long-term thermal discharge may change natural community dynamics. For example, elevated water temperature has been identified as a potential factor contributing to harmful algae blooms (ICES 1991), which can lead to rapid growth of phytoplankton populations and subsequent oxygen depletion, sometimes resulting in fish kills. Some evidence indicates that elevated water temperatures in freshwater streams and rivers in the northeastern United States caused by anthropogenic impacts may be responsible for increased algal growth, which has been suggested as a possible factor in the diminished stocks of rainbow smelt (Moring 2005).

Alteration of salinity regimes

The discharge of water with elevated salinity levels from desalination plants may be a potential source of impacts to fishery resources. Waste brine is either discharged directly to the ocean or passed through sewage treatment plants. Although some studies have found desalination plant effluent to not produce toxic effects in marine organisms (Bay and Greenstein 1994), there may be indirect effects of elevated salinity on estuarine and marine communities, such as forcing juvenile fish into areas that could increase their chances of being preyed upon by other species. Conversely, treated freshwater effluent from municipal wastewater plants can produce localized reductions in salinity and could subject juvenile fish to conditions of less than optimal salinity for growth and development (Hanson et al. 2003).

Changes in local current patterns

In addition to changes in temperature and salinity, local current patterns can be altered by outfall discharges or by the structures themselves. These changes can be related to changes in the rate of sedimentation around the outfall, the volume of water discharged, and the size and location of the structures.

Release of radioactive wastes

Both natural and anthropogenic sources of radionuclides exist in the environment (ICES 1991). Potential sources of anthropogenic radioactive wastes include nonpoint sources, such as storm water runoff and atmospheric sources (e.g., coal-burning power plants) and point sources, such as industrial facilities (e.g., uranium mining and milling fuel lubrication) and nuclear power plant discharges (ICES 1991; NEFMC 1998). Fish exposed to radioactive wastes can accumulate radioisotopes in tissues, causing toxicity to other marine organisms and consumers (ICES 1991). The identification of radioactive wastes from industrial and nuclear power plant discharges was a focus of concern during the 1980s (ICES 1991). However, most studies since then have found trends of decreasing releases of artificial radionuclides from industrial and nuclear power plant discharges and reduced tissue-burdens in sampled fish and shellfish to levels similar to naturally occurring radionuclides (ICES 1991).

Ballast water discharges

Commercial cargo-carrying and recreational vessels are the primary type of vector that transports marine life around the world, some of which become exotic, invasive species that can alter the structure and function of aquatic ecosystems (Valiela 1995; Carlton 2001; Niimi 2004). Ballast water discharges, occurring when ships take on additional cargo while at a port, are one of the largest pathways for the introduction and spread of aquatic nuisance species (ANS). The introduction of ANS can have wide reaching impacts to the aquatic ecosystem, the economy, and human health. Many ANS species are transported and released in ballast in their larval stages, become bottom-dwelling as adults, and include sea anemones, marine worms, barnacles, crabs, snails, clams, mussels, bryozoans, sea squirts, and seaweeds (Carlton 2001). In addition, some species are transported and released as adults, including diatoms, dinoflagellates, copepods, and jellyfish (Carlton 2001). Invasive, exotic species can displace native species and increase competition with native species and can potentially alter nutrient cycling and energy flow leading to cascading and unpredictable ecological effects (Carlton 2001). Additional discussion of the effects of introduced species can be found in the chapters on Introduced/Nuisance Species and Aquaculture and Marine Transportation.

Behavioral effects

Discharge facility effluents have the potential to alter the behavior of riverine, estuarine, and marine species by changing the chemical and physical attributes of the habitat and water column in the vicinity of the outfall. These include attractions to the increase in flow velocity and altered temperature regimes at the discharge point and changes in predator/prey interactions. Changes in temperature regimes can artificially attract species and alter their normal seasonal migration behavior, resulting in cold shock and mortality of fishes when ambient temperatures are colder and the flow of heated water is ceased during a facility shutdown (Pilati 1976). Shorelines physically altered with outfall structures may also disrupt the migratory patterns and pathways of fish and invertebrates (Williams and Thom 2001).

Physiological effects

Point-source discharges can cause a wide range of physiological effects on aquatic resources including both lethal and sublethal effects. Alteration of temperature, salinity, and dissolved oxygen concentration regimes have been shown to effect the normal physiology of marine organisms and can retard or accelerate egg and larval development and time of hatching (Blaxter 1969). Fish subjected to abnormally cold or hot temperatures from water discharges will either leave the affected area or acclimate to the change if it is within the species' thermal tolerance zone (Pilati 1976). However, a sudden change in ambient temperature can cause thermal shock and result in death to the fish, or the thermal shock may debilitate a fish and make it susceptible to predation (Pilati 1976). Temperature plays an important role in determining the survival and fitness of coldwater species, such as Atlantic salmon, and can affect the normal growth and development of eggs and fry (Blaxter 1969; Spence et al. 1996).

Water intake and outfall facilities can also have widespread chemical effects on aquatic organisms. These effects are discussed in the Chemical Effects: Water Discharge Facilities chapter.

Construction-related impacts of water discharge facilities

The physical effects of constructing water discharge facilities can result from a number of activities, including releasing suspended sediments and associated pore-water in the construction zone; removal of bottom sediments and subsequent suspended sediments; turbidity and alteration of benthic habitats from dredging; releasing drill mud or cuttings from a directional drilling operation; and the loss or conversion of the existing benthic habitat and water column from placement of fill pipelines, and shoreline stabilization structures (e.g., riprap, headwalls). The impacts associated with constructing water intake and discharge structures and facilities are similar in nature and have been discussed in more detail in the Intake Facilities section of this chapter.

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

1. Conduct a thorough environmental assessment of proposed site locations for water discharge facilities prior to granting any regulatory permits. The assessments should include detailed investigations on the utilization of the aquatic environment by resident and transient species, including the migratory pathways of marine and diadromous fishes. Physical and chemical parameters of the proposed site should be included, such as sediment and substrate characteristics, hydrological dynamics of tides and currents, and temperature and salinity regimes.
2. Develop outfall design (e.g., modeling concentrations within the predicted plume or likely extent of deposition within the zone of influence) by using site specific, hydrological data with input from appropriate resource agencies.
3. Select appropriate point-source discharge locations by using information on the concentrations of living marine resources based upon site-specific, biological assessments. Sensitive and highly productive areas and habitats, such as shellfish beds, sea grass beds, hardbottom reefs should be avoided. Reduce potentially high velocities by diffusing effluent to acceptable velocities.
4. Regulate discharge temperatures (both heated and cooled effluent) such that they do not appreciably alter ambient temperatures and cause a change in species assemblages and ecosystem function in the receiving waters. Strategies should be implemented to diffuse the heated effluent.

5. Use land-treatment and upland disposal/storage techniques where possible. Use of vegetated wetlands as natural filters and pollutant assimilators for large-scale discharges should be limited to those instances where other less damaging alternatives are not available and the overall environmental and ecological suitability of such an action has been demonstrated.
6. Avoid siting pipelines and treatment facilities in wetlands and streams. Since pipeline routes and treatment facilities should not necessarily be water-dependent with regard to positioning, the priority should be to avoid their placement in wetlands or other fragile coastal habitats. Avoiding placement of pipelines within streambeds and wetlands will also reduce inadvertent infiltration into conveyance systems and retain natural hydrology of local streams and wetlands.
7. Ensure that all discharge water from outfall structures meets state and federal water quality standards. Whenever feasible, discharge pipes should extend a substantial distance offshore and be buried deep enough to not affect shoreline processes. Buildings and associated structures should be set well back from the shoreline to preclude the need for bank armoring.

References for Physical Effects: Water Intake and Discharge Facilities

Auld AH, Schubel JR. 1978. Effects of suspended sediment on fish eggs and larvae: a laboratory assessment. Estuarine and Coastal Marine Science 6(2):153-64.

Barker SL, Townsend DW, Hacunda JS. 1981. Mortalities of Atlantic herring, *Clupea h. harengus*, smooth flounder, *Liopsetta putnami*, and rainbow smelt, *Osmerus mordax*, larvae exposed to acute thermal shock. Fisheries Bulletin 79(1):1981.

Bay S, Greenstein D. 1994. Toxic effects of elevated salinity and desalination waste brine. In: Cross J, editor. Southern California coastal water research project. Annual report 1992-1993. Westminister (CA): SCCWRP. p 149-53.

Bejda AJ, Phelan BA, Studholme AL. 1992. The effects of dissolved oxygen on growth of young-of-the-year winter flounder. Environmental Biology of Fishes 34:321-7.

Blaber S, Blaber T. 1980. Factors affecting the distribution of juvenile estuarine and inshore fish. Journal of Fish Biology 17:143-62.

Blaxter JHS. 1969. Development: eggs and larvae. In: Hoar WS, Randall DJ, editors. Fish physiology-Vol. 3, reproduction and growth, bioluminescence, pigments, and poisons. New York (NY): Academic Press. p 177-252.

Bowling A, Mackin K. 2003. The state of the Ipswich River. Ipswich (MA): Ipswich River Watershed Association. 9 p.

Brawn VM. 1960. Temperature tolerance of unacclimated herring (*Clupea harengus* L.). Journal of the Fisheries Research Board of Canada 17(5):721-3.

Breitburg DL. 1988. Effects of turbidity on prey consumption by striped bass larvae. Transactions of the American Fisheries Society 117(1):72-7.

Carlton JT. 2001. Introduced species in U.S. coastal waters: environmental impacts and management priorities. Arlington (VA): Pew Oceans Commission. 28 p.

Christie RW, Allen DE, Jobsis GJ. 1993. Diversions. In: Bryan CF, Rutherford DA, editors. Impacts on warmwater streams: guidelines for evaluation. Little Rock (AR): Southern Division, American Fisheries Society. p 181-6.

Dadswell MJ, Rulifson RA. 1994. Macrotidal estuaries: a region of collision between marine animals and tidal power development. Biological Journal of the Linnean Society 51(1-2):93-113.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Enright JT. 1977. Power plants and plankton. Marine Pollution Bulletin 8(7):158-61.

Fajen OF, Layzer JB. 1993. Agricultural practices. In: Bryan CF, Rutherford DA, editors. Impacts on warmwater streams: guidelines for evaluation. Little Rock (AR): Southern Division, American Fisheries Society. p 257-67.

[FERC] Federal Energy Regulatory Commission. 2005. Draft environmental impact statement, Down Landing LNG and Logan Lateral projects. Washington (DC): FERC Office of Energy Projects.

Ferraro SP, Swartz C, Cole FA, Schults DW. 1991. Temporal changes in the benthos along a pollution gradient: discriminating the effects of natural phenomena from sewage-industrial wastewater effects. Estuarine, Coastal and Shelf Science 33:383-407.

Galtsoff P. 1964. The American oyster *Crassostrea virginica* Gmelin. Fisheries Bulletin 64:1-480.

Gregory SV, Bisson PA. 1997. Degradation and loss of anadromous salmonid habitat in the Pacific Northwest. In: Stouder DJ, Bisson PA, Naiman RJ, editors. Pacific salmon and their ecosystems: status and future options. New York (NY): Chapman and Hall. p 277-314.

Grimes CB. 1975. Entrapment of fishes on intake water screens at a stream electric generating station. Chesapeake Science 16(3):172-7.

Hanson CH, White JR, Li HW. 1977. Entrapment and impingement of fishes by power plant cooling water intakes: an overview. Marine Fisheries Review 39:7-17.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Heimbuch DG, Dunning DJ, Ross QE, Blumberg AF. 2007. Assessing potential effects of entrainment and impingement on fish stocks of the New York-New Jersey Harbor estuary and Long Island Sound. Transactions of the American Fisheries Society 136:492-508.

Helvey M. 1985. Behavioral factors influencing fish entrapment at offshore cooling-water intake structures in southern California. Marine Fisheries Review 47:18-26.

Helvey M, Dorn PB. 1987. Selective removal of reef fish associated with an offshore cooling-water intake structure. Journal of Applied Ecology 24(1):1-12.

[ICES] International Council for the Exploration of the Sea. 1991. Report of the study group on ecosystem effects of fishing activities. 11-15 March 1991; Lowestoft, England. Copenhagen (Denmark): ICES C.M. 1991/G:7 Session Y.

Kennedy VS, Twilley RR, Kleypas JA, Cowan JH, Hare SR. 2002. Coastal and marine ecosystems and global climate change: potential effects on U.S. resources. Arlington (VA): Pew Center on Global Climate Change. 51 p.

Langford TE, Utting NJ, Holmes RHA. 1978. Factors affecting the impingement of fishes on power station cooling-water intake screens. In: McLusky DS, Berry AJ, editors. Physiology and behaviour of marine organisms. Oxford (England) and New York (NY): Pergamon Press. p 281-8.

Levinton J. 1982. Marine ecology. New York (NY): Prentice-Hall. 526 p.

Markham VD. 2006. U.S. national report on population and the environment. New Canaan (CT): Center for Environment and Population. 67 p.

Messieh SN, Rowell TW, Peer DL, Cranford PJ. 1991. The effects of trawling, dredging and ocean dumping on the eastern Canadian continental shelf seabed. Continental Shelf Research 11(8-10):1237-63.

Moazzam M, Rizvi SHN. 1980. Fish entrapment in the seawater intake of a power plant at Karachi coast. Environmental Biology of Fishes 5:49-57.

Moring J. 2005. Recent trends in anadromous fishes. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 25-42.

[NOAA] National Oceanic and Atmospheric Administration. 1994. Experimental fish guidance devices. Position statement. [Internet]. Long Beach (CA): National Marine Fisheries Service, Southwest Region. [cited 2008 Jul 24]. 7 p. Available from: http://swr.nmfs.noaa.gov/hcd/exprscrn.pdf.

Nelson DA, Wheeler JL. 1997. The influence of dredging-induced turbidity and associated contaminants upon hatching success and larval survival of winter flounder, *Pleuronectes americanus*, a laboratory study. Milford (CT): Connecticut Department of Environmental Protection. Final Report on Grant CWF #321-R. 57 p.

Neumann G, Pierson WJ Jr. 1966. Principles of physical oceanography. Englewood Cliffs (NJ): Prentice-Hall International. 545 p.

Newcombe CP, Jensen JO. 1996. Channel suspended sediment and fisheries: a synthesis for quantitative assessment of risk and impact. North American Journal of Fisheries Management 16(4):693-727.

Newcombe CP, MacDonald DD. 1991. Effects of suspended sediments on aquatic ecosystems. North American Journal of Fisheries Management 11(1):72-82.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the

proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol. 1.
Niimi AJ. 2004. Role of container vessels in the introduction of exotic species. Marine Pollution Bulletin 49(9-10):778-82.

Pilati DA. 1976. Cold shock: biological implications and a method for approximating transient environmental temperatures in the near-field region of a thermal discharge. Science of the Total Environment 6(3):227-37.

Rago PJ. 1984. Production forgone: an alternative method for assessing the consequences of fish entrainment and impingement losses at power plants and other water intakes. Ecological Modelling 24(1-2):79-111.

Richkus WA, McLean R. 2000. Historical overview of the efficacy of two decades of power plant fisheries impact assessment activities in Chesapeake Bay. Environmental Science and Policy 3(Supplement 1):283-93.

Spence BC, Lomnicky GA, Hughes RM, Novitzki RP. 1996. An ecosystem approach to salmonid conservation. [Internet]. Corvallis (OR): ManTech Environmental Research Services Corp. TR-4501-96-6057. [cited 2008 Jul 9]. Available from: http://www.nwr.noaa.gov/Publications/Reference-Documents/ManTech-Report.cfm.

Tavolaro J. 1984. A sediment budget study of clamshell dredging and ocean disposal activities in the New York Bight. Environmental Geology and Water Science 6(3):133-40.

Travnichek VH, Zale AV, Fisher WL. 1993. Entrainment of ichthyoplankton by a warmwater hydroelectric facility. Transactions of the American Fisheries Society 122(5):709-16.

Turner A, Millward GE. 2002. Suspended particles: their role in estuarine biogeochemical cycles. Estuarine, Coastal and Shelf Science 55:857-83.

[USCG] US Coast Guard. 2006. Environmental impact statement/environmental impact report for the Neptune LNG deepwater port license application. Washington (DC): USCG-2005-22611.

[USEPA] US Environmental Protection Agency. 2003. National management measures for the control of non-point pollution from agriculture. [Internet]. Washington (DC): US EPA Office of Water. EPA-841-B-03-004. [cited 2008 Jul 9]. Available from: http://www.epa.gov/owow/nps/agmm/index.html.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1999. Status review of anadromous Atlantic salmon in the United States. Hadley (MA): USFWS. 131 p.

Valiela I. 1995. Marine ecological processes, 2nd ed. New York (NY): Springer-Verlag.

[WDFW] Washington Department of Fish and Wildlife. 1998. Fish passage technical assistance: surface diversion and fish screen terminology. [Internet]. Olympia (WA): Washington

Department of Fish and Wildlife; c1998-1999; [cited 2007 Dec 28]. Available from: http://wdfw.wa.gov/hab/engineer/scrnterm.htm.

Wilber DH, Clarke DG. 2001. Biological effects of suspended sediments: a review of suspended sediment impacts on fish and shellfish with relation to dredging activities in estuaries. North American Journal of Fisheries Management 21(4):855-75.

Wilbur AR, Pentony MW. 1999. Human-induced nonfishing threats to essential fish habitat in the New England region. American Fisheries Society Symposium 22:299-321.

Williams GD, Thom RM. 2001. Marine and estuarine shoreline modification issues. [Internet]. White paper submitted to Washington Department of Fish and Wildlife, Washington Dept. of Ecology, Washington Dept. of Transportation. Sequim (WA): Battelle Memorial Institute. [cited 2008 Jul 9]. 99 p. Available from: http://wdfw.wa.gov/hab/ahg/marnrsrc.htm.

CHAPTER NINE: AGRICULTURES AND SILVICULTURE

Croplands, Rangelands, Livestock, and Nursery Operations

Introduction

Substantial portions of croplands, rangelands, and commercial nursery operations are connected, either directly or indirectly, to coastal waters where point and nonpoint pollution can have an adverse effect on aquatic habitats. According to the US Environmental Protection Agency's (US EPA) 2000 National Water Quality Inventory, agriculture was the most widespread source of pollution for assessed rivers and lakes (USEPA 2002a). In that report, agriculture was responsible for 18% of all river-mile impacts and 14% of all lake-acre impacts in the United States. In addition, 48% of all impaired river miles and 41% of all impaired lake acres were attributed to agriculture (USEPA 2002a). Impacts to fishery habitat from agricultural and nursery operations can result from: (1) nutrient loading; (2) introduction of animal wastes; (3) erosion; (4) introduction of salts; (5) pesticides; (6) sedimentation; and (7) suspended silt in water column (USEPA 2002a).

Release of nutrients/eutrophication

Nutrients in agricultural land are found in several different forms and originate from various sources, including: (1) commercial fertilizers containing nitrogen, phosphorus, potassium, secondary nutrients, and micronutrients; (2) manure from animal production facilities; (3) legumes and crop residues; and (4) irrigation water (USEPA 2002a). In addition, agriculturals land are characterized by poorly maintained dirt roads, ditches, and drains that transport sediments and nutrients directly into surface waters. In many instances, headwater streams have been replaced by a constructed system of roads, ditches, and drains that deliver nutrients directly to surface waters (Larimore and Smith 1963). Worldwide, the production of fertilizers is the largest source of anthropogenic nitrogen mobilization, although atmospheric deposition exceeds fertilizer production as the largest nonpoint source of nitrogen to surface waters in the northeastern United States (Howarth et al. 2002). Human activity is estimated to have increased nitrogen input to the coastal water of the northeastern United States, specifically to Chesapeake Bay, MD/VA, by 6- to 8-fold (Howarth et al. 2002). Castro et al. (2003) estimated that the mid-Atlantic and southeast regions contained between 24-37% agricultural lands, with fertilizers and manure applications representing the highest nitrogen sources for those watersheds. The Pamlico Sound-Pungo River, NC, and Chesapeake Bay estuaries contained the highest percent of nitrogen sources coming from agriculture from the mid-Atlantic region (Castro et al. 2003). The second leading cause of pollution in streams and rivers in Pennsylvania has been attributed to agriculture, primarily nutrient loading and siltation (Markham 2006).

Nitrogen and phosphorus are the two major nutrients from agriculture sources which degrade water quality. The main forces controlling nutrient movement from land to water are runoff, soil infiltration, and erosion. Introduction of these nutrients into aquatic systems can promote aquatic plant productivity and decay leading to cultural eutrophication (Waldichuk 1993). Eutrophication can adversely affect the quality and productivity of fishery habitats in rivers, lakes, estuaries, and near-shore, coastal waters. Eutrophication can cause a number of secondary effects, such as increased turbidity and water temperature, accumulation of dead organic material, decreased dissolved oxygen, and the proliferation of aquatic vegetation. Cultural eutrophication has resulted in widespread damage to the ecology of the Chesapeake Bay, causing nuisance algal blooms, loss of productive shellfish and blue crab (*Callinectes sapidus*) habitat, and destruction of submerged aquatic vegetation (SAV) beds (Duda 1985). Nearly 80% of the nutrient loads intothe Chesapeake

Bay can be attributed to nonpoint sources, and agriculture accounted for the majority of those (USEPA 2003b). Agriculture accounts for approximately 40% and 48% of nitrogen and phosphorus loads, respectively, to the Chesapeake Bay (USEPA 2003b). Chronic eutrophication has severely impacted the historically productive recreational and commercial fisheries of the Chesapeake Bay.

While eutrophication generally causes increased growth of aquatic vegetation, it has been shown to be responsible for wide spread losses of SAV in many urbanized estuaries (Deegan and Buchsbaum 2005). By stimulating the growth of macroalgae, such as sea lettuce (*Ulva lactuca*), eutrophication can alter the physical structure of seagrass meadows, such as eelgrass (*Zostera marina*), by decreasing shoot density and reducing the size and depth of beds (Short et al. 1993; MacKenzie 2005). These alterations can result in the destruction of habitat that is critical for developing juvenile fish and can severely impair biological food chains (Hanson et al. 2003).

Groundwater is also susceptible to nutrient contamination in agricultural lands composed of sandy or other coarse-textured soil (USGS 1999). Nitrate, a highly soluble and mobile form of nitrogen, can leach rapidly through the soil profile and accumulate in groundwater, especially in shallow zones (USEPA 2003a). In the eastern United States, nitrogen contamination of groundwater is generally higher in areas that receive excessive applications of agriculture fertilizers and manure, most notably in mid-Atlantic states like Delaware, Maryland, and Virginia (i.e., the Delmarva Peninsula) (USEPA 2003a). When discharged through seeps and drains, or by direct subsurface flow to water bodies, groundwater can be a significant source of nutrients to surface waters (Hanson et al. 2003). Phosphorus from agricultural sources, such as manure and fertilizer applications and tillage, can also be a significant contributor to eutrophication in freshwater and estuarine ecosystems. Cultivation of agricultural land greatly increases erosion and with it the export of particle-bound phosphorus.

Livestock waste (manure), including fecal and urinary wastes of livestock and poultry, processing water and the feed, bedding, litter, and soil with which they become intermixed, is reported to be the single largest source of phosphorus contamination in the United States (Howarth et al. 2002). Because cattle are often allowed to graze in riparian areas, nutrients that are consumed elsewhere are often excreted in riparian zones that can impact adjacent aquatic habitats (Hanson et al. 2003). Because grazing processes remove or disturb riparian vegetation and soils, runoff that carries additional organic wastes and nutrients into aquatic habitats is accelerated (Hanson et al. 2003). Pollutants contained and processed in rangelands, pastures, or confined animal facilities can be transported by storm water runoff into aquatic environments. These pollutants may include oxygen-demanding substances such as nitrogen and phosphorus; organic solids; salts; bacteria, viruses, and other microorganisms; metals; and sediments that increase organic decomposition (USEPA 2003a). Increased nutrient levels resulting from processed water or manure causes excessive aquatic plant growth and algae. The decomposition of aquatic plants depletes dissolved oxygen in the water, creating anoxic or hypoxic conditions that can lead to fish kills. For example, six individual spills from animal waste lagoons in North Carolina during 1995 totaled almost 30 million gallons; including one spill that involved 22 million gallons of swine waste that was responsible for a fish kill along a 19-mile stretch of the New River (USEPA 2003a). Animal wastes from farms in the United States produce nearly 1.5 billion tons of nitrogen and phosphate-laden wastes each year that contribute to nutrient contamination in approximately 27,999 miles of rivers and groundwater (Markham 2006). The release of animal wastes from livestock production facilities have led to reductions in productivity of riverine, estuarine, and marine habitats because of eutrophication.

Introduction of pathogens

Stormwater runoff from agriculture, particularly livestock manure, typically contains elevated levels of pathogens, including bacteria, viruses, and protozoa (USEPA 2003a). Pathogens are generally a concern to human health because of 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 of fecal contamination, 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). See also the chapter on Introduced/Nuisance Species and Aquaculture for more information on pathogens.

Reduced dissolved oxygen

Reduced (hypoxic) or depleted (anoxic) oxygen conditions within estuarine waters as a result of cultural eutrophication may be one of the most severe problems facing coastal waters in the United States (Deegan and Buchsbaum 2005), and agriculture is a major contributing source in some areas. In general, extensive hypoxia has been more chronic in river-estuarine systems in the southern portion of the northeast coast (i.e., Narragansett Bay, RI, to Chesapeake Bay) than in the northern portion (Whitledge 1985; O'Reilly 1994; NOAA 1997). In 2001 approximately 50% of the deeper waters of the Chesapeake Bay had reduced dissolved oxygen concentrations (USEPA 2003b).

Warm temperatures, high metabolic sediment demand, and water column stratification, conditions that can be common at night during summer months, may lead to low dissolved oxygen concentrations in bottom waters (Deegan and Buchsbaum 2005). Hypoxia in estuaries north of Cape Cod, MA, are uncommon because of strong mixing and flushing characteristics of their waters in the northern New England region. However, high nutrient loads into aquatic habitats from livestock and croplands can cause hypoxic or anoxic conditions that can result in fish kills in rivers and estuaries in other areas of the northeast coast (USEPA 2003a; Deegan and Buchsbaum 2005), and they can potentially alter long-term community dynamics (NRC 2000; Castro et al. 2003). Chronic low-dissolved oxygen conditions can lower the growth and survivorship of finfish and shellfish. For example, 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).

Altered temperature regimes

Increased siltation in shallow aquatic habitats caused by erosion from croplands and livestock operations can result in increased water temperature (Duda 1985). In addition to accelerating bank erosion, loss of riparian vegetation resulting from livestock grazing can increase the amount of solar radiation reaching streams and rivers resulting in an increase in water temperatures (Moring 2005). 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 2003a). The temperature regimes of cold-water fish, such as Atlantic salmon (*Salmo salar*) and rainbow smelt (*Osmerus mordax*), may be exceeded in some rivers and streams of the northeastern United States and lead to local extirpation of these species. The removal of riparian vegetation can also lower water temperatures during winter, which can increase the formation of ice and delay the development of incubating fish eggs and alevins (Hanson et al. 2003). Some evidence indicates that elevated water temperatures in freshwater streams and rivers in the northeastern United States may be responsible for increased algal growth, which has been suggested as a possible factor in the diminished stocks of rainbow smelt (Moring 2005). In the watersheds of eastern Maine, blueberry and cranberry processing plants discharge processing water into rivers important to Atlantic salmon spawning and migration. These facilities are permitted to discharge water at temperatures known to be lethal to both juvenile and adult Atlantic salmon (USFWS and NMFS 1999).

Siltation, sedimentation, and turbidity

As discussed above, siltation, sedimentation, and turbidity impacts related to agricultural activities are generally a result of soil erosion. Agricultural lands are also characterized by poorly maintained dirt roads, ditches, and drains that transport sediments directly into surface waters. 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, and when the sediments settle they can cover oysters and shells which prevents oyster larvae from settling on them (USEPA 2003a; MacKenzie 2007). The largest source of sediment into Chesapeake Bay, for example, is from agriculture. Approximately 63% of the over 5 million pounds of sediment delivered each year to tidal waters of the Chesapeake Bay comes from agricultural sources (MacKenzie 1983; USEPA 2003b) and results in devastating impacts to shellfish and SAV. Wide-spread agricultural deforestation during the 18^{th} and 19^{th} centuries contributed to large sediment loads in the James,VA; York, VA; Rappahannock,VA; Potomac, WV/VA/MD/DC; Patuxent, MD; Choptank, DE/MD; and Nanticoke, DE/MD, Rivers and which may have contributed to the decline of Atlantic sturgeon (*Acipenser oxyrinchus*) populations in the Chesapeake Bay watershed (USFWS and NMFS 1998).

In addition to the affects described in greater detail within the Bank and Soil Erosion subsection of this chapter, contaminants such as pesticides, phosphorus, and ammonium are transported with sediment in an adsorbed state, such that they may not be immediately available to aquatic organisms. However, alteration in water quality, such as decreased oxygen concentration or changes in water alkalinity, may cause these chemicals to be released from the sediment (USEPA 2003a). Consequently, the impacts to aquatic organisms associated with siltation and sedimentation may be combined with the affects of pollution originating from the agricultural lands.

Altered hydrological regimes

There are both direct and indirect affects of agriculture activities on the hydrology of coastal watersheds. Direct alterations of hydrology can occur from water diversion projects used for crop irrigation and livestock operations. The volume and timing of freshwater delivery to estuaries can be altered by water diversions, such as for agriculture, which in turn can increase the salinity of coastal ecosystems and diminish the supply of sediments and nutrients to estuaries (Deegan and Buchsbaum 2005). Agriculture activities use large volumes of water for irrigation, accounting for one-third of all US water withdrawals in 2000 and the second largest source of total water use after thermoelectric energy (Markham 2006).

Water withdrawal for agriculture can have adverse affects on anadromous fish, particularly Atlantic salmon, which use rivers in the Gulf of Maine for spawning and migration. Water withdrawals pose a threat to life stages of Atlantic salmon and their habitat in the Machias, Pleasant, and Narraguagus Rivers in Maine (USFWS and NMFS 1999). Freshwater was diverted from eastern Maine watersheds in the late 1990s to irrigate approximately 6,000 acres of blueberry agricultural activities, and that acreage was expected to double by the year 2005 (USFWS and NMFS 1999). The withdrawal of water may also affect the productivity of oyster beds in the eastern United States, because the distribution of oysters is largely governed by water salinity. When water is withdrawn, oyster beds are forced to move upstream and into smaller areas and often closer to cities where pollution may affect commercial marketing of the oysters (MacKenzie 2007).

Altered hydrology and flood plain storage patterns around estuaries can effect water residence time, temperature, and salinity and can 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 hydrodynamics can affect estuarine circulation, including short-term (diel) and longer term (seasonal or annual) changes (Deegan and Buchsbaum 2005). In addition, counter current flows set up by freshwater discharges into estuaries are important for larvae and juvenile fish entering those estuaries. The diurnal behavioral adaptations of marine and estuarine species allow larvae and early juveniles to concentrate in estuaries. Reductions in freshwater flows caused by increased freshwater withdrawals can disrupt counter current flows and larval transport into estuaries (Deegan and Buchsbaum 2005). The quality and quantity of freshwater flows into estuaries are important in maintaining suitable conditions for spawning, egg, larval, and juvenile development for many estuarine-dependent species.

Indirect affects occur when sediments are transported from agricultural lands via soil erosion and are deposited in roadside ditches, streams, rivers, and navigation channels, which decrease the capacity of watersheds to attenuate the affects of flooding. The morphology of streams and rivers can be altered by eroded soil from improper livestock grazing and croplands, changing the stream width and depth and the timing and magnitude of stream flow (USEPA 2003a). In addition, sediment deposited in lakes and navigation channels reduces the storage capacity of those systems and necessitates more frequent dredging (USEPA 2003a).

Impaired fish passage

Sediments transported from agricultural lands via soil erosion can change the morphology of streams and rivers. As a result, alteration of stream width and depth and the timing and magnitude of stream flow can impair the ability of anadromous fish to reach upstream spawning habitats. Roads that are constructed to access agriculture lands and for livestock may impede or prohibit migrating fish. For example, culverts constructed under roads to allow for water flow can alter the velocity and volume of water in streams and inhibit the ability of fish to migrate through the structure (Furniss et al. 1991). Additional information on fish passage impairments can be reviewed in the Alteration of Freshwater Systems chapter of this report.

Change in community structure and species composition

Cropland and livestock operations can result in community-level impacts to riverine and estuarine ecosystems. As mentioned above, fertilizers applied to agricultural lands enter streams, rivers, and estuaries through stormwater runoff and groundwater sources (e.g., seeps and subsurface flows) and may result in eutrophication. Eutrophication can cause a number of secondary effects, such as increased turbidity and water temperature, accumulation of dead organic material, decreased dissolved oxygen, and the proliferation of macroalgae, such as sea lettuce (MacKenzie 2005). These alterations can then result in the destruction of habitat for small or juvenile fish and severely impair biological food chains (Hanson et al. 2003). For example, eelgrass beds growing in deeper areas of estuaries tend to be impacted more than shallower areas because those beds are very sensitive to light attenuation as a result of eutrophication (Deegan and Buchsbaum 2005). Species that depend upon eelgrass beds may be forced into shallower, potentially less desirable habitats. Declines in commercially and recreationally important finfish in Waquoit Bay, MA, have followed a concomitant decline in eelgrass beds for that area (Deegan and Buchsbaum 2005). Similarly, eelgrass wasting disease was documented to be responsible for severe declines in bay scallop (*Argopectin irradians*) landings along the east coast in the 1930s (Buchsbaum 2005).

Other impacts from agricultural activities such as soil erosion and release of fine sediments can alter aquatic communities through siltation and alteration of benthic substrates. Waldichuk (1993) identified a number of impacts to Pacific salmon (*Oncorhynchus* spp.) caused by activities related to agriculture, such as siltation in spawning, egg incubation and feeding habitats, impaired respiration and abrasion of gills from suspended particles, and failure of egg hatching resulting from low dissolved oxygen. The cumulative effect from the degradation of riverine habitats can inhibit or preclude restoration efforts of salmon populations to historic ranges by altering the community. Release of nutrients from fertilizers applied to croplands, livestock manure, and erosion of soils can reduce the dissolved oxygen levels in aquatic habitats through storm water runoff. Reduced dissolved oxygen in the water or sediments can change community composition to coastal habitats, particularly in areas with restricted water circulation such as coastal ponds, subtidal basins, and salt marsh creeks (Deegan and Buchsbaum 2005). Chronic hypoxia caused by cultural eutrophication can permanently alter the species composition and productivity of these areas.

Entrainment and impingement

Water diverted and extracted for agriculture use can entrain (i.e., draw into flow system) and impinge (i.e., capture onto filter screens) aquatic organisms. Entrainment and impingement generally affects eggs, larvae, and early juvenile fish and invertebrates that cannot actively avoid the currents created at the water intake opening (ASMFC 1992). Long-term water withdrawal may adversely affect fish and invertebrate populations as well as their prey by adding another source of mortality to the early life stage which often determines recruitment and year-class strength (Hanson et al. 2003). Refer to the Physical Affects: Water Intake and Discharge Facilities chapter in this report for additional information on entrainment and impingement.

Bank and soil erosion

Soil erosion in US farmland is estimated to occur seven times as fast as soil formation (Markham 2006). Soil erosion can lead to the transport of fine sediment that may be associated with a wide variety of pollutants from agricultural land into the aquatic environment. The presence of livestock in the riparian zone accelerates sediment transport rates by increasing surface soil erosion (Hanson et al. 2003), loss of vegetation caused by trampling, and streambank erosion resulting from shearing or sloughing (Platts 1991). Increased sedimentation in aquatic systems can increase turbidity and the temperature of the water, reduce light penetration and dissolved oxygen, smother fish spawning areas and food supplies, decrease the growth of SAV, clog the filtering capacity of filter feeders, clog and harm the gills of fish, interfere with feeding behaviors of certain species, cover shells on oyster beds, and significantly lower overall biological productivity (MacKenzie 1983; Duda 1985; USEPA 2003a). Soil eroded and transported from cropland usually contains a higher percentage of finer and less dense particles, which tend to have a higher affinity for adsorbing pollutants such as insecticides, herbicides, trace metals, and nutrients (Duda 1985; USEPA 2003a). One of the consequences of erosional runoff from agricultural land is that it necessitates more frequent dredging of navigational channels (USEPA 2003a), which may result in transportation to and disposal of contaminated sediments in areas important to fisheries production and other marine biota (Witman 1996). Deposition of sediments from erosional runoff can also decrease the storage capacity of roadside ditches, streams, rivers, and navigation channels, resulting in more frequent flooding (USEPA 2003a).

Loss and alteration of riparian-wetland areas

Functioning riparian-wetland areas require stable interactions between geology, soil, water, and vegetation in order to maintain productive riverine ecosystems. When functioning properly, riparian-wetland areas can: (1) reduce erosion and improve water quality by dissipating stream energy; (2) filter sediment and runoff from floodplain development; (3) support denitrification of nitrate-contaminated groundwater; (4) improve floodwater retention and groundwater discharge; (5) develop root masses that stabilize banks from scouring and slumping; (6) develop ponding and channel characteristics necessary to provide habitat for fish, waterfowl, and invertebrates; and (7) support biodiversity (USEPA 2003a). Agriculture activities have the potential to degrade riparian habitats. In particular, improper livestock grazing along riparian corridors can eliminate or reduce vegetation by trampling and increase streambank erosion by shearing or sloughing (Platts 1991). These effects tend to increase the streambank angle, which increases stream width, decreases stream depth, and alters or eliminates fish habitat (USEPA 2003a). As discussed above, the transport of eroded soil from the streambank to streams and rivers impacts water quality and aquatic habitats. Removing riparian vegetation also increases the amount of solar radiation reaching the stream and can result in higher water temperatures.

Reduced soil infiltration and soil compaction

Tillage of croplands aerates the upper soil but tends to compact fine textured soils just below the depth of tillage, thus altering infiltration. Use of farm machinery on cropland and adjacent roads causes further compaction, reducing infiltration and increasing surface runoff (Hanson et al. 2003).

Johnson (1992) and Platts (1991) reviewed studies related to livestock grazing and concluded that heavy grazing nearly always decreases infiltration, reduces vegetative biomass, and increases bare soil. Compaction of rangelands generally increases with grazing intensity, although site-specific soil and vegetative conditions are also important factors in determining the effects of soil compaction (Kauffman and Krueger 1984). Reduced soil infiltration and compaction caused by agriculture are two of the factors that accelerate erosion and release of sediments and contaminants in aquatic habitats.

Salts are present in varying amounts in all soils because of the natural weathering process, but agricultural lands that have poor subsurface drainage can lead to high salt concentrations. Likewise, irrigation water, whether from ground or surface water sources has a natural base load of dissolved mineral salts. Irrigation return flows convey the salt to the receiving streams or groundwater reservoirs. If the amount of salt in the return flow is low in comparison to the total stream flow, water quality may not be degraded to the extent that aquatic functions are impaired. However, if the process of water diversion and the return flow of saline drainage water is repeated many times along a stream or river, downstream habitat quality can become progressively degraded (USEPA 2003a). The accumulation of salts, particularly on irrigated croplands, tends to cause soil dispersion, structure breakdown, and decreased infiltration (USEPA 2003a). While salts are generally a greater pollutant for freshwater ecosystems than for estuarine systems, they may adversely affect anadromous fish that depend upon freshwater systems for crucial portions of their life cycles (USEPA 2003a).

Land-use change (post-agriculture)

When demands for developable land are sufficiently high, the value of land in developed use will exceed its value in agricultural use. In general, conversion of land from agricultural to urban uses is largely irreversible according to the US Department of Agriculture. In the continental United States, census data from urban areas have shown more than a doubling of agricultural land conversion from 25.5 million acres to 55.9 million acres between 1960 and 1990 (USDA 2005). While impacts on aquatic ecosystems from agriculture may be problematic in some areas, conversion of croplands and rangelands to urban and industrial uses may be more harmful in the long-term. Between 1992 and 1997 the state of New York lost approximately 90,000 acres of prime farmland to residential and commercial development, which was 140% faster than in the previous five years (Markham 2006). Refer to the Coastal Development chapter in this report for more information on the impacts of land-use change.

Release of pesticides, herbicides, and fungicides

The term "pesticide" is a collective description of hundreds of chemicals used to protect crops from damaging organisms with different sources and fates in the aquatic environment and that have varying toxic effects on fish and other aquatic organisms (USEPA 2003a). Pesticides can be divided into four categories according to the target pest: insecticides, herbicides, fungicides, and nematicides (USEPA 2003a). Agricultural activities are a major nonpoint source of pesticide pollution in coastal ecosystems (Hanson et al. 2003). Large quantities of pesticides, perhaps 18-20 pounds of pesticide active ingredient per acre, are applied to vegetable crops in coastal areas to control insect and plant pests (Scott et al. 1999). Soil eroded and transported from croplands and rangelands usually contains a higher percentage of finer and less dense particles, which tend to have a higher affinity for adsorbing pollutants such as insecticides and herbicides (Duda 1985; USEPA 2003a). In addition, agricultural lands are typically characterized by poorly maintained dirt roads, ditches and drains that transport sediments, nutrients, and pesticides directly into surface waters. In many instances, roads, ditches, and drains have replaced headwater streams, and these constructed systems deliver pollutants directly to surface waters (Larimore and Smith 1963). Pesticides are frequently detected in freshwater and estuarine systems that provide fishery habitat.

The most common pesticides include insecticides, herbicides, and fungicides. These are used for pest control on forested lands, agricultural crops, tree farms, and nurseries. Pesticides can enter the aquatic environment as single chemicals or complex mixtures. Direct applications, surface runoff, aerial drift, leaching, agricultural return flows, and groundwater intrusions are all examples of transport processes that deliver pesticides to aquatic ecosystems (Hanson et al. 2003).

Most studies evaluating pesticides in runoff and streams generally find that concentrations can be relatively high near the application site and soon after application but are significantly reduced further downstream and with time (USEPA 2003a). However, some pesticides used in the past, such as dichlorodiphenyl trichloroethane (DDT), are known to persist in the environment for years after application. Chlorinated pesticides, such as DDT, and some of the breakdown products are known to cause malformation and fatality in eggs and larvae, alter respiration, and disrupt central nervous system functions in fish (Gould et al. 1994). In addition, pesticides containing organochlorine compounds accumulate and persist in the fatty tissue and livers of fish and could be a threat to human health for those who consume contaminated fish (Gould et al. 1994).

Pesticides may bioaccumulate in organisms by first being adsorbed by sediments and detritus which are ingested by zooplankton and then eaten by planktivores, which in turn are eaten by fish (ASMFC 1992). For example, the livers of winter flounder from Boston and Salem Harbors, MA, contained the highest concentrations of DDT found on the east coast of the United

States and were ranked first and third, respectively, in the country in terms of total pesticides (Larsen 1992). In the Pocomoke River, MD/DE, a tributary of the Chesapeake Bay, agricultural runoff (primarily from poultry farms) was identified as one of the major sources of contaminants (Karuppiah and Gupta 1996). Blueberry and cranberry agriculture is an important land use in eastern Maine watersheds and involves the use of a number of pesticides, herbicides, and fungicides that may cause immediate mortalities to juvenile Atlantic salmon or can have indirect effects when chemicals enter rivers (USFWS and NMFS 1999). One study investigating the effects of two different classes of pesticides (organochlorines and organophosphates) in South Carolina estuaries found significant affects on populations of the dominant macrofauna species, daggerblade grass shrimp (*Palaemonetes pugio*), and mummichogs (*Fundulus heteroclitus*) (Scott et al. 1999). The study found impacts from pesticide runoff on daggerblade grass shrimp populations may cause community-level disruptions in estuaries; however, the authors concluded that implementation of integrated pest management, best management practices, and retention ponds could significantly reduce the levels of nonpoint source runoff from agriculture (Scott et al. 1999).

Endocrine disruptors

Studies have recently focused on a group of chemicals, called "endocrine disruptors," that when present at extremely low concentrates can interfere with fish endocrine systems. Some of these chemicals act as "environmental hormones" that may mimic the function of the sex hormones androgen and estrogen (Thurberg and Gould 2005). Some of the chemicals shown to be estrogenic include some polychlorinated biphenyl (PCB) congeners, dieldrin, DDT, phthalates and alkylphenols (Thurberg and Gould 2005), which have had or still have applications in agriculture. Several studies have found vitellogenin, a yolk precursor protein, in male fish in the North Sea estuaries (Thurberg and Gould 2005). Metals have also been implicated in disrupting endocrine secretions of marine organisms, potentially disrupting natural biotic processes (Brodeur et al. 1997). However, the long-term effect of endocrine-disrupting substances on aquatic life is not well understood and demands serious attention by the scientific and resource policy communities.

Conservation measures and best management practices for croplands, rangelands, livestock, and nursery operations (adapted from Hanson et al. 2003)

1. Recommend field and landscape buffers to provide cost-effective protection against the cumulative effects of multiple pollutant discharges associated with agricultural activities, including riparian forests, alley cropping, contour buffer strips, crosswind trap strips, field borders, filter strips, grassed waterways with vegetative filters, herbaceous wind barriers, vegetative barriers, and windbreak/shelterbelts.
2. Protect and restore soil quality with natural controls that affect permeability and water holding capacity, nutrient availability, organic matter content, and biological activity of the soil. Some examples of best management practices include cover cropping, crop sequence, sediment basins, contour farming, conservation tillage, crop residue management, grazing management, and the use of low-impact farming equipment.
3. Promote efficient use and appropriate applications of pesticides and irrigated water. Sound agricultural practices include use of integrated pest management, irrigation management, soil testing, and appropriate timing of nutrient applications.
4. Encourage protection and restoration of rangelands with practices such as rotational grazing systems or livestock distribution controls, exclusion of livestock from riparian and aquatic areas,

livestock-specific erosion controls, reestablishment of vegetation, or extensive brush management correction.

5. Avoid locating new confined animal facilities or expansion of existing facilities near riparian habitat, surface waters, and areas with high leaching potential to surface or groundwater. Ensure that adequate nutrient and wastewater collection facilities are in place.
6. Minimize water withdrawals for irrigation and promote water conservation measures, such as water reuse.
7. Site roads for agricultural lands to avoid sensitive areas such as streams, wetlands, and steep slopes.
8. Include best management practices (BMPs) for agricultural road construction plans, including erosion control, avoidance of side casting of road materials into streams, and using only native vegetation in stabilization plantings.
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.

Silviculture and Timber Harvest Activities

Introduction

The growth and harvest of forestry products are major land-use types for watersheds along the east coast, particularly in New England, and can have short-term and long-term impacts to riverine habitat (USFWS and NMFS 1999). Forestry is the dominant land-use type in the watersheds of the Dennys, East Machias, Machias, Pleasant, and Narraguagus Rivers in Maine (USFWS and NMFS 1999). Forests that once covered up to 95% of the Chesapeake Bay watershed now cover only 58%, primarily because of land clearing for agriculture and timber (USEPA 2003b). Timber harvest generally removes the dominant vegetation; converts mature and old-growth upland and riparian forests to tree stands or forests of early seral stage; reduces the permeability of soils; increases sedimentation from surface runoff and mass wasting processes; alters hydrologic regimes; and impairs fish passage through inadequate design, construction, and maintenance of stream crossings (Hanson et al. 2003). Silviculture practices can also increase water temperatures in streams and rivers, increase impervious surfaces, and decrease water retention capacity in watersheds (USFWS and NMFS 1999). These watershed changes may result in inadequate river flows; increase stream bank and streambed erosion; sedimentation and siltation of riparian and stream habitat; increase the amount of woody debris; and increase of run-off and associated contaminants (e.g., from herbicides) (Sigman 1985; Hicks et al. 1991; Hanson et al. 2003). Debris (i.e., wood and silt) is released into the water as a result of timber harvest activities and can smother benthic habitat. Poorly placed or designed road construction can cause erosion, producing additional silt and sediment that can impact stream and riparian habitat. Deforestation can alter or impair natural habitat structures and dynamics of the ecosystem.

Four major categories of silviculture activities that can impact fishery habitat are: (1) construction of logging roads; (2) creation of barriers; (3) removal of streamside vegetation; and (4) input of pesticide and herbicide treatments to aquatic habitats.

Release of nutrients/eutrophication

After logging activities, concentrations of plant nutrients in streams and rivers may increase for several years and up to a decade (Hicks et al. 1991). Excess nutrients, combined with increased light regimes caused by the removal of riparian vegetation, can stimulate algal growth; however, the effects of nutrient increases on salmonid populations are not well understood (Hicks et al. 1991). An estimated 41.5 million pounds of nitrogen per year from silviculture activities alone are released into the Chesapeake Bay watershed, contributing to phytoplankton blooms, chronic hypoxia (low dissolved oxygen concentrations), and die-off of SAV (USEPA 2003b).

Reduced dissolved oxygen

Small wood debris and silt resulting from timber harvesting can smother benthic habitat and reduce dissolved oxygen levels in streams (Hicks et al. 1991; Hanson et al. 2003). Fine organic material introduced into streams following logging can result in increased oxygen demand and reduced exchange of surface and intergravel water (Hicks et al. 1991). While low oxygen conditions may not directly kill salmon embryos and alevins in streams after logging, emergent juveniles may have reduced viability (Hicks et al. 1991). Introduction of nutrients into aquatic systems can promote aquatic plant productivity and decay leading to cultural eutrophication (Waldichuk 1993). Anoxic (without oxygen) or hypoxic (low oxygen) conditions have caused widespread ecological problems for the Chesapeake Bay, resulting in a variety ecosystem impacts including the loss of shellfish beds and reductions of fish stocks in the Bay (USEPA 2003b). According to Chesapeake Bay Program modeling, approximately 15% of the nitrogen loads entering the Chesapeake Bay watershed each year are from forestry activities (USEPA 2003b).

Altered temperature regimes

Removing streamside vegetation to construct logging access roads and logging adjacent to streams or rivers increase the amount of solar radiation reaching the water body and can increase water temperatures (Beschta et al. 1987; Hicks et al. 1991). In studies conducted in Alaska, researchers found that maximum temperatures in logged streams without riparian buffers exceeded that of unlogged streams by up to 5ºC, but did not reach lethal temperatures (Hanson et al. 2003). In cold climates, the removal of riparian vegetation can result in lower water temperatures during winter, increasing the formation of ice and damaging and delaying the development of incubating fish eggs and alevins (Hanson et al. 2003). In freshwater habitats of the northeastern United States, the temperature tolerances of cold-water fish such as Atlantic salmon and rainbow smelt may be exceeded leading to local extirpation of the species (USFWS and NMFS 1999). However, increased water temperatures can also increase primary and secondary production, which may lead to greater availability of food for fish (Hicks et al. 1991).

Siltation, sedimentation, and turbidity

Sedimentation in streams resulting from timber harvesting activities can reduce benthic community production, cause mortality of incubating salmon eggs and alevins, reduce the amount of habitat available for juvenile salmon, and lower the productivity of oyster beds (MacKenzie 1983; Hicks et al. 1991; Hanson et al. 2003). Fine sediments deposited in salmon spawning gravel can reduce interstitial water flow, causing reduced dissolved oxygen concentrations, and they can physically trap emerging fry in the gravel (Hicks et al. 1991). Fine sediments on stream bottoms and in suspension can also reduce primary production and invertebrate abundance, reducing the availability of prey for fish (Hicks et al. 1991). Sedimentation in riparian habitat resulting from logging activities can reduce streamside vegetation that impacts bank stabilization, increasing solar radiation reaching the stream. In addition, suspended sediments can alter the behavior and feeding efficiencies of salmonids following timber harvesting (Hicks et al. 1991). Sawdust and pulp from sawmills and lumber companies can also enter streams and rivers and adversely affect benthic habitats of anadromous fish (Moring 2005).

Deforestation and silviculture activities have contributed to excessive amounts of sediments in Chesapeake Bay, which have led to adverse affects on benthic communities like SAV, oysters, and clams (USEPA 2003b). Nearly 1 million tons of sediments are estimated to enter the Chesapeake Bay each year from forestry activities alone, which accounts for approximately 20% of the total sediment loads into the Bay (USEPA 2003b).

Bank and soil erosion and altered hydrological regimes

Timber harvesting may result in inadequate or excessive surface and stream flows, increased stream bank and streambed erosion, and the loss of complex instream habitats. Clear cutting large areas of forests can alter the hydrologic characteristics of watersheds, such as water temperature, and result in greater seasonal and daily variation in stream discharge and flows (Hicks et al. 1991; Hanson et al. 2003).

In addition, logging road construction can destabilize slopes and increase erosion and sedimentation. Mass wasting and surface erosion are the two major types of erosion that can occur from logging road construction. Mass movement of soils, commonly referred to as landslides or debris slides, is associated with timber harvesting and road building on high hazard soils and unstable slopes. The result is increased erosion and sediment deposition in down-slope waterways. Erosion from roadways is most severe when poor construction practices are employed that do not include properly located, designed, and installed culverts or when proper ditching is not utilized (Furniss et al. 1991).

Altered hydrology and flood plain storage patterns around estuaries can effect water residence time, temperature, and salinity and can 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).

Alteration and loss of vegetation

By removing vegetation, timber harvesting tends to decrease the absorptive capability of the groundcover vegetation. This, in turn, increases surface runoff during periods of high precipitation. These effects can destabilize slopes, increase erosion, and cause sedimentation and debris input to streams (Hanson et al. 2003). Reductions in the supply of large woody debris to streams can result when old-growth forests are removed, with resulting loss of habitat complexity that is important for successful salmonid spawning and rearing (Hicks et al. 1991; Hanson et al. 2003). Removing riparian vegetation increases the amount of solar radiation reaching the stream and can result in higher water temperatures during summer months. A loss of riparian vegetation can also reduce stream water temperatures during the winter months (Beschta et al. 1987; Hicks et al. 1991).

Impaired fish passage

Poorly placed or ill-designed culverts placed as part of road construction can negatively affect access to riverine habitat by fish. Stream crossings (e.g., bridges and culverts) on forest roads are often inadequately designed, installed, and maintained, and they frequently result in full or partial barriers to both the upstream and downsteam migration of adult and juvenile fish (Hanson et al. 2003). Perched culverts, in which the culvert invert at the downstream end is above the water level of the downstream pool, create waterfalls that can be physical barriers to migrating fish. Undersized culverts can accelerate stream flows to the point that these structures become velocity barriers for migrating fish. Blocked culverts can result in displacement of the stream from the downstream channel to the roadway or roadside ditch (Hanson et al. 2003). Blocked culverts often result from installation of undersized culverts or inadequate maintenance to remove debris. In addition, culverts and bridges deteriorate structurally over time, and failure to replace or remove them at the end of their useful life may cause partial or total blockage of fish passage.

Release of pesticides, herbicides, and fungicides

Riparian vegetation is an important component of rearing habitat for fish, providing shade for maintaining cool water temperatures, food supply, channel stability, and structure (Furniss et al. 1991). Herbicides that are used to suppress terrestrial vegetation can negatively impact these habitat functions (USFWS and NMFS 1999). In addition, insecticides applied to forests to control pests can interfere with the smoltification process of Atlantic salmon, preventing some fish from successfully making the transition from fresh to salt water. Matacil, one pesticide used in the Maine timber industry, is known to contain an endocrine disrupting chemical (USFWS and NMFS 1999). These chemicals act as "environmental hormones" that may mimic the function of the sex hormones androgen and estrogen (Thurberg and Gould 2005). Refer to the Chemical Effects: Water Discharge Facilities chapter for more information on endocrine disruptors. Other possible affects to Atlantic salmon from pesticides may include altered chemical perception of home stream odor and osmoregulatory ability (USFWS and NMFS 1999).

Conservation measures and best management practices for silviculture and timber harvest activities

1. Encourage timber operations to be located as far from aquatic habitats as possible. Buffer zones of 100 ft for first- and second-order streams and greater than 600 feet for fourth- and fifth-order streams are recommended.
2. Ensure that all silviculture and timber operations incorporate conservation plans that include control of nonpoint source pollution, protecting important habitat through landowner agreements, maintaining riparian corridors, and monitoring and controlling pesticide use.
3. Incorporate watershed analysis into timber and silviculture projects. Attention should be given to the cumulative effects of past, present, and future timber sales within a watershed.
4. Logging roads should be sited to avoid sensitive areas such as streams, wetlands, and steep slopes.
5. Include BMPs for timber forest road construction plans, including erosion control, avoidance of side casting of road materials into streams, and using only native vegetation in stabilization plantings.
6. 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.

Timber and Paper Mill Processing Activities

Introduction

Timber and paper mill processing activities can affect riverine and estuarine habitats through both chemical and physical means. Timber and lumber processing can release sawdust and wood chips in riverine and estuarine environments where they may impact the water column and benthic habitat of fish and invertebrates. These facilities may also either directly or indirectly release contaminants, such as tannins and lignin products, into aquatic habitats (USFWS and NMFS 1999). Pulp manufacturing converts wood chips or recycled paper products into individual fibers by chemical and/or mechanical means, which are then used to produce various paper products. Paper and pulp mills use and can release a number of chemicals that are toxic to aquatic organisms, including chlorine, dioxins, and acids (Mercer et al. 1997), although a number of these chemicals have been reduced or eliminated from the effluent stream by increased regulations regarding their use.

Chemical contaminant releases

Approximately 80% of all US pulp tonnage comes from kraft or sulfate pulping which uses sodium-based alkaline solutions, such as sodium sulfide and sodium hydroxide (USEPA 2002b). Kraft pulping reportedly involves less release of toxic chemicals, compared to other processes such as sulfite pulping (USEPA 2002b). Paper and pulp mills may also release a number of toxic chemicals used in the process of bleaching pulp for printing and wrapping paper products. The bleaching process may use chlorine, sulfur derivatives, dioxins, furans, resin acids, and other chemicals that are known to be toxic to aquatic organisms (Mercer et al. 1997). These chemicals have been implicated in various abnormalities in fish, including skin and organ tissue lesions, fin necrosis, gill hyperplasia, elevated detoxifying enzymes, impaired liver functions, skeletal deformities, increased incidence of parasites, disruption of the immune system, presence of tumors, and impaired growth and reproduction (Barker et al. 1994; Mercer et al. 1997). Because of concern about the release of dioxins and other contaminants, considerable improvements in the bleaching process have reduced or eliminated the use of elemental chlorine. Approximately 96% of all bleached pulp production uses chlorine-free bleaching technologies (USEPA 2002b).

An endocrine disrupting chemical, 4-nonylphenol, has been used in pulp and paper mill plants in Maine and has been shown to interfere with smoltification processes and the chemical perception of home range, and osmoregulatory ability in Atlantic salmon (USFWS and NMFS 1999). Other studies have implicated pulp and paper effluents in altered egg production, gonad development, sex steroids, secondary sexual characteristics, and vitellogenin concentration in male fish, which is considered to be an indicator of estrogenicity (Kovacs et al. 2005). A study investigating the prevalence of a microsporan parasite found in winter flounder in Newfoundland (Canada) waters observed infestations in the liver, kidney, spleen, heart, and gonads of fish collected downstream from pulp and paper mills, whereas fish collected from pristine sites harbored cysts of the parasite in only the digestive wall (Khan 2004). In addition, flounder with a high prevalence of parasite infections throughout multiple organs were found to have significant impairments to growth, organ mass, reproduction, and survival that were not observed in fish sampled from pristine locations, suggesting a link between those affects and effluent discharged by the pulp and paper mills (Khan 2004).

Entrainment and impingement

Pulp and paper mills require large amounts of water and energy in the manufacturing process. For example, a bleached kraft pulp mill can utilize 4,000-12,000 gallons of water per ton of pulp produced (USEPA 2002b). Diverting water from streams, rivers, and estuaries for pulp and paper mills can entrain and impinge eggs, larvae, and juveniles and may impact local populations of fish and invertebrates. Information is not available on the potential magnitude of entrainment and impingement impacts from wood, pulp, and paper mills. Refer to Physical Effects: Water Intake and Discharge Facilities for more information on entrainment and impingement impacts.

Thermal discharge

Pulp and paper production involves thermal and chemical processing to convert wood fibers to pulp or paper and may result in the release of effluent water with higher than ambient temperatures. There is a potential for cold-water fish such as Atlantic salmon and rainbow smelt to be adversely affected by these facilities. However, information is not available on the potential magnitude of thermal discharge impacts from wood, pulp, and paper mills.

Reduced dissolved oxygen

Pulp and paper mill wastewaters generally contain sulfur compounds with a high biological oxygen demand (BOD), suspended solids, and tannins (USEPA 2002b). The release of these contaminants in mill effluent can reduce dissolved oxygen in the receiving waters. According to the US EPA, however, all kraft pulp mills and nearly all US paper mills have chemical recovery systems in place and primary and secondary wastewater treatment systems installed to remove particulates and BOD (USEPA 2002b).

Conversion of benthic substrate

Sawdust and pulp from sawmills and lumber processing facilities can enter streams and rivers, adversely affecting benthic habitats for anadromous fish (Moring 2005). Pulp and paper mill effluent can contain solid particulates and a high BOD that can alter the benthic habitat of receiving water bodies. The impacts to benthic habitat from past practices of wood, pulp, and paper mills are evident today in some streams and rivers of Maine, including the Penobscot River from Winterport to Bucksport (USFWS and NMFS 1998). Most of the bottom substrate in this stretch of the Penobscot River is covered by bark and sawdust, which substantially reduces the diversity of benthic organisms (USFWS and NMFS 1998). However, chemical recovery systems and wastewater treatment systems should reduce or eliminate most solid wastes from the effluent stream.

Alteration of light regimes

Lumber, pulp, and paper mills releasing effluent containing solids, a high BOD, and tannins can reduce water clarity and alter the light regimes in receiving waters. This can adversely affect primary production and SAV in riverine and estuarine habitat where these facilities are located. Information is not available on the potential magnitude of light regime impacts from wood, pulp, and paper mills.

Conservation measures and best management practices for timber and paper mill processing activities

1. Ensure that lumber, pulp, and paper mills have adequate chemical recovery systems and wastewater treatment systems installed to reduce or eliminate most toxic chemicals and solid wastes from the effluent stream. Ensure that effluent streams do not elevate the ambient water temperatures of the receiving water bodies.
2. Discourage the construction of new lumber, pulp, and paper mills adjacent to riverine and estuarine waters that contain productive fisheries resources. New facilities should be sited so as to avoid the release of effluents in wetlands and open water habitats.
3. 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.

4. Incorporate watershed analysis into new lumber, pulp, and paper mill facilities, with consideration for the cumulative effects of past, present, and future impacts within the watershed.

References for Agriculture and Silviculture

[ASMFC] Atlantic States Marine Fisheries Commission. 1992. Fishery management plan for inshore stocks of winter flounder. Washington (DC): ASMFC. FMR No. 21. 138 p.

Barker DE, Khan RA, Hooper R. 1994. Bioindicators of stress in winter flounder, *Pleuronectes americanus*, captured adjacent to a pulp and paper mill in St. George's Bay, Newfoundland. Canadian Journal of Fisheries and Aquatic Sciences 51(10):2203-9.

Bejda AJ, Phelan BA, Studholme AL. 1992. The effects of dissolved oxygen on growth of young-of-the-year winter flounder. Environmental Biology of Fishes 34:321-7.

Beschta RL, Bilby RE, Brown GW, Holtby LB, Hofstra TD. 1987. Stream temperature and aquatic habitat: fisheries and forestry interactions. In: Salo EO, Cundy TW, editors. Streamside management: forestry and fishery interactions. Seattle (WA): University of Washington, College of Forest Resources. p 191-232.

Brodeur JC, Sherwood G, Rasmussen JB, Hontela A. 1997. Impaired cortisol secretion in yellow perch (*Perca flavescens*) from lakes contaminated by heavy metals: *in vivo* and *in vitro* assessment. Canadian Journal of Fisheries and Aquatic Sciences 54(12):2752-8.

Buchsbaum R. 2005. The role of overfishing, pollution, and habitat degradation on marine fish and shellfish populations of New England: summary and conclusions. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. 175 p.

Buck EH, Copeland C, Zinn JA, Vogt DU. 1997. *Pfiesteria* and related harmful blooms: natural resource and human health concerns. [Internet]. Washington (DC): National Council for Science and the Environment. Congressional Research Service Report for Congress 97-1047 ENR. [cited 2008 Jul 9]. Available from: http://www.cnie.org/NLE/CRSreports/marine/mar-23.cfm.

Castro MS, Driscoll CT, Jorden TE, Reay WG, Boynton WR. 2003. Sources of nitrogen to estuaries in the United States. Estuaries 26(3):803-14.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Duda AM. 1985. Environmental and economic damage caused by sediment from agricultural nonpoint sources. Water Resources Bulletin 21(2):225-34.

Furniss MJ, Roelofs TD, Yee CS. 1991. Road construction and maintenance. In: Meehan WR, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. Special Publication 19 ed. Bethesda (MD): American Fisheries Society. p 297-323.

Gould E, Clark PE, Thurberg FP. 1994. Pollutant effects on dermersal fishes. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 30-41.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Hicks BJ, Hall JD, Bisson PA, Sedell JR. 1991. Responses of salmonids to habitat changes. In: Meehan WR, editor. Influences of forest and rangeland management on salmonid fishes and their habitat. Special Publication 19 ed. Bethesda (MD): American Fisheries Society. p 483-518.

Howarth RW, Sharpley A, Walker D. 2002. Sources of nutrient pollution to coastal waters in the United States: implications for achieving coastal water quality goals. Estuaries 25(4b):656-76.

Johnson KL. 1992. Management for water quality on rangelands through best management practices: the Idaho approach. In: Naiman RJ, editor. Watershed management: balancing sustainability and environmental change. New York (NY): Springer-Verlag. p 415-41.

Karuppiah M, Gupta G. 1996. Impact of point and nonpoint source pollution on pore waters of two Chesapeake Bay tributaries. Ecotoxicology and Environmental Safety 35:81-5.

Kauffman JB, Krueger WC. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. Journal of Range Management 37(5):430-8.

Kennedy VS, Twilley RR, Kleypas JA, Cowan JH, Hare SR. 2002. Coastal and marine ecosystems and global climate change: potential effects on U.S. resources. Arlington (VA): Pew Center on Global Climate Change. 51 p.

Khan RA. 2004. Effect, distribution, and prevalence of *Glugea stephani* (Microspora) in winter flounder (*Pleuronectes americanus*) living near two pulp and paper mills in Newfoundland. Journal of Parasitology 90(2):229-33.

Kovacs T, Martel P, Ricci M, Michaud J, Voss R. 2005. Further insights into the potential of pulp and paper mill effluents to affect fish reproduction. Journal of Toxicology and Environmental Health 68(Part A):1621-41.

Larimore RW, Smith PW. 1963. The fishes of Champaign County, Illinois, as affected by 60 years of stream changes. Illinois Natural History Survey Bulletin 28:299-382.

Larsen PF. 1992. An overview of the environmental quality of the Gulf of Maine. In: The Gulf of Maine. Silver Spring (MD): NOAA Coastal Ocean Program Synthesis Series No. 1. p 71-95.

MacKenzie CL Jr. 1983. To increase oyster production in the northeastern United States. Marine Fisheries Review 45(3):1-22.

MacKenzie CL Jr. 2005. Removal of sea lettuce, *Ulva* sp., in estuaries to improve the environments for invertebrates, fish, wading birds, and eelgrass, *Zostera marina*. Marine Fisheries Review 67(4):1-8.

MacKenzie CL Jr. 2007. Causes underlying the historical decline in eastern oyster (*Crassostrea virginica* Gmelin, 1791) landings. Journal of Shellfish Research 26(4):927-38.

Markham VD. 2006. U.S. national report on population and the environment. New Canaan (CT): Center for Environment and Population. 67 p.

Mercer IRG, Barker DE, Khan RA. 1997. Stress-related changes in cunner, *Tautogolabrus adspersus*, living near a paper mill. Bulletin of Environmental Contamination and Toxicology 58(3):442-7.

Moring J. 2005. Recent trends in anadromous fishes. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 25-42.

[NOAA] National Oceanic and Atmospheric Administration. 1997. NOAA's estuarine eutrophication survey. Vol. 2: Mid-Atlantic Region. Silver Spring (MD): Office of Ocean Resources Conservation and Assessment. 51p.

[NRC] National Research Council. 2000. Clean coastal waters: understanding and reducing the effects of nutrient pollution. Washington (DC): National Academy Press. 405 p.

O'Reilly JE. 1994. Nutrient loading and eutrophication. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 25-30.

Platts WS. 1991. Livestock grazing. In: Meehan WR, editor. Influences of forest and rangeland management on salmonid fishes and their habitats. Bethesda (MD): American Fisheries Society. Special Publication 19. p 389-423.

Scott GI, Fulton MH, Moore DW, Wirth EF, Chandler GT, Key PB, Daugomah JW, Strozier ED, Devane J, Clark JR, and others. 1999. Assessment of risk reduction strategies for the management of agricultural non-point source pesticide runoff in estuarine ecosystems. Toxicology and Industrial Health 15:200-13.

Short FT, Burdick DM, Wolf JS, Jones GE. 1993. Eelgrass in estuarine research reserves along the East Coast, U.S.A. Part I: declines from pollution and disease and Part II: management of eelgrass meadows. Silver Spring (MD): NOAA Coastal Ocean Program Publication. 107 p.

Shumway SE, Kraeuter JN, editors. 2000. Molluscan shellfish research and management: charting a course for the future. Final Proceedings from the Workshop; 2000 Jan; Charleston, SC. Washington (DC): Department of Commerce. 156 p.

Sigman M. 1985. Impacts of clearcut logging on the fish and wildlife resources of southeast Alaska. Juneau (AK): Alaska Dept. of Fish and Game, Habitat Division. Technical Report 85-3.

Thurberg FP, Gould E. 2005. Pollutant effects upon cod, haddock, pollock, and flounder of the inshore fisheries of Massachusetts and Cape Cod Bays. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 43-66.

[USDA] US Department of Agriculture. 2005. Land use, value, and management: urbanization and agricultural land. [Internet]. Washington (DC): Economic Research Service. [updated 2005 Jun 28; cited 2007 Dec 21]. Available from: http://www.ers.usda.gov/Briefing/LandUse/urbanchapter.htm.

[USEPA] US Environmental Protection Agency. 2002a. National water quality inventory: 2000 Report to Congress. [Internet]. Washington (DC): US EPA Office of Water. EPA-841-R-02-001. [cited 2008 Jul 9]. Available from: http://www.epa.gov/305b/2000report/.

[USEPA] US Environmental Protection Agency. 2002b. Profile of the pulp and paper industry, 2nd ed. [Internet]. Washington (DC): US EPA Office of Compliance Sector Notebook Project. EPA-310-R-02-002. [cited 2008 Jul 24]. 135 p. Available from: http://www.epa.gov/compliance/resources/publications/assistance/sectors/notebooks/pulp.html.

[USEPA] US Environmental Protection Agency. 2003a. National management measures for the control of non-point pollution from agriculture. [Internet]. Washington (DC): US EPA Office of Water. EPA-841-B-03-004. [cited 2008 Jul 9]. Available from: http://www.epa.gov/owow/nps/agmm/index.html.

[USEPA] US Environmental Protection Agency. 2003b. Technical support document for identification of Chesapeake Bay designated uses and attainability. [Internet]. Annapolis (MD): US EPA, Region III, Chesapeake Bay Program Office. EPA 903-R-03-004. [cited 2008 Jul 24]. 207 p. Available from: http://www.epa.gov/region03/chesapeake/uaasupport.htm.

[USEPA] US Environmental Protection Agency. 2005. National management measures to control nonpoint source pollution from urban areas. Washington (DC): US EPA Office of Water. EPA-841-B-05-004. 518 p.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1998. Status review of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*). [Internet]. Hadley (MA): USFWS. [cited 2008 Jul 24]. 93 p. + appendices. Available from: http://www.nmfs.noaa.gov/pr/pdfs/statusreviews/atlanticsturgeon.pdf.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1999. Status review of anadromous Atlantic salmon in the United States. Hadley (MA): USFWS. 131 p.

[USGS] US Geological Survey. 1999. The quality of our Nation's waters: nutrients and pesticides. [Internet]. Reston (VA): USGS Circular 1225. 82 p. [cited 2007 Dec 28]. Available from: http://pubs.usgs.gov/circ/circ1225/pdf/index.html.

Waldichuk M. 1993. Fish habitat and the impact of human activity with particular reference to Pacific salmon. In: Parsons LS, Lear WH, editors. Perspectives on Canadian marine fisheries management. Canadian Bulletin of Fisheries and Aquatic Sciences 226: 295-337.

Whitledge TE. 1985. Nationwide review of oxygen depletion and eutrophication in estuarine and coastal waters: Northeast region. Upton (NY): Brookhaven National Laboratory.

Witman JD. 1996. Dynamics of Gulf of Maine benthic communities. In: Dow D, Braasch E, editors. The health of the Gulf of Maine ecosystem: cumulative impacts of multiple stressors. Hanover (NH): Darmouth College Regional Association for Research on the Gulf of Maine (RARGOM). Report 96-1. p 51-69.

CHAPTER TEN: INTRODUCED/NUISANCE SPECIES AND AQUACULTURE

Introduced/Nuisance Species

Introduction

Introductions of nonnative invasive species into marine and estuarine waters are a significant threat to living marine resources in the United States (Carlton 2001). Nonnative species can be released intentionally (i.e., fish stocking and pest control programs) or unintentionally during industrial shipping activities (e.g., ballast water releases), aquaculture operations, recreational boating, biotechnology, or from aquarium discharge (Hanson et al. 2003; Niimi 2004). Hundreds of species have been introduced into US waters from overseas and from other regions around North America, including finfish, shellfish, phytoplankton, bacteria, viruses, and pathogens (Drake et al. 2005). The rate of introductions has increased exponentially over the past 200 years, and it does not appear that this rate will level off in the near future (Carlton 2001).

In New England and the mid-Atlantic region, a number of fish, crabs, bryozoans, mollusks, tunicates, and algae species have been introduced since colonial times (Deegan and Buchsbaum 2005). New introductions continue to occur, such as *Convoluta convoluta*, a small carnivorous flatworm from Europe that has invaded the Gulf of Maine (Carlton 2001; Byrnes and Witman 2003); *Didemnum* sp., an invasive species of tunicate that has invaded Georges Bank and many coastal areas in New England (Pederson et al. 2005); the Asian shore crab (*Hemigrapsus sanguineus*) that has invaded Long Island Sound, NY/CT, (Carlton 2001) and other coastal areas; and *Codium fragile* spp. *tomentosoides,* an invasive algal species from Japan that has invaded the Gulf of Maine (Pederson et al. 2005).

Introduced species may thrive best in areas where there has been some level of environmental disturbance (Vitousek et al. 1997; USFWS and NMFS 1999; Minchinton and Bertness 2003). For example, in riverine systems alteration in temperature and flow regimes can provide a niche for nonnative species to invade and dominate over native species such as salmon (USFWS and NMFS 1999). Invasive species introductions can result in negative impacts to the environment and to society, with millions of dollars being expended for research, control, and management efforts (Carlton 2001).

The impacts associated with introduced/nuisance species can involve habitat, species, and genetic-level effects. Introduced/nuisance species can impact the environment in a variety of ways, including: (1) habitat alterations; (2) trophic alterations; (3) gene pool alterations; (4) alterations to communities and competition with native species; (5) introduced diseases; (6) changes in species diversity; (7) alteration in the health of native species; and (8) impacts to water quality. The following is a review of the potential environmental impacts associated with the introduction of nonnative aquatic invasive/nuisance species into marine, estuarine, and freshwater ecosystems.

Habitat alterations

Introduced species can have severe impacts on the quality of habitat (Deegan and Buchsbaum 2005). Nonnative aquatic plant species can infest water bodies, impair water quality, cause anoxic conditions when they die and decompose, and alter predator-prey relationships. Fish may be introduced into an area to graze and biologically control aquatic plant invasions. However, introduced fish may also destroy habitat, which can eliminate nursery areas for native juvenile fishes, accelerate eutrophication, and cause bank erosion (Kohler and Courtenay 1986).

Habitat has been altered by the introduction of invasive species in New England. For example, the green crab (*Carcinus maenus*) an exotic species from Europe, grazes on submerged aquatic vegeation and can interfere with eelgrass restoration efforts (Deegan and Buchsbaum 2005). *Didemnum* sp. is an invasive tunicate that has colonized the northern edge of Georges Bank, as well as many coastal areas in New England. This filter-feeding organism forms dense mats that encrust the seafloor, which can prevent the settlement of benthic organisms, reduce food availability for juvenile scallops and groundfish, and smother organisms attached to the substrate (e.g., Atlantic sea scallops [*Placopectin magellanicus*] in spat and juvenile stages) (Pederson et al. 2005; Valentine et al. 2007) and could have impacts to productive fishing grounds in New England and elsewhere. There is no evidence at this time that the spread of the tunicate on Georges Bank will be held in check by natural processes other than smothering by moving sediments; however, its offshore distribution may be limited by temperatures too low for reproduction (Valentine et al. 2007).

An invasive species of algae from Japan, *Codium fragiles* spp. *tomentosoides*, also referred to as deadman's fingers, has invaded subtidal and intertidal marine habitats in the Gulf of Maine and mid-Atlantic. Deadman's fingers can outcompete native kelp and eelgrass, thus destroying habitat for finfish and shellfish species (Pederson et al. 2005). The common reed (*Phragmites australis*) a nonnative marsh grass, has invaded coastal estuaries and can exclude native brackish and salt marsh plant species such as smooth cordgrass (*Spartina alterniflora*) from their historic habitat (Burdick et al. 2001; Minchinton and Bertness 2003; Deegan and Buchsbaum 2005). *Phragmites* invasions can increase the sedimentation rate in marshes and reduce intertidal habitat available for fish species in New England (Deegan and Buchsbaum 2005).

Trophic alterations and competition with native species

Introduced species can alter the trophic structure of an ecosystem via increased competition for food and space between native and nonnative species (Kohler and Courtenay 1986; Caraco et al. 1997; Strayer et al. 2004; Deegan and Buchsbaum 2005) as well as through predation by introduced species on native species (Kohler and Courtenay 1986). Competition may result in the displacement of native species from their habitat or a decline in recruitment, which are factors that can collectively contribute to a decrease in population size (Kohler and Courtenay 1986). For example, introductions of the invasive zebra mussel (*Dreissena polymorpha*) in the Hudson River, NY/NJ, estuary coincided with a decline in the abundance, decreased growth rate, and a shift in the population distribution of commercially and recreationally important species (Strayer et al. 2004). Zebra mussels have altered trophic structure in the Hudson River estuary by withdrawing large quantities of phytoplankton and zooplankton from the water column, thus competing with planktivorous fish. Phytoplankton is the basis of the food web, and altering the trophic levels at the bottom of the food web could have a detrimental, cascading effect on the aquatic ecosystem. Increased competition for food between the zebra mussel and open-water commercial and recreational species such as the American shad (*Alosa sapidissima*) and black sea bass (*Centropristis striata*) has been associated with large, pervasive alterations in young-of-the-year fish, which can result in interspecies competition and alterations in trophic structure (Strayer et al. 2004; Deegan and Buchsbaum 2005).

Predation on native species by nonnative species may increase the mortality of a species and could also alter the trophic structure (Kohler and Courtenay 1986). Whether the predation is on the eggs, juveniles, or adults, a decline in native forage species can affect the entire food web (Kohler and Courtenay 1986). For example, the Asian shore crab invaded Long Island Sound and has an aggressive predatory behavior and voracious appetite for crustaceans, mussels, young clams, barnacles, periwinkles, polychaetes, macroalgae, and salt marsh grasses. The removal of the forage base by this invasive crab could have a ripple effect throughout the food web that could restructure communities along the Atlantic coast (Tyrrell and Harris 2000; Brousseau and Baglivo 2005).

Alterations to communities

Introductions of nonnative species may result in alterations to communities and an increase in competition for food and habitat (Deegan and Buchsbaum 2005). For example, the green crab is an exotic species from Europe which preys on native soft-shelled clams and newly settled winter flounder (*Pseudopleuronectes americanus*) (Deegan and Buchsbaum 2005).

Nonnative marsh grass introductions can alter habitat conditions, resulting in changes in the fauna of salt marsh habitat. Alterations to communities have been noted in areas in which native marsh cordgrass habitat has been invaded by the invasive, exotic *Phragmites* (Posey et al. 2003). *Phragmites* has been implicated in alteration of the quality of intertidal habitats, including: lower abundance of nekton in *Phragmites* habitat; reduced utilization of this habitat by other species during certain life stages (Weinstein and Balletto 1999; Able and Hagan 2000); decreased density of gastropods, oligochaetes, and midges (Posey et al. 2003); decreased bird abundance and species richness (Benoit and Askins 1999); and avoidance of *Phragmites* by juvenile fishes (Weis and Weis 2000).

Gene pool alterations

Native species may hybridize with introduced species that have a different genetic makeup (Kohler and Courtenay 1986), thus weakening the genetic integrity of wild populations and decreasing the fitness of wild species via breakup of gene combinations (Goldburg et al. 2001). Aquaculture operations have the potential to be a significant source of nonnative introductions into North American waters (Goldburg and Triplett 1997; USCOP 2004). Escaped aquaculture species can alter the genetic characteristics of wild populations when native species interbreed with escaped nonnative or native aquaculture species (USFWS and NMFS 1999).

In the Gulf of Maine, the wild Atlantic salmon (*Salmo salar*) population currently exhibits poor marine survival and low spawning stock and is in danger of becoming extinct, which makes the species particularly vulnerable to genetic modification via interbreeding with escaped aquaculture species. Any genetic modification combined with other threats such as reduced water levels, parasites and diseases, commercial and recreational fisheries, loss of habitat, poor water quality, and sedimentation may threaten or potentially extirpate the wild salmon stock in the Gulf of Maine (USFWS and NMFS 1999). Refer to the Aquaculture section of this chapter for a more detailed discussion on impacts from aquaculture operations.

Introduced diseases

Introduced aquatic species are often vectors for disease transmittal that represent a significant threat to the integrity and health of native aquatic communities (Kohler and Courtenay 1986). Bacteria, viruses, and parasites may be introduced advertently or inadvertently and can reduce habitat quality (Hanson et al. 2003). The introduction of pathogens can have lethal or sublethal effects on aquatic organisms and has the potential to impair the health and fitness level of wild fish populations. Sources of introduced pathogens include industrial shipping, recreational boating, dredging activities, sediment disposal, municipal and agricultural runoff, wildlife feces, septic systems, biotechnology labs, aquariums, and transfer of oyster spat and other species to new areas for aquaculture or restoration purposes (ASMFC 1992; Boesch et al. 1997).

Parasite and disease introductions into wild fish and shellfish populations can be associated with aquaculture operations. These diseases have the potential to lower the fitness level of native species or contribute to the decline of native populations (USFWS and NMFS 1999). Examples include the MSX (multinucleated sphere unknown) oyster disease introduced through the Pacific oyster (*Crassostrea gigas*) which contributed to the decline of native oyster (*Crassostrea virginica*) populations in Delaware Bay, DE/NJ, and Chesapeake Bay, MD/VA, (Burreson et al. 2000; Rickards and Ticco 2002) and the Infectious Salmon Anemia (ISA) that has spread from salmon farms in New Brunswick, Canada, to salmon farms in Maine (USFWS and NMFS 1999). Refer to the Aquaculture section of this chapter for more information regarding diseases introduced through aquaculture operations.

Changes in species diversity

Introduced species can rapidly dominate a new area and can cause changes within species communities to such an extent that native species are forced out of the invaded area or undergo a decline in abundance, leading to changes in species diversity (Omori et al. 1994). For example, changes in species distribution have been seen in the Hudson River, where the invasion of zebra mussels caused localized changes in phytoplankton levels and trophic structure that favored littoral zone species over open-water species. The zebra mussel invasion resulted in a decline in abundance of open-water fishes (e.g., American shad) and an increase in abundance for littoral zone species (e.g., sunfishes) (Strayer et al. 2004). Shifts in the distribution and abundance of species caused by introduced species can effect the diversity of species in an area.

Alterations in species diversity have been noted in areas in which native *Spartina alterniflora* habitat has been invaded by the exotic haplotype, *Phragmites australis* (Posey et al. 2003). *Phragmites* can rapidly colonize a marsh area, thus changing the species of marsh grass present at that site. In addition, *Phragmites* invasions have been shown to change species use patterns and abundance at invaded sites, potentially causing a cascading of effects to the species richness and diversity of a community.

Benthic species diversity can be altered by the introduction of shellfish for aquaculture purposes (Kaiser et al. 1998) and for habitat restoration projects. Cultivation of shellfish such as hard clams often requires the placement of gravel or crushed shell on the substrate. Changes in benthic structure can result in a shift in the community at that site (e.g., from a polychaete to a bivalve and nemertean dominated benthic community) which may have the effect of reduced diversity (Simenstad and Fresh 1995; Kaiser et al. 1998). However, community diversity may be enhanced by the introduction of aquaculture species and/or the modification of the substrate (Simenstad and Fresh 1995). In addition, changes in species diversity may occur as a result of oyster habitat restoration. Oyster reefs provide habitat for a variety of resident and transient species (Coen et al. 1999), so restoration activities that introduce oysters into an area may result in localized changes in species diversity, as reef-building organisms and fish are attracted to the restoration site. Refer to the section on Aquaculture of this chapter for more information regarding altered species diversity caused by aquaculture activities.

Alterations in the health of native species

The health of native species can be impaired by the introduction of new species into an area. A number of factors may contribute to reduced health of native populations, including: (1) competition for food may result in a decrease in the growth rate and local abundance (Strayer et al. 2004) or the decline in the entire population (USFWS and NMFS 1999) of native species; (2) aggressive and fast growing nonnative predators can reduce the populations of native species (Pederson et al. 2005); (3) diseases represent a significant threat to the integrity and health of native aquatic communities and can decrease the sustainability of the native population (Kohler and

Courtenay 1986; USFWS and NMFS 1999; Rickards and Ticco 2002; Hanson et al. 2003); and (4) the genetic integrity of native species may be compromised through hybridization with introduced species (Kohler and Courtenay 1986), which can also decrease the fitness of wild species via breakup of gene combinations (Goldburg et al. 2001). The factors listed above, in combination with potential impact on the habitats of native species, can collectively result in long-term impacts to the health of native species (Burdick et al. 2001; Minchinton and Bertness 2003; Deegan and Buchsbaum 2005; Pederson et al. 2005).

Impacts to water quality

Invasive species can affect water quality in marine, estuarine, and riverine environments because they have the potential to outcompete native species and dominate habitats. For example, nonnative aquatic plant species, which may not have natural predators in their new environments, can proliferate within water bodies, impair water quality, and cause anoxic conditions when they die and decompose. Fish species such as grass carp (*Ctenopharyngodon idella*) and tilapia (Cichlidae), introduced to control noxious weeds, can accelerate eutrophication through fecal decomposition of nutrients previously stored in the plants (Kohler and Courtenay 1986). In addition, fish introduced to control invasive plant species can increase turbidity in the water column from the grazing behavior itself (Kohler and Courtenay 1986).

Introduced nonnative algal species from anthropogenic sources such as ballast water and shellfish transfer (e.g., seeding) combined with nutrient overloading may increase the intensity and frequency of algal blooms. An overabundance of algae can degrade water quality when they die and decompose, which depletes oxygen levels in an ecosystem. Oxygen depletion can result in ecological "dead zones," reduced light transmittance in the water column, seagrass and coral habitat degradation, and large-scale fish kills (Deegan and Buchsbaum 2005).

Conservation measures and best management practices for impacts on aquatic habitats from introduced/nuisance species

1. Do not introduce exotic species for aquaculture purposes unless a thorough scientific evaluation and risk assessment is performed. Aquaculturist should be encouraged to only culture native species in open-water operations.
2. Prevent or discourage boaters, anglers, aquaculturists, traders, and other potential handlers of introduced species from accidental or purposeful introduction of species into ecosystems where these species are not native. In addition, measures should be taken to prevent the movement or transfer of exotic species into other waters.
3. Encourage vessels to perform a ballast water exchange in marine waters (in accordance with the US Coast Guard's voluntary regulations) to minimize the possibility of introducing exotic species into estuarine habitats. Ballast water taken on in marine waters will contain fewer organisms, and these organisms will be less likely to become invasive in estuarine conditions than are species transported from other estuaries.
4. Discourage vessels that have not performed a ballast water exchange from discharging their ballast water into estuarine receiving waters.
5. Require vessels brought from other areas over land via trailering to clean any surfaces that may harbor nonnative plant or animal species (e.g., propellers, hulls, anchors, fenders). Bilges should be emptied and cleaned thoroughly with hot water or a mild bleach solution. These activities should be performed in an upland area to prevent introduction of nonnative species to aquatic environments during the cleaning process.

6. Encourage natural resource managers to provide outreach materials on the potential impacts resulting from releases of nonnative species into the natural environment.
7. Limit importation of ornamental fishes to licensed dealers.
8. Use only local, native fish for live seafood or bait.
9. Encourage natural resource managers to identify areas where invasive species have become established at an early time in the infestation and pursue efforts to remove them, either manually or by other methods.
10. Encourage natural resource managers to identify methods that eradicate or reduce the spread of invasive species (e.g., reducing *Phragmites* in coastal marshes by mitigating the effects of tidal restrictions).
11. Treat effluent from public aquaria displays, laboratories, and educational institutes that are using exotic species prior to discharge for the purpose of preventing the introduction of viable animals, plants, reproductive material, pathogens, or parasites into the environment.

Aquaculture

Introduction

Aquaculture is defined as the controlled cultivation and harvest of aquatic organisms, including finfish, shellfish, and aquatic plants (Goldburg et al. 2001, 2003). Aquaculture operations are conducted at both land and water facilities. Land-based aquaculture systems include ponds, tanks, raceways, and water flow-through and recirculating systems. Water-based aquaculture systems include netpens, cages, ocean ranching, longline culture, and bottom culture (Goldburg and Triplett 1997).

Aquaculture can provide a number of socio-economic benefits, including food provision, improved nutrition and health, generation of income and employment, diversification of primary products, and increased trade earnings through the export of high-value products (Barg 1992). Aquaculture can also provide environmental benefits by supporting stocking and release of hatchery-reared organisms, countering nutrient and organic enrichment in eutrophic waters from the culture of some mollusk and seaweed species, and because aquaculture operations relies on good water quality, the prevention and control of aquatic pollution (Barg 1992).

However, freshwater, estuarine, and marine aquaculture operations have the potential to adversely impact the habitat of native fish and shellfish species. The impact of aquaculture facilities varies according to the species cultured, the type and size of the operation, and the environmental characteristics of the site. Intensive cage and floating netpen systems typically have a greater impact because aquaculture effluent is released directly into the environment. Pond and tank systems are less harmful to the environment because waste products are released in pulses during cleaning and harvesting activities rather than continuously into the environment (Goldburg et al. 2001). The relative impact of finfish and shellfish aquaculture differs depending on the foraging behavior of the species. Finfish require the addition of a large amount of feed into the ecosystem, which can result in environmental impacts impacts from the introduction of the feed, but also from the depletion of species harvested to provide the feed. Bivalves are filter feeders and typically do not require food additives; however, fecal deposition can result in benthic and pelagic habitat impacts, changes in trophic structure (Kaspar et al. 1985; Grant et al. 1995), and nutrient and phytoplankton depletion (Dankers and Zuidema 1995).

Similar to the introduced/nuisance species section of this chapter, aquaculture activities can effect fisheries at both a habitat and species-level. Typical environmental impacts resulting from aquaculture production include: (1) impacts to the water quality from the discharge of organic wastes and contaminants; (2) seafloor impacts; (3) introductions of exotic invasive species; (4) food web impacts; (5) gene pool alterations; (6) changes in species diversity; (7) sediment deposition; (8) introduction of diseases; (9) habitat replacement or exclusion; and (10) habitat conversion. The following is a review of the known and potential environmental impacts associated with the cultivation and harvest of aquatic organisms in land- and water-based aquaculture facilities.

Discharge of organic wastes

Aquaculture operations can degrade the quality of the water column and the benthic environment via the discharge of organic waste and other contaminants (Goldburg et al. 2001; USCOP 2004). Organic waste includes uneaten fish food, urine, feces, mucus, and byproducts of respiration, which can have an adverse effect on both benthic and pelagic organisms when released into marine, estuarine, and riverine environments.

Uneaten fish food can contribute a significant amount of nutrients to the ecosystem at aquaculture sites (Kelly 1992; Goldburg and Triplett 1997). Farmed fish are typically fed "forage fish" of low economic value, such as anchovies (Engraulidae) and menhaden (*Brevoortia* sp.), which are either fed directly to aquaculture species or processed into dry feed pellets. However, these "forage fish," while having low economic value, may be highly important to other species and the aquatic ecosystem. A large percentage of nutrients contained in farmed fish food are lost to the environment through organic waste. As much as 80% of total nitrogen and 70% of total phosphorus fed to farmed fish may be released into the water column through fish wastes (Goldburg et al. 2001).

In New England, the majority of aquaculture operations are located in Maine, with Cobscook Bay being the primary site of finfish aquaculture operations. Recent research in Cobscook Bay and in neighboring waters of New Brunswick, Canada, has shown the primary sources of nutrients in the area are finfish aquaculture operations and the open ocean (Goldburg et al. 2001). Research conducted at an aquaculture facility with 200,000 salmon has revealed that the amount of nitrogen, phosphorus, and feces discharged from the facility are equivalent to that released from untreated sewage produced by 20,000, 25,000, and 65,000 people, respectively (Goldburg et al. 2001).

The release of high concentrations of nutrients can negatively affect an aquatic system through eutrophication. Eutrophication of an aquatic system can occur when nutrients, such as nitrogen and phosphorus, are released in high concentrations and over long periods of time. Eutrophication can stimulate the growth of algae and other primary producers and, in some cases, may develop into "algal blooms" (Hopkins et al. 1995; Goldburg et al. 2001; Deegan and Buchsbaum 2005). Although the effects of eutrophication are not necessarily always adverse, they are often extremely undesirable and include: (1) increased incidence, extent, and persistence of noxious or toxic species of phytoplankton; (2) increased frequency, severity, spatial extent, and persistence of low oxygen conditions; (3) alteration in the dominant phytoplankton species and the nutritional-biochemical "quality" of the phytoplankton community; and (4) increased turbidity of the water column because of the presence of algae blooms (O'Reilly 1994).

Oxygen can be depleted in the water column during bacterial degradation of algal tissue or when algal respiration exceeds oxygen production and can result in hypoxic or anoxic "dead zones," reduced water clarity, seagrass habitat degradation, and large-scale fish kills (Deegan and Buchsbaum 2005). Algal blooms may contain species of phytoplankton such as dinoflagellates that can produce toxins, cause toxic blooms (e.g., red tides), kill large numbers of fish, contaminate shellfish beds, and cause health problems in humans. Coastal and estuarine ecosystems in the United States are already moderately to severely eutrophic (Goldburg et al. 2001; Goldburg and

Triplett 1997) and are expected to worsen in 70% of all coastal areas over the next two decades (USEPA 2001). Consequently, the frequency and severity of toxic algal blooms could increase in the future. Refer to the Coastal Development and Chemical Effects: Water Discharge Facilities chapters for more information on eutrophication and harmful algal blooms.

Discharge of contaminants

In addition to organic waste, chemicals and other contaminants that are discharged as part of the aquaculture process can affect benthic and pelagic organisms (Hopkins et al. 1995; Goldburg and Triplett 1997). Chemicals are typically released directly into the water, including antibiotics that fight disease; pesticides that control parasites, algae, and weeds; hormones that initiate spawning; vitamins and minerals to promote fish growth; and anesthetics to ease handling of fish during transport. These chemical agents are readily dispersed into marine, estuarine, and freshwater systems and can be harmful to natural communities. Few chemicals have been approved for disease treatment in US aquaculture operations, although veterinarians can prescribe human and animal drugs use in food fish (Goldburg et al. 2001).

Antibiotics are given to fish and shrimp via injections, baths, and oral treatments (Hopkins et al. 1995; Goldburg and Triplett 1997). The most common method of oral administration is the incorporation of drugs into feed pellets, which results in a greater dispersion of antibiotics in the marine environment. Antibiotics, including those toxic to humans, typically bind to sediment particles, may remain in the environment for an extended period of time, can accumulate in farmed and wild fish and shellfish populations, and can harm humans when ingested.

Herbicides are chemicals used to control aquatic weeds in freshwater systems, and algicides are herbicides specifically formulated to kill algae; dissolved oxygen levels in ponds can be reduced when the algae die and decompose. A common ingredient in algicides is copper, which is toxic to aquatic organisms. Applications of herbicides or algicides must be carefully considered for their toxicity to aquaculture organisms and to humans, as well as their tendency to bioaccumulate in fish and shellfish tissues (Goldburg and Triplett 1997). While these chemicals may not be applied within riverine or estuarine systems, they may find their way there through stormwater runoff. Pesticides must also be carefully monitored for their effects on aquatic organisms and habitat. For example, antifouling compounds such as copper and organic tin compounds were historically used in the aquaculture industry to prevent fouling organisms from attaching to aquaculture structures. These chemicals accumulate in farmed and wild organisms, especially in shellfish species, and the use of organic tin compounds is now banned for use in both Washington and Maine. Aquaculturalists have used the insecticide, Sevin, for 35 years in Willapa Bay, WA, to control burrowing shrimp that destabilize sediment. Sevin kills other organisms such as the Dungeness crab (*Cancer magister*), and it should be used in moderation to minimize the impacts of the aquaculture industry on other important commercial fisheries (Goldburg and Triplett 1997). For additional information on the release of pesticides, refer to the Agriculture and Silviculture and Coastal Development chapters of this report.

Seafloor impacts

Aquaculture operations not only can cause environmental impacts through the discharge of contaminants and organic wastes, but these operations can also affect the seafloor as a result of the deposition of waste products, the placement of aquaculture structures on the seafloor, and the harvesting of aquaculture species.

Aquaculture operations can have a wide range of biological, chemical, and physical impacts on seafloor habitat stemming from organic material deposition, shading effects, damage to habitat from aquaculture structures and operations, and harvesting with rakes and dredges (USFWS and NMFS 1999; Goldburg et al. 2001). Organic material deposition beneath netpens and cages can smother organisms, change the chemical and biological structure of sediment, alter species biomass and diversity, and reduce oxygen levels. The physical and chemical conditions present at the aquaculture site will influence the degree to which organic waste affects the benthic community. At aquaculture sites with slower currents and softer sediments, benthic community impacts will generally be localized; whereas sites with stronger currents and coarser sediments will generally have widely distributed but less intense benthic community effects downstream of the site.

At both land-based and water-based aquaculture facilities, accumulations of large amounts of carbon and nutrient-rich sediment may produce anaerobic conditions in sediments and cause the release of hydrogen sulfide and methane, two gases toxic to fish (Goldburg and Triplett 1997). In Maine, seafloor impacts resulting from sediment deposition at salmon farms include the growth of the bacterial mold *Beggiatoa* sp., which degrades water quality and subsequently lowers species diversity and biomass beneath the pens (Goldburg and Triplett 1997).

Suspended shellfish culture techniques may cause changes in benthic community structure similar to those conditions found under netpens. Filter-feeding shellfish "package" phytoplankton and other food particles into feces and pseudofeces, which are deposited on the seafloor and may cause local changes in benthic community structure (Grant et al. 1995; Goldburg and Triplett 1997). In Kenepuru Sound, New Zealand, a mussel aquaculture site consistently showed a higher organic nitrogen pool than at the reference site, indicating that organic nitrogen was accumulating in the sediments below the mussel farm (Kaspar et al. 1985). The benthic community at the mussel farm was composed of species adaptable to low-oxygen levels that live in fine-textured, organically rich sediments, while the reference site consisted of species that typically reside in highly oxygenated water (Kaspar et al. 1985).

Aquaculture structures can have direct impacts on seafloor habitat, including shading of seafloor habitat by netpens and cages (NEFMC 1998; USFWS and NMFS 1999). Shading can impede the growth of SAV that provides shelter and nursery habitat to fish and their prey species (Barnhardt et al. 1992; Griffin 1997; Deegan and Buchsbaum 2005). Seagrasses and other sensitive benthic habitats may also be impacted by the dumping of shells onto the seafloor for use in shellfish aquaculture operations (Simenstad and Fresh 1995). Shell substratum helps to stabilize the benthos and improve growth and survival of the cultured shellfish species. The placement of this material on the bottom not only causes a loss in seagrass and other habitat, but substrate modification also induces a localized change in benthic community composition (Simenstad and Fresh 1995).

Harvesting practices also have the potential to adversely affect seafloor habitat. Perhaps the most detrimental is the mechanical harvesting of shellfish (e.g., the use of dredges). Polychaete worms and crustaceans may be removed or buried during dredging activities (Newell et al. 1998). Mechanical harvesting of shellfish may also adversely affect benthic habitat through direct removal of seagrass and other reef-building organisms (Goldburg and Triplett 1997).

Introductions of exotic invasive species

Aquaculture operations have the potential to be a significant source of nonnative introductions into North American waters (Goldburg and Triplett 1997; USCOP 2004). The cultivation of nonnative species becomes problematic when fish escape or are intentionally released into the marine environment. As discussed in the above section on introduced/nuisance species, introduced species can reduce biodiversity, alter species composition, compete with native species for food and habitat, prey on native species, inhibit reproduction, modify or destroy habitat, and introduce new parasites or diseases into an ecosystem (Goldburg and Triplett 1997; USFWS and

NMFS 1999). Impacts from introduced aquaculture species may result in the displacement or extinction of native species, which is believed to be a contributing factor in the decline of seven endangered or threatened fish species populations listed under the Endangered Species Act (Goldburg and Triplett 1997).

In Maine, escaped aquaculture salmon can disrupt redds (i.e., spawning nests) of wild salmon, transfer disease or parasites, compete for food and habitat, and interbreed with wild salmon (USFWS and NMFS 1999). Escaped aquaculture salmon represent a significant threat to wild salmon in Maine because even at low levels of escapement, aquaculture salmon can represent a large proportion of the salmon returns in some rivers. Escaped Atlantic salmon have been documented in the St. Croix, Penobscot, East Machias, Dennys, and Narraguagus rivers in Maine. Escapees represented 89% and 100% of the documented runs for the Dennys River in 1994 and 1997, respectively, and 22% of the documented run for the Narraguagus River in 1995 (USFWS and NMFS 1999). In 2000, only 22 wild Atlantic salmon in Maine were documented as returning to spawn in their native rivers; however, total adult returning spawners may have numbered approximately 150 fish (Goldburg et al. 2001).

Cultivating a reproductively viable European stock of Atlantic salmon in Maine waters poses a risk to native populations because of escapement and the subsequent interbreeding of genetically divergent populations (USFWS and NMFS 1999). The wild Atlantic salmon population in the Gulf of Maine currently exhibits poor marine survival and low spawning stock size, is particularly vulnerable to genetic modification, and is in danger of becoming extinct. Dilution of the gene pool, when combined with environmental threats such as reduced water levels, parasites and diseases, commercial and recreational fisheries, loss of habitat, poor water quality, and sedimentation could extirpate the wild salmon stock in the Gulf of Maine (USFWS and NMFS 1999). For additional discussions on this topic, refer to the subsection in this chapter on Gene Pool Alterations.

Food web impacts

Aquaculture operations have the potential to impact food webs via localized nutrient loading from organic waste and by large-scale removals of oceanic fish for dry-pellet fish feed (Goldburg and Triplett 1997). As reviewed in previous sections of this chapter, nutrients in discharged organic waste may affect local populations by changing community structure and biodiversity. These localized changes may have broader implications to higher trophic level organisms. For example, biosedimentation at a mussel aquaculture site had a strong effect on benthic community structure both below and adjacent to mussels grown on rafts (Kaspar et al. 1985). Benthic species located beneath and adjacent to mussel rafts included sponges, tunicates, and calcareous polychaete worms, while benthic species at the reference site included bivalve mollusks, brittle stars, crustaceans, and polychaete worms. The shift in benthic community structure at the shellfish aquaculture site may have had implications in higher trophic levels in the ecosystem.

Large-scale removals of anchovy, herring, sardine, jack mackerel, and other pelagic fishes for the production of fish feed has an impact on the food web. Approximately 27% (31 million metric tons) of the world's fish harvest is now used to produce fish feeds, and about 15% of this is used in aquaculture production (Goldburg and Triplett 1997). Feeding fish to other fish on a commercial scale is highly energy-inefficient and may have environmental implications and impacts on other species. Higher trophic levels depend on small pelagic fishes for growth and survival, so the net removal of protein can have significant effects on sea birds, mammals, and commercially important fish species (Goldburg and Triplett 1997).

Gene pool alterations

Escaped aquaculture species can alter the genetic characteristics of wild populations when native species interbreed with escaped nonnative or native aquaculture species or escaped genetically engineered aquaculture species (USFWS and NMFS 1999; Goldburg et al. 2001; USCOP 2004). Interbreeding of the wild population with escaped nonnative species is problematic, as discussed in the Introduced/Nuisance Species section of this chapter. Interbreeding of the wild population with escaped, native species may also be problematic because of the genetic differences between the escaped native and the wild native populations. Aquaculture operations often breed farmed fish for particular traits, such as smaller fins, aggressive feeding behavior, and larger bodies. Therefore, the genetic makeup of escaped native and wild native fish may be different, and interbreeding may decrease the fitness of wild populations through the breakup of gene combinations and the loss of genetic diversity (Goldburg et al. 2001).

Atlantic salmon aquaculture in New England has been established from Cape Cod, MA, north to Canada, although most of this activity is clustered at the Maine-New Brunswick border. In 1994, thousands of Atlantic salmon escaped from an aquaculture facility during a storm event; many of these fish spread into coastal rivers in eastern Maine (Moring 2005). In 2000, a similar storm event in Maine resulted in the escapement of 100,000 salmon from a single farm, which is more than 1,000 times the documented number of native adult Atlantic salmon. Canada is experiencing similar problems with aquaculture escapees and the interbreeding of wild and farmed salmon populations. In 1998, 82% of the young salmon leaving the Magaguadavic River in New Brunswick originated from aquaculture farms (Goldburg et al. 2001). Escapees can and do breed with wild populations of Atlantic salmon, which is a concern because interbreeding can alter the genetic makeup of native stocks (Moring 2005).

Escaped genetically engineered aquaculture species may exacerbate the problem of altering the gene pool of native fish stocks. Genetically engineered (i.e., transgenic) species are being developed by inserting genes from other species into the DNA of fish for the purpose of altering performance, improving flesh quality, and amplifying traits such as faster growth, resistance to diseases, and tolerance to freezing temperatures (Goldburg and Triplett 1997; Goldburg et al. 2001). For example, genetically engineered Atlantic salmon have an added hormone from chinook salmon that promotes faster growth, which may reduce costs for growers (Goldburg et al. 2001, 2003). Although no transgenic fish products are commercially available in the United States, at least one company has applied for permission through the Food and Drug Administration to market a genetically-engineered Atlantic salmon for human consumption (Goldburg et al. 2001, 2003). Transgenic aquaculture escapees could impair wild Atlantic salmon stocks via competition, predation, and expansion into new regions. Interbreeding could weaken the genetic integrity of wild salmon populations and have long-term, irreversible ecological effects (Goldburg et al. 2001).

Impacts to the water column and water quality

Aquaculture may impact the water column via organic and contaminant discharge from land- and water-based aquaculture sites (NEFMC 1998). As discussed in other sections of this chapter, aquaculture discharges include nutrients, toxins, particulate matter, metabolic wastes, hormones, pigments, minerals, vitamins, antibiotics, herbicides, and pesticides. Water quality in the vicinity of finfish aquaculture operations may be impaired by the discharge of these compounds. The water column may become turbid as a result of this discharge, which can degrade overall habitat conditions for fish and shellfish in the area. Discharge may contribute to nutrient loading, which may lead to eutrophic conditions in the water column. Eutrophication often results in oxygen depletion, finfish and shellfish kills, habitat degradation, and harmful algal blooms that may impact human health.

Shellfish aquaculture operations have the potential to improve water quality by filtration of nutrients and suspended particles from the water column (Newell 1988). However, bivalves may contribute to the turbidity of the pelagic environment via their waste products (Kaspar et al. 1985; Grant et al. 1995). These waste products are expelled as feces and pseudofeces, which can be suspended into the water column, thus contributing to nutrient loads near aquaculture sites. Nutrient overenrichment often results in oxygen depletion, toxic gas generation, and harmful algal blooms, thus impairing the water quality near shellfish aquaculture sites. Therefore, both finfish and shellfish aquaculture operations have the potential to adversely affect water quality beneath aquaculture structures and in the surrounding environment. For additional information on discharge of nutrients and its subsequent effects on the water column via eutrophication and algal blooms, see the subsections on the Discharge of Organic Wastes and Discharge of Contaminants in this chapter, as well as the chapters on Agriculture and Silviculture, Coastal Development, and Alteration of Freshwater Systems of this report.

Changes in species diversity

Species diversity and abundance may change in the vicinity of aquaculture farms as a result of effluent discharges or habitat modifications that alter environmental conditions. Changes in species diversity may occur through increased organic waste in pelagic and benthic environments, modification to bottom habitat, and the attraction of predators to the farmed species. Accumulated organic waste beneath aquaculture structures may change benthic community structure. In Maine, salmon netpen aquaculture can alter the benthos by shifting microbial and macrofaunal species to those adapted to enriched organic sediments. At one netpen site, epibenthic organisms were more numerous near the pen than at reference sites, suggesting that benthic community structure can be altered by salmon aquaculture in coastal Maine waters (Findlay et al. 1995).

Cultivated mussels can alter species diversity via biodeposition. Benthic habitat can shift from communities of bivalve mollusks, brittle stars, crustaceans, and polychaete worms to communities of sponges, tunicates, and calcareous polychaete worms beneath mussel aquaculture sites. The difference between the two sites represents a change in species diversity from those that typically reside in highly oxygenated water to those species adaptable to low-oxygen levels that can live in areas with fine-textured, organically rich sediments (Kaspar et al. 1985).

Benthic habitat modification at shellfish aquaculture sites can alter species diversity (Kaiser et al. 1998). Cultivation of shellfish such as hard clams requires the placement of gravel or crushed shell on the substrate. Seed clams are placed on the substrate in bags or directly on substrate covered with protective plastic netting. Benthic structure at shellfish aquaculture sites can therefore shift from polychaete-dominated communities to bivalve and nemertean-dominated communities, which could have repercussions for other trophic levels (Simenstad and Fresh 1995; Kaiser et al. 1998). However, community diversity may be enhanced by the introduction of aquaculture species and the modification of the substrate. For example, the placement of gravel in the intertidal area, the placement of substrates suitable for macroalgal attachment, or predator exclusion nets in some habitats may enhance epibenthos diversity and standing stock (Simenstad and Fresh 1995).

Open water netpens may alter species diversity by attracting wild fish or other predators to the aquaculture site (Vita et al. 2004). Wild benthic and pelagic species are attracted to uneaten pellet feed and other discharged effluent, which can result in impacts to the food web (Vita et al. 2004). Predators such as seals, sea lions, and river otters may also be attracted to aquaculture pens to feed on farmed species, which can alter communities in the vicinity of aquaculture sites (Goldburg et al. 2001).

Sediment deposition

The effects of sediment deposition include eutrophication of the water column; toxic algal blooms; hypoxic or anoxic zones caused by microbial degradation; and the spread of contaminants such as antibiotics, herbicides, pesticides, hormones, pigments, minerals, and vitamins. The impacts of sediment deposition from discharged organic waste and contaminants on the water column and on the seafloor have been discussed in the Discharge of Organic Wastes, Discharge of Contaminants, Seaflood Impacts, Food Web Impacts, Changes in Species Diversity, and Habitat Exclusion and Replacement/Conversion subsections of this chapter.

Introduction of diseases

Parasite and disease introductions into wild fish and shellfish populations are often associated with aquaculture operations and have the potential to lower the fitness level of native species or contribute to the decline of native populations. For example, in the 1940s and 1950s, scientists inadvertently introduced a new disease into eastern US waters when they attempted to restore declining populations of the eastern oyster (*Crassostrea virginica*) via the introduction of the Pacific oyster (*Crassostrea gigas*) (Burreson et al. 2000; Rickards and Ticco 2002). *Haplosporidium nelsoni* is a protistan parasite that causes MSX oyster disease and was present amongst the Pacific oysters introduced in east coast waters. MSX spread from Delaware Bay to the Chesapeake Bay and contributed to the decline in the native oyster population. MSX and another pathogenic disease, Dermo (*Perkinsus marinus*), have collectively decimated the native oyster population remaining along the much of the eastern US coast (Rickards and Ticco 2002).

In eastern Maine and New Brunswick, an outbreak of two diseases in both wild and cultured stocks of Atlantic salmon suggests that cultured stocks are acting as reservoirs of diseases and are now passing them on to wild stocks (Moring 2005). In addition to diseases, sea lice are a flesh-eating parasite that has been passed from farmed salmon to wild salmon when wild salmon migrate through coastal waters. Sea lice also can serve as a host for Infectious Salmon Anemia (ISA), which is a virus that has spread from salmon farms in New Brunswick to salmon farms in Maine (USFWS and NMFS 1999). The ISA virus causes fatalities in salmon at aquaculture facilities, and this virus has been detected in both escaped farmed salmon and wild salmon populations. ISA first appeared in New Brunswick in 1996, was detected in the United States in 2001, and represents a significant threat to wild salmon populations (Goldburg et al. 2001).

Habitat exclusion and replacement/conversion

Aquaculture operations require the use of space, which results in the conversion of natural aquatic habitat that could have been used by native organisms for spawning, feeding, and growth. Approximately 321,000 acres of fresh water habitat and 64,000 acres of salt-water habitat have been converted for use in aquaculture operations in the United States (Goldburg et al. 2001). Aquaculture facilities may exclude aquatic organisms from their native habitat through the placement of physical barriers to entry or through changes in environmental conditions at aquaculture sites. Nets, cages, concrete, and other barriers exclude aquatic organisms from entering the space in which the aquaculture structures are placed. By effectively acting as physical barriers for wild populations, these formerly usable areas are no longer available as habitat for fish and shellfish species to carry out their life cycles. Aquaculture facilities may physically exclude wild stocks of fish, such as Atlantic salmon, from reaching critical spawning habitat upstream of the facilities (Goldburg et al. 2001).

Changes in environmental conditions at the aquaculture site may also exclude aquatic organisms from their native habitat. Discharge of organic waste and contaminants beneath aquaculture netpens and cages may render pelagic and benthic habitat unusable through nutrient loading and the subsequent effects of eutrophication. Low dissolved oxygen caused by eutrophication may force native species out of their habitat, while harmful algal blooms can cause widespread fish kills or exclude fish from areas affected by the outbreak (Goldburg and Triplett 1997). In the case of large shellfish aquaculture operations, filtering bivalves can also decrease the amount and type of nutrients and phytoplankton available to other species. This reduction in nutrients and phytoplankton can stimulate competition between populations of cultured and native species (Dankers and Zuidema 1995). Nutrient and phytoplankton removal could have a cascade effect on the trophic structure of the ecosystem (NEFMC 1998), which may eventually cause mobile species to relocate to other areas. Nonetheless, bivalves grown in open-water mariculture facilities can provide similarily beneficial filtering functions as native bivalves by contributing to the control nutrients, suspended sediments, and water column phytoplankton dynamics.

Aquaculture can result in the replacement or conversion of the natural benthic and pelagic community in the area surrounding the facility. For example, shellfish aquaculture can eliminate seagrass beds when shell material is dumped on the seafloor (Simenstad and Fresh 1995). Seagrass beds in the vicinity of shellfish culture operations may be eliminated during harvesting, which may temporarily reduce levels of biodiversity by reducing habitat for other marine species. Habitat conversion also takes place at netpen sites in which sediment deposition causes underlying habitat to become eutrophic. Sensitive benthic habitats beneath the netpens, such as seagrasses, may be eliminated or degraded by poor water quality conditions, thus converting viable habitat to unusable or less productive seafloor area (Goldburg and Triplett 1997).

Although the effects of replacement and exclusion of habitat by aquaculture facilities are often negative, there may be some positive effects of the structures. For example, cages, anchoring systems, and other devices can increase the structural complexity to the benthic and pelagic environment, which can provide shelter and foraging habitat for some native species. Open-water shellfish mariculture operations can provide some of the same habitat benefits as natural shellfish beds, such as refugia from predation and feeding habitat for juvenile and adult mobile species. Under some conditions, seafloor productivity may increase near aquaculture sites.

Conservation measures and best management practices for aquaculture

1. Assess the aquatic resources in the area when siting new aquaculture facilities, including benthic communities, the proximity to wild stocks, migratory corridors, competing resource uses (e.g., commercial fishing, recreational uses, other aquaculture facilities), hydrographic conditions, and upstream habitat uses.
2. Avoid siting of aquaculture operations in or near sensitive benthic communities, such as submerged aquatic vegetation.
3. Avoid enclosing or impounding tidally influenced wetlands for mariculture purposes.
4. Ensure that aquaculture operations adequately address disease issues to minimize risks to wild stocks.
5. Employ methods to minimize escape from culture facilities to minimize potential genetic impacts and to prevent disruption of natural aquatic communities.
6. Design aquaculture facilities to meet applicable environmental standards for wastewater treatment and sludge control.

7. Locate aquaculture facilities to minimize discharge effects on habitat and locate water intakes to minimize entrainment of native fauna.
8. Evaluate and control the use of antibiotics, pesticides, and herbicides in aquaculture operations. Avoid direct application of carbaryl or other pesticides in water.
9. Consider biological controls to reduce pest populations, such as small, native species that feed on sea lice and fouling organisms.
10. Reduce the metabolic stress of aquaculture species in order to eliminate or reduce the need for using chemicals. Measures to reduce stress include improving water quality, lowering stock densities, and minimizing handling of fish.
11. Use aquaculture gear designed to minimize entanglement of native species attracted to the aquaculture operation (e.g., predators, such as marine mammals and birds).
12. Exclude exotic species from aquaculture operations until a thorough scientific evaluation and risk assessment is performed.
13. Locate aquaculture facilities rearing nonnative species upland and use closed-water circulation systems.
14. Treat effluent from public aquarium displays, laboratories, and educational institutes that are using exotic species prior to discharge for the purpose of preventing the introduction of viable animals, plants, reproductive material, pathogens, or parasites into the environment.
15. Consider growing several cultured species together, such as finfish, shellfish, algae, and hydroponic vegetables to reduce nutrient and sediment loads on the ecosystem.
16. Develop a monitoring program at the site to evaluate habitat and water quality impacts and the need for corrective measures through adaptive management.

References for Introduced/Nuisance Species and Aquaculture

Able KW, Hagan SM. 2000. Effects of common reed (*Phragmites australis*) invasion on marsh surface macrofauna: response of fishes and decapod crustaceans. Estuaries 23:633-46.

[ASMFC] Atlantic States Marine Fisheries Commission. 1992. Fishery management plan for inshore stocks of winter flounder. Washington (DC): ASMFC. FMR No. 21. 138 p.

Barg UC. 1992. Guidelines for the promotion of environmental management of coastal aquaculture development. [Internet]. Rome (Italy): Food and Agriculture Organization of the United Nations (FAO). FAO Fisheries Technical Paper 328. [cited 2008 Jul 24]. 122 p. Available from: http://www.fao.org/docrep/T0697E/t0697e01.htm.

Barnhart RA, Boyd MJ, Pequegnat JE. 1992. The ecology of Humboldt Bay, California: an estuarine profile. Arcata (CA): US Fish and Wildlife Service. USFWS Biological Report 1. 121 p.

Benoit LK, Askins RA. 1999. Impact of the spread of *Phragmites* on the distribution of birds in Connecticut marshes. Wetlands 19:194-208.

Boesch DF, Anderson DM, Horner RA, Shumway SE, Tester PA, Whitledge TE. 1997. Harmful algal blooms in coastal waters: options for prevention, control and mitigation. Silver Spring (MD): NOAA Coastal Ocean Office. NOAA Coastal Ocean Program Decision Analysis Series No. 10. 46 p + appendix.

Brousseau DJ, Baglivo JA. 2005. Laboratory investigations of food selection by the Asian shore crab, *Hemigrapsus sanguineus*: algal versus animal preference. Journal of Crustacean Biology 25(1):130-4.

Burdick DM, Buchsbaum R, Holt E. 2001. Variation in soil salinity associated with expansion of *Phragmites australis* in salt marshes. Environmental and Experimental Botany 46(3):247-61.

Burreson EM, Stokes NA, Friedman CS. 2000. Increased virulence in an introduced pathogen: *Haplosporidium nelsoni* (MSX) in the Eastern oyster *Crassostrea virginica*. Journal of Aquatic Animal Health 12(1):1-8.

Byrnes J, Witman JD. 2003. Impact assessment of an invasive flatworm, *Convoluta convoluta*, in the southern Gulf of Maine. Journal of Experimental Marine Biology and Ecology 293(2):173-91.

Caraco NF, Cole JJ, Raymond PA, Strayer DL, Pace ML, Findlay SEG, Fischer DT. 1997. Zebra mussel invasion in a large, turbid river: phytoplankton response to increased grazing. Ecology 78(2):588-602.

Carlton JT. 2001. Introduced species in U.S. coastal waters: Environmental impacts and management priorities. Arlington (VA): Pew Oceans Commission. 28 p.

Coen LD, Luckenback MW, Breitburg DL. 1999. The role of oyster reefs as essential fish habitat: A review of current knowledge and some new perspectives. In: Benaka LR, editor. Proceedings of the Sea Grant symposium on fish habitat and rehabilitation. Bethesda (MD): American Fisheries Society Symposium, No. 22. p 438-54.

Dankers N, Zuidema DR. 1995. The role of the mussel (*Mytilus edulis* L.) and mussel culture in the Dutch Wadden Sea. Estuaries 18(1A):71-80.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Drake LA, Jenkins PT, Dobbs FC. 2005. Domestic and international arrivals of NOBOB (no ballast on board) vessels to lower Chesapeake Bay. Marine Pollution Bulletin 50(5):560-5.

Findlay RH, Watling L, Mayer M. 1995. Environmental impact of salmon net-pen culture on marine benthic communities in Maine: a case study. Estuaries 18(1A):145-79.

Goldburg RJ, Elliott MS, Naylor RL. 2001. Marine aquaculture in the United States: environmental impacts and policy options. Arlington (VA): Pew Oceans Commission.

Goldburg RJ, Elliott MS, Naylor RL. 2003. Guiding sustainable marine aquaculture. In: Pew Ocean Commission. America's living oceans: charting a course for sea change. Arlington (VA): Pew Oceans Commission. p 73-79.

Goldburg R, Triplet T. 1997. Murky waters: environmental effects of aquaculture in the United States. Washington (DC): Environmental Defense Fund.

Grant J, Hatcher A, Scott DB, Pocklington P, Schafer CT, Winters GV. 1995. A multi-disciplinary approach to evaluating impacts of shellfish aquaculture on benthic communities. Estuaries 18(1A):124-44.

Griffin K. 1997. Commercial oyster cultivation and eelgrass ecology in Tillamook Bay, Oregon: a literature review and synthesis. Garibaldi (OR): Tillamook Bay National Estuary Project. 36 p.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Hopkins JS, Sandifer PA, Devoe MR, Holland AF, Browdy CL, Stokes AD. 1995. Environmental impacts of shrimp farming with special reference to the situation in the continental United States. Estuaries 18(1A):25-42.

Kaiser MJ, Laing I, Utting SD, Burnell GM. 1998. Environmental impacts of bivalve mariculture. Journal of Shellfish Research 17(1):59-66.

Kaspar HF, Gillespie PA, Boyer IC, MacKenzie AL. 1985. Effects of mussel aquaculture on the nitrogen cycle and benthic communities in Kenepuru Sound, Marlborough Sounds, New Zealand. Marine Biology 85:127-36.

Kelly LA. 1992. Dissolved reactive phosphorus release from sediments beneath a freshwater cage aquaculture development in West Scotland. Hydrobiologia 235/236:569-72.

Kohler CC, Courtenay WR Jr. 1986. American Fisheries Society position on introductions of aquatic species. Fisheries 11(2):39-42.

Minchinton TE, Bertness MD. 2003. Disturbance-mediated competition and the spread of *Phragmites australis* in a coastal marsh. Ecological Applications 13(5):1400-16.

Moring J. 2005. Recent trends in anadromous fishes. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 25-42.

Newell RC, Seiderer LJ, Hitchcock DR. 1998. The impact of dredging works in coastal waters: a review of the sensitivity to disturbance and subsequent recovery of biological resources on the sea bed. In: Ansell AD, Gibson RN, Barnes M, editors. Oceanography and marine biology: an annual review. Vol. 36. Oban, Argyll (Scotland): UCL Press. p 127-78.

Newell RIE. 1988. Ecological changes in Chesapeake Bay: are they the result of overfishing the American Oyster, *Crassostrea virginica*? In: Understanding the estuary: advances in Chesapeake Bay research: proceedings of a conference; 1988 Mar 29-31; Baltimore, MD. Edgewater (MD): Chesapeake Research Consortium Publication 129. p 536-46.

[NEFMC] New England Fishery Management Council. 1998. Final Amendment #11 to the Northeast multispecies fishery management plan, Amendment #9 to the Atlantic sea scallop fishery management plan, Amendment #1 to the Monkfish fishery management plan, Amendment #1 to the Atlantic salmon fishery management plan, and components of the proposed Atlantic herring fishery management plan for essential fish habitat, incorporating the environmental assessment. Newburyport (MA): NEFMC Vol. 1.

Niimi AJ. 2004. Role of container vessels in the introduction of exotic species. Marine Pollution Bulletin 49(9-10):778-82.

Omori M, VanderSpoel S, Norman CP. 1994. Impact of human activities on pelagic biogeography. Progress in Oceanography 34(2-3):211-9.

O'Reilly JE. 1994. Nutrient loading and eutrophication. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 25-30.

Pederson J, Bullock R, Carlton J, Dijkstra J, Dobroski N, Dyrynda P, Fisher R, Harris L, Hobbs N, Lambert G, and others. 2005. Marine invaders in the Northeast: rapid assessment survey of non-native and native marine species of floating dock communities. Report of the August 3-9, 2003 Survey. Cambridge (MA): MIT Sea Grant College, Publication No. 05-3. 40 p.

Posey MH, Alphin TD, Meyer DL, Johnson JM. 2003. Benthic communities of common reed *Phragmites australis* and marsh cordgrass *Spartina alterniflora* marshes in Chesapeake Bay. Marine Ecology Progress Series 261:51-61.

Rickards WL, Ticco PC. 2002. The Suminoe oyster, *Crassostrea ariakensis*, in Chesapeake Bay: current status and near-term research activities. Charlottesville (VA): University of Virginia, Virginia Sea Grant.

Simenstad CA, Fresh KL. 1995. Influence of intertidal aquaculture on benthic communities in Pacific Northwest estuaries: scales of disturbance. Estuaries 18(1A):43-70.

Strayer DL, Hattala KA, Kahnle AW. 2004. Effects of an invasive bivalve (*Dreissena polymorpha*) on fish in the Hudson River estuary. Canadian Journal of Fisheries and Aquatic Sciences 61(6):924-41.

Tyrrell MC, Harris LG. 2000. Potential impact of the introduced Asian shore crab, *Hemigrapsus sanguineus*, in northern New England: diet, feeding preferences, and overlap with the green crab, *Carcinus maenas*. In: Pederson J, editor. Marine bioinvasions: proceedings of the 1st National Conference; 1999 Jan 24-27; Cambridge (MA): MIT SeaGrant College Program. p 208-20.

[USCOP] US Commission on Ocean Policy. 2004. An ocean blueprint for the 21st century. Final report. [Internet]. Washington (DC): USCOP. [cited 2008 Jul 24]. Available from: http://www.oceancommission.gov.

[USEPA] US Environmental Protection Agency. 2001. National coastal condition report. [Internet]. Washington (DC): US EPA Office of Research and Development/Office of Water. EPA-620/R-01/005. [cited 2008 Jul 24]. Available from: http://www.epa.gov/owow/oceans/nccr/downloads.html.

[USFWS], [NMFS] US Fish and Wildlife Service, National Marine Fisheries Service. 1999. Status review of anadromous Atlantic salmon in the United States. Hadley (MA): USFWS. 131 p.

Valentine PC, Collie JS, Reid RN, Asch RG, Guida VG, Blackwood DS. 2007. The occurrence of the colonial ascidian *Didemnum* sp. on Georges Bank gravel habitat–ecological implications and potential effects on groundfish and scallop fisheries. Journal of Experimental Marine Biology and Ecology 342(1):179-181.

Vita R, Marin A, Madrid JA, Jimenez-Brinquis B, Cesar A, Marin-Guirao L. 2004. Effects of wild fishes on waste exportation from a Mediterranean fish farm. Marine Ecology Progress Series 277:253-61.

Vitousek PM, D' Antonio CM, Loope LL, Rejmanek M, Westbrooks R. 1997. Introduced species: a significant component of human-caused global change. New Zealand Journal of Ecology 21(1):1-16.

Weinstein MP, Balletto JH. 1999. Does the common reed, *Phragmites australis*, affect essential fish habitat? Estuaries 22:793-802.

Weis JS, Weis P. 2000. Behavioral responses and interactions of estuarine animals with an invasive plant: a laboratory analysis. Biological Invasions 2:305-14.

CHAPTER ELEVEN: GLOBAL EFFECTS AND OTHER IMPACTS

Climate Change

Introduction

The earth's climate has changed throughout geological history because of a number of natural factors that affect the radiation balance of the planet, such as changes in earth's orbit, the output of the sun, and volcanic activity (IPCC 2007a). These natural changes in the earth's climate have resulted in past ice ages and periods of warming that take place over several thousand years. An example of changes to earth's climate over recent geological timeframes caused by natural factors has been observed in slowly rising global temperatures and sea levels since the end of the Pleistocene epoch (about 10,000 years before present). However, the rate of warming observed over the past 50 years is unprecedented in at least the previous 1,300 years (IPCC 2007a). The Intergovernmental Panel on Climate Change (IPCC) concludes that recent human-induced increases in atmospheric concentrations of greenhouse gases are expected to cause much more rapid changes in the earth's climate than have previously been experienced (IPCC 2007a). The buildup of greenhouse gases (primarily carbon dioxide) is a result of burning fossil fuels and forests and from certain agricultural activities. Other greenhouse gases released by human activities include nitrous oxide, methane, and chlorofluorocarbons. The global atmospheric concentration of carbon dioxide has increased from about 280 ppm during preindustrial times to 379 ppm in 2005, which far exceeds the natural range over the last 650,000 years (180-300 ppm) as determined from ice cores (IPCC 2007a).

In the Fourth Assessment Report of the IPCC, the Contribution of Working Group I issued the following conclusions (IPCC 2007a):

> Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level. Most of the observed increase in globally averaged temperatures since the mid-20th century is *very likely* due to the observed increase in anthropogenic greenhouse gas concentrations.

In order to consider various possible futures for climate change effects, the IPCC developed a series of models, or scenarios, based upon different levels of greenhouse gas emissions. The higher-emissions scenario represented fossil fuel-intensive economic growth and global human population that peaks around 2050 and then declines. This model assumes atmospheric carbon dioxide concentrations to reach about 940 ppm by 2100, or about three times preindustrial levels (Frumhoff et al. 2007). The lower-emissions scenario also represents a global human population that peaks around 2050 but assumes a much faster shift to less fossil fuel-intensive industries and more resource-efficient technologies. This model assumes carbon dioxide concentrations to peak around 2050 and then to decline to about 550 ppm by 2100, which is about double preindustrial levels (Frumhoff et al. 2007).

Based on current global climate models for greenhouse gas emission scenarios, some of the 2007 IPCC report conclusions were:

1. By 2100 average global surface air temperatures will increase by 1.8°C (lower-emissions scenario) to 4.0°C (higher-emissions scenario) above 2000 levels. The most drastic warming will occur in northern latitudes in the winter.
2. Sea level rose 12-22 cm in the 20th century and may rise another 18-38 cm (lower-emissions scenario) and as high as 26-59 cm (higher-emissions scenario) by 2099. However, these projections were based upon contributions from increased ice flow from Greenland and Antarctica at rates observed for the 1993-2003 period. If this contribution were to grow linearly with global average temperature change, the upper ranges for sea level rise would increase by an additional 10-20 cm.
3. Global precipitation is likely to increase, with more precipitation and more intense storms in the mid to high latitudes in the northern hemisphere.
4. Increasing atmospheric carbon dioxide concentrations may acidify the oceans, reducing pH levels by 0.14 and 0.35 units by 2100, adding to the present decrease of 0.1 units since preindustrial times.

The average annual atmospheric temperature across the northeastern United States has risen by approximately 0.8ºC since 1900, although this warming trend has increased to approximately 0.3ºC per decade since 1970 (Frumhoff et al. 2007). Most climate models indicate the region will experience continued increased warming over the next century (Frumhoff et al. 2007; IPCC 2007a). Climate change models predict increased warming under the lower-emissions scenario to be 2.2-4.2°C and 3.8-7.2°C under the higher-emissions scenario by 2100 in New England and eastern Canada (Frumhoff et al. 2007). Over the next several decades, the greatest temperature changes are expected to be in the wintertime and early spring with warm periods expected to increase in frequency and duration (Nedeau 2004). For example, the average winter temperature in over the next few decades are expected to increase 1.4-2.2°C under both emission scenarios, while average summer temperature increases are expected to be 0.8-1.9°C (Frumhoff et al. 2007). However, by the end of the century, the average winter temperature is expected to increase 4.4-6.7°C under the higher-emissions scenario, while summer temperature is expected to increase 3.3-7.8°C (Frumhoff et al. 2007). Long-term increases in average temperatures, the frequency and intensity of extreme temperature and climatic events, and the timing of seasonal temperature changes can have adverse effects on ecosystem function and health. Combined with extreme precipation and drought and rising sea levels, these effects have the potential to result in considerable adverse changes to the northeast region's ecosystems.

Primary impacts of global climate change that may threaten riverine, estuarine, and marine fishery resources include:

1. Increasing rates of sea-level rise and intensity and frequency of coastal storms and hurricanes will increase threats to shorelines, wetlands, and coastal ecosystems;
2. Marine and estuarine productivity will change in response to reductions in ocean pH and alterations in the timing and amount of freshwater, nutrients, and sediment delivery;
3. High water temperatures and changes in freshwater delivery will alter estuarine stratification, residence time, and eutrophication and;
4. Increased ocean temperatures are expected to cause poleward shifts in the ranges of many marine organisms, including commercial species, and these shifts may have secondary effects on their predators and prey.

These affects may be intensified by other ecosystem stresses (pollution, harvesting, habitat destruction, invasive species), leading to more significant environmental consequences. It should be noted that while the general consensus among climate scientists today indicates a current and future warming of the earth's climate caused by emissions of greenhouse gases from anthropogenic sources, the anticipated effects at regional and local levels are less understood. Consequently, there are degrees of uncertainty regarding the specific effects to marine organisms and communities and their habitats from climate change. For example, although most climate models predict an increase in extreme rainfall events in the northeast region of the United States, the regional projections for average annual precipitation and runoff vary considerably (Scavia et al. 2002).

This section attempts to address some of the possible effects of global climate change to fishery resources in the northeast region of the United States. The effects discussed in this report reflect the general topics identified by participants of the Technical Workshop on Impacts to Coastal Fishery Habitat from Nonfishing Activities. However, other possible effects and consequences of climate change have been suggested, some of which may be inconsistent with those described in this report. A complete and thorough discussion of this rapidly-developing area of science is beyond the scope of this report. For a more thorough assessment of impacts caused by climate change, we recommend the reader refer to the publications cited in this chapter, as well as new research that will emerge subsequent to this report.

Alteration of hydrological regimes

The hydrologic cycle controls the strength, timing, and volume of freshwater input, as well as the chemical and sediment load to estuaries and coastal waters (Scavia et al. 2002). Precipitation across the continental United States has increased by about 10% in the past 100 years or so, primarily reflected in the heavy and extreme daily precipitation events (Karl and Knight 1998; USGS 2005). This trend is also evident in the northeastern US region, which has experienced an increase in annual average precipitation by about 5-10% since 1900 (Frumhoff et al. 2007). In addition, increased early spring streamflows have occurred over the past century in New England, possibly a result of earlier melting of winter snowpack caused by increased air temperatures and/or greater rainfall (Hodgkins and Dudley 2005).

The IPCC Working Group II Report on Climate Change Impacts, Adaptation, and Vulnerability (IPCC 2007b) concluded that by mid-century average annual river runoff and water availability are projected to increase by 10-40% at high latitudes and in some wet tropical areas and decrease by 10-30% over some dry regions at mid-latitudes and in the dry tropics. For the northeastern United States, climate change models indicate an increase in precipitation over the next 100 years (Frumhoff et al. 2007; IPCC 2007b). By the end of the century, the average annual precipitation is expected to increase by about 10%; however, the average winter precipitation is expected to increase 20-30%, and a much greater proportion of the precipitation would be expected to fall as rain rather than snow (Frumhoff et al. 2007; IPCC 2007b). Climate models also predict more frequent, heavy-precipitation events, which are expected to increase the probability of high-flow events in Maine, New Hampshire, and Vermont streams and rivers by about 80% during late winter and spring (Frumhoff et al. 2007). These changes in the intensity and frequency of high-flow events have the potential to increase the export of nutrients, contaminants, and sediments to our estuaries. Climate-related changes in the northeast region may alter the timing and amount of water availability. For example, increased temperatures during summer months can increase evapotranspiration rates. Combined with reduced summer rainfall, these changes can cause reductions in soil moisture and streamflows that may lead to seasonal drought (Frumhoff et al. 2007).

Accelerated sea-level rise resulting from climate change threatens coastal wetlands through inundation, erosion, and saltwater intrusion (Kennedy et al. 2002; Scavia et al. 2002). The quantity of freshwater discharges affects salt marshes because river flow and runoff deliver sediments that are critical for marshes to maintain or increase its elevation. An increase in freshwater discharge could increase supply of sediment and allow coastal wetlands to cope with sea-level rise (Scavia et al. 2002). However, some coastal areas may experience a decrease in precipitation and freshwater runoff, causing salt marsh wetlands to become sediment-starved and ultimately lost as sea levels rise and marshes are drowned (Kennedy et al. 2002). Greater periods of drought leading to a decrease in freshwater discharge might also cause salinity stress in salt marshes. Rising sea levels will also allow storm surges to move further inland and expose freshwater wetlands to high salinity waters.

Estuaries may be affected by changes in precipation and freshwater discharge from rivers and runoff from land. Precipitation patterns and changes in freshwater inflow can influence water residence time, salinity, nutrient delivery, dilution, vertical stratification, and phytoplankton growth and abundance (Scavia et al. 2002). Patterns of more frequent heavy-precipitation events during winter and spring months and increased temperature and reduced rainfall during summer months may exacerbate existing nutrient over-enrichment and eutrophication conditions that already stress estuarine systems (Scavia et al. 2002; Frumhoff et al. 2007).

A decline in the atmospheric pressure at the sea surface in the central Arctic during the late 1980s led to increased delivery of warmer, higher-salinity Atlantic water into the Arctic Ocean, mainly via the Barents Sea (Greene and Pershing 2007). In addition, there has been an increase in continental melting of permafrost, snow, and ice which, combined with increased precipitation, has resulted in greater river discharge into the Arctic Ocean over the past three decades. This is believed to have led to accelerated sea ice melting and reductions in Arctic sea ice. Although the relative importance of human versus natural climate forces in driving the observed changes in atmospheric and ocean circulation patterns continues to be debated, it has led to an enhanced outflow of low-salinity waters from the Arctic and general freshening of shelf waters from the Labrador Sea to the Mid-Atlantic Bight beginning in the early 1990s (Greene and Pershing 2007). Increased freshwater input in the upper layers of the ocean results in increased stratification, which suppresses upwelling of nutrients into the upper regions of the ocean and generally reduces the productivity of phytoplankton (Kennedy et al. 2002). Conversely, increased freshwater flux and stratification could also lead to enhanced biological productivity in some systems by enabling organisms to remain longer in the photic zone (Scavia et al. 2002). Greene and Pershing (2007) reported enhanced ocean stratification caused by increased freshwater outflow from the Arctic during the 1990s. They attributed increased phytoplankton and zooplankton production and abundance during the autumn, a period when primary production would otherwise be expected to decline, with enhanced freshening of the Northwest Atlantic shelf (Greene and Pershing 2007). Although some climate models predict a net decrease in global phytoplankton productivity under doubled atmospheric carbon dioxide conditions caused by increased thermal stratification and reduced nutrient upwelling, simple extrapolation to particular northeast marine waters is difficult (Kennedy et al. 2002). The climatic variability associated with natural, large-scale phenomena such as the El Nino-Southern Oscillation and the North Atlantic Oscillation/Northen Hemisphere Annular Mode effects water column mixing and stratification on regional and global scales and has implications on the productivity of the oceans. These natural phenomena may act in tandem with, or in opposition to, anthropogenic climate change (Kennedy et al. 2002).

A number of computer climate models indicate a slowing of the “overturning” process of ocean waters, known as the thermohaline circulation (THC). This phenomenon appears to be driven by a reduction in the amount of cold and salty, and hence, more dense water sinking into the depths of the ocean. In fact, surface waters of the North Atlantic Ocean have been warming in recent decades and parts of the North Atlantic Ocean are also becoming less salty (Nedeau 2004).

In the North Atlantic, a weakening of the THC is related to wintertime warming and increased freshwater flow into the Arctic Ocean and the North Atlantic Ocean (Nedeau 2004). An increased weakening of the THC could lead to a complete shut down or southward shift of the warm Gulf Stream, as was experienced during the last glacial period (Nedeau 2004). However, the response of the THC to global climate change remains uncertain, and predictions are dependent upon future greenhouse gas emissions and temperature increases (Kennedy et al. 2002). On a regional level, changes in ocean current circulation patterns may alter temperature regimes, vertical mixing, salinity, dissolved oxygen, nutrient cycles, and larval dispersal of marine organisms in the northeast coastal region, ultimately leading to a net reduction in oceanic productivity (Nedeau 2004).

Alteration of temperature regimes

Sea surface temperatures of the northeastern US coast have increased more than 0.6°C in the past 100 years, and are projected to increase by another 3.8-4.4°C under the high-emissions scenario and by 2.2-2.8°C under the lower-emissions scenario over the next 100 years (Frumhoff et al. 2007). The IPCC Working Group II Report (IPCC 2007b) concluded there is “high confidence” that observed changes in marine and freshwater biological systems are associated with rising water temperatures, including: (1) shifts in ranges and changes in algal, plankton, and fish abundance in high-latitude oceans; (2) increased algal and zooplankton abundance in high-latitude and high-altitude lakes; and (3) range changes and earlier migrations of fish in rivers.

Temperature affects nearly every aspect of marine environments, from cellular processes to ecosystem function. The distribution, abundance, metabolism, survival, growth, reproduction, productivity, and diversity of marine organisms will all be affected by temperature changes (Kennedy et al. 2002; Nedeau 2004). Most marine organisms are able tolerate a specific temperature range and will become physiologically stressed or die after exposure to temperatures above or below the normal range. At sublethal levels, temperature extremes can effect the growth and metabolism of organisms, as well as behavior and distribution patterns. Reproduction timing and the rates of egg and larval development are dependent upon water temperatures. The reproductive success of some cold water fish species may be reduced if water temperatures rise above the optimum for larval growth (Mountain 2002). For example, cold-adapted species, such as winter flounder (*Pseudopleuronectes americanus*), Atlantic cod (*Gadus morhua*), Atlantic salmon (*Salmo salar*), and ocean quahog (*Artica islandica*) may not be able to compete with warm-adapted species if coastal water temperatures increase, particularly for those populations that may be living near the southern distribution limit (Kennedy et al. 2002).

The predicted increase in water temperatures resulting from climate change, combined with other factors such as increased precipitation and runoff, may alter seasonal stratification in the northeast coastal waters. Stratification could affect primary and secondary productivity by altering the composition of phytoplankton and zooplankton, thus affecting the growth and survival of fish larvae (Mountain 2002). In the northeast Atlantic, studies have found shifts in the timing and abundance of plankton populations with increasing ocean temperatures (Edwards and Richardson 2004; Richardson and Schoeman 2004). Edwards and Richardson (2004) found long term trends in the timing of seasonal peaks in plankton populations with increasing sea surface temperatures. However, the magnitude of the shifts in seasonal peaks were not equal among all trophic groups, suggesting alterations in the synchrony of timing between primary, secondary, and tertiary production. Richardson and Schoeman (2004) reported effects of increasing sea surface temperatures on phytoplankton abundances in the North Sea. Phytoplankton production tended to increase as cooler ocean areas warmed, probably because higher water temperatures boost phytoplankton metabolic rates. However, in warmer ocean areas phytoplankton became less abundant as sea surface temperatures increased further, possibly because warm water blocks nutrient-rich deep water from rising to the upper strata where phytoplankton exist (Richardson and Schoeman 2004). These effects have been implicated as a factor in the decline in North Sea cod stocks (Edwards and Richardson 2004; Richardson and Schoeman 2004). Impacts to the base of the food chain would not only affect fisheries but will impact entire ecosystems.

Mountain (2002) predicted a northward shift in the distributional patterns of many species of fish because of increasing water temperatures in the Mid-Atlantic region as a result of climate change. Nearly thirty years of standardized catch data on the northeast continental shelf revealed significant surface and bottom water temperature anomalies that resulted in changes to the distribution of 26 out of 30 fish species examined (Mountain and Murawski 1992). Increased water temperatures were correlated with fish moving northward or shallower to cooler water (Mountain and Murawski 1992). Perry et al. (2005) investigated the distributional patterns of demersal fish species in the North Sea and found two-thirds of all species examined shifted in latitude or depth or both in response to increasing water temperatures. This study reported that most of the species with shifting distributions had moved north or to greater depths in areas of cooler waters. Temperature induced shifts in the distribution of fish have implications for stock recruitment success and abundance. Based on the projected sea surface temperature increases under the higher-emission scenarios, Frumhoff et al. (2007) predicted bottom temperatures by the year 2100 on Georges Bank would approach the 30ºC threshold of thermally-suitable habitat and practical limit of Atlantic cod distribution. The 26ºC threshold for the growth and survival of young cod would be exceeded by the end of the century under both emission scenarios on Georges Bank (Frumhoff et al. 2007).

The frequency of diseases and pathogens may increase with warming ocean temperatures caused by climate change. For example, Dermo, a disease that affects commercially valuable oysters, exhibits higher infection rates with increased temperature and salinity. Warm, dry periods (e.g., summer drought) may make oysters more susceptible to this disease. Extremely warm waters in New England and the mid-Atlantic regions are suspected as playing a role causing disease and mortality events in American lobsters (*Homarus americanus*), including lobster-shell disease, parasitic paramoebiasis, and calcinosis (Frumhoff et al. 2007). The eelgrass wasting disease pathogen (*Labyrinthula zosterae*) has reduced eelgrass beds throughout the east coast in the past and may become more problematic because of its preference for higher salinity waters and warmer water (both of which are expected in some estuaries because of sea-level rise) (Nedeau 2004).

Changes in dissolved oxygen concentrations

Dissolved oxygen concentrations are influenced by the temperature of the water. Because warmer water holds less oxygen than does colder water, increased water temperatures will reduce the dissolved oxygen 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. Increased precipitation and freshwater runoff into estuaries would 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 in estuaries with excess nutrients (Kennedy et al. 2002; Scavia et al. 2002; Nedeau 2004). Increased vertical stratification of the water column occurs with increasing freshwater inflow and decreasing salinities, resulting from greater precipitation and storm water input. In addition, increased water temperatures in the upper strata of the water column also increase water column stratification.

Some species may be adversely affected by increasing surface water temperatures caused by climate change as they seek cooler and deeper waters. Deeper areas may be susceptible to hypoxic conditions near the bottom in stratified, poorly mixed estuarine and marine environments and would be unfavorable to many species. The habitats of aquatic species may be "squeezed" by warming surface waters and hypoxic bottom waters, resulting in greater physiologic stress and metabolic costs or death if the stress does not abate (Kennedy et al. 2002). However, an increase in coastal storm frequency and intensity, as predicted with some climate models, may contribute to some increase in vertical mixing of shallow habitats and reduce the effects of stratification.

Some phytoplankton populations may respond positively to increases in water temperatures and available carbon dioxide, which most climate models project are likely as a result of global warming (IPCC 2007a). Increased precipitation and runoff can increase the nutrient loads entering estuaries and marine waters that further exacerbate the proliferation of algae in nearshore waters. As algae die and begin to sink to the bottom, the decomposition of this increased organic material will consume more oxygen in the water, increasing the occurrence of hypoxic and anoxic conditions in coastal waters (Nedeau 2004).

Nutrient loading and eutrophication

Nitrate driven eutrophication is one of the greatest threats to the integrity of many estuaries in the northeast region (NRC 2000; Cloern 2001; Howarth et al. 2002). Increases in the amount of precipitation are very likely in northern latitudes (IPCC 2007a), and excess nutrients exported from watersheds and delivered to estuarine and marine waters may increase if freshwater flow from rivers and stormwater discharges are greater. Higher nutrient loads may increase the incidence of eutrophication and harmful algal blooms, which can cause hypoxia or anoxia in nearshore coastal waters. These effects on water quality can also negatively impact benthic communities and submerged aquatic vegetation (SAV). The environmental effects of excess nutrients or sediments are the most common and significant causes of SAV decline worldwide (Orth et al. 2006).

Release of contaminants

Increased precipitation and freshwater runoff may increase because of climate change and may lead to increased contaminant loading in coastal waters. Contaminants, such as hydrocarbons, metals, organic and inorganic chemicals, sewage, and wastewater materials, can be flushed from the watershed and exported to coastal waters, especially if the frequency and intensity of storms and floods are affected (Kennedy et al. 2002). These contaminants may be stored in coastal sediments or taken up directly by biota (e.g., bacteria, plankton, shellfish, or fish) and could ultimately affect fisheries and human health. Sea-level rise would inundate lowland sites near the coast, many of which contain hazardous substances that could leach contaminants into nearshore habitats (Bigford 1991).

Loss of wetlands and other fishery habitat

Global warming is expected to accelerate the rate of sea-level rise by expanding ocean water and melting alpine glaciers over the next century (Schneider 1998; IPCC 2007a). Average global sea levels rose 12-22 cm between 1900 and 2000 and are expected to rise another 18-38 cm (lower-emissions scenario) and as high as 26-59 cm (higher-emissions scenario) by 2100 (IPCC 2007a). In the US Atlantic coast, relative sea levels over the last century have risen approximately 18 cm in Maine and as much as 44 cm in Virginia (Zervas 2001). Sea-level rise may affect diurnal tide ranges, causing coastal erosion, increasing salinity in estuaries, and changing the water content of shoreline soils. Accelerated sea-level rise threatens coastal habitats with inundation, erosion, and saltwater intrusion (Scavia et al. 2002; Frumhoff et al. 2007). Sea-level rise may indundate salt marshes and coastal wetlands, at which point shorelines will either need to build upward (accrete) to keep pace with rising sea levels or migrate inland to keep pace with drowning/erosion on the seaward edge. In cases where the upland edge is blocked by steep topography (e.g., bluffs) or human development (e.g., shoreline protection structures) coastal wetlands including salt marsh will be lost (Scavia et al. 2002; Frumhoff et al. 2007; IPCC 2007b). Conservative estimates of losses to saline and freshwater wetlands from sea-level rise range from 47-82% of the nation's coastal wetlands, or approximately 2.3-5.7 million acres (Bigford 1991). Shoreline protection structures can also prevent the shoreward migration of SAV necessitated by sea-level rise (Orth et al. 2006).

Worldwide distribution, productivity, and function of SAV may be effected by climate change. Perhaps most critical to SAV are impacts from increases in seawater temperature resulting from the greenhouse gas effect; secondary impacts of changing water depths and tidal range caused by sea-level rise, altered current circulation patterns and current velocities; changes in salinity regimes; and potential impacts on plant photosynthesis and productivity resulting from increased ultraviolet-B radiation and carbon dioxide concentrations (Short and Neckles 1999).

The distribution and productivity of coastal wetlands may be effected by rising sea levels, altered precipitation patterns, changes in the timing and delivery of freshwater and sediment, and increases in atmospheric carbon dioxide and temperature (Scavia et al. 2002). Increased atmospheric carbon dioxide could increase plant production for some coastal wetland species, assuming other factors such as nutrients and precipitation are not limiting. However, rising sea levels may inhibit the growth of some brackish and freshwater marshes and swamps.

Shoreline erosion

Millions of cubic yards of sand are placed on northeast coastal beaches each year by state and federal governments to combat shoreline erosion. In addition, a variety of hard structures such as seawalls, revetments, groins, and jetties have been installed to protect eroding shorelines. Yet some areas of the northeast, such as Cape Cod, MA, Long Island, NY/CT, and coastal New Jersey, continue to experience a net loss of shoreline and have been identified by the US Geological Survey as being particularly at risk from sea-level rise (Frumhoff et al. 2007). It is uncertain how these engineering measures might affect the ability of natural processes to respond to future sea-level rise (Gutierrez et al. 2007). There exists a high degree of uncertainty in predicting long-term shoreline changes because of the uncertainty of the rate of future sea-level rise and the complex interactions of regional sediment budgets, coastal geomorphology, and anthropogenic influences, such as beach nourishment and seawall construction. However, Gutierrez et al. (2007) reported an increased likelihood for erosion and shoreline retreat for all types of mid-Atlantic coastal shorelines, including an increased likelihood for overwash and inlet breaching and the possibility of segmentation or disintegration of some barrier islands.

An increase in freshwater discharge, storm frequency and intensity, and sea-level rise can lead to increased erosion rates along coastal shorelines (Scavia et al. 2002). The loss of riparian and salt marsh vegetation because of climate change effects could serve as a feedback loop that reduces the ability of wetlands to withstand further increases in sea level and storm effects, which may exacerbate the effects of coastal erosion.

Alteration of salinity regimes

Vertical mixing in coastal waters is influenced by several factors, including water temperatures and freshwater input, so warmer temperatures may affect the thermal stratification of estuaries (Nedeau 2004). Climate models project increased average temperatures and precipitation, particularly during the winter, in the northeastern US region (Frumhoff et al. 2007). Hotter and drier summers and warmer, wetter winters will alter the timing and volume of freshwater runoff and river flows. If freshwater flow from rivers is reduced or increased, salinities in rivers and estuaries will be altered which will have profound affects on the distribution and life history requirements of coastal fisheries. For example, increased freshwater input into estuaries would lower salinities in salt marsh habitat which could enhance conditions for invasive exotic plants that prefer low-salinity conditions, such as *Phragmites* or purple loosestrife (*Lythrum salicaria*). Increased freshwater runoff will increase vertical stratification of estuaries and coastal waters, which could have indirect effects on estuarine and coastal ecosystems (Kennedy et al. 2002). For example, upwelling of deep, nutrient-rich seawater could be reduced, leading to reductions in primary productivity in coastal waters. Rising sea levels could cause estuarine wetlands to be inundated with higher salinity seawater, altering the ecological balance of highly productive fishery habitat.

Alteration of weather patterns

Numerous long-term changes in climate have already been observed at continental, regional, and ocean basin scales, including changes in Arctic temperatures, ice, ocean salinity, wind patterns; and increased occurrences of extreme weather events including droughts, heavy precipitation, heat waves, and intensity of tropical cyclones (IPCC 2007a).

There is observational evidence for an increase in intense tropical cyclone activity in the North Atlantic since the 1970s, correlated with increased tropical sea-surface temperatures (IPCC 2007a). Increases in the amount of precipitation are very likely in high latitudes, and extra-tropical storms are projected to move poleward (Frumhoff et al. 2007; IPCC 2007a). Although there continues to be debate over the link between global warming and increased hurricane frequency, observed ocean warming is a key condition for the formation and strengthening of hurricanes (Frumhoff et al. 2007). The integrity of shorelines and wetlands would be threatened by increased intensity and frequency of coastal storms and hurricanes resulting from climate change. The loss of coastal wetland vegetation and increased erosion of shorelines and riparian habitats caused by storms would have an adverse effect on the integrity of aquatic habitats. Reductions in dissolved oxygen concentrations and salinity are phenomena associated with coastal storms and hurricanes, and most aquatic systems require weeks or months to recover following severe storms (Van Dolah and Anderson 1991). Increased frequency and intensity of storms could lead to chronic disturbances and have adverse consequences on the health and ecology of coastal rivers and estuaries.

Changes in water alkalinity

Increasing atmospheric carbon dioxide concentrations can alter seawater carbonate chemistry by lowering seawater pH, carbonate ion concentration, and carbonate saturation state and by increasing dissolved carbon dioxide concentration (Riebesell 2004). According to the IPCC Working Group I Fouth Asssessment, increasing atmospheric carbon dioxide concentrations may acidify the oceans, reducing pH levels by 0.14 and 0.35 units by 2100 (IPCC 2007a). The uptake of anthropogenic carbon since 1750 has led to an average decrease in pH of 0.1 units; however, the effects of observed ocean acidification on marine ecosystems are unclear at this time (IPCC 2007b).

Increased acidity in oceans is expected to effect calcium carbonate availability in seawater, which would lower the calcification rates in marine organisms (e.g., mollusks and crustaceans, some plankton, hard corals) (IPCC 2007b). Alteration of water alkalinity could have severe impacts on primary and secondary production, which have implications at the ecosystem level (Orr et al. 2005). Increasing atmospheric carbon dioxide concentrations and altered seawater carbonate chemistry could have a range of effects, including physiological changes to marine plankton on the organismal level, changes in ecosystem structure and regulation, and large scale shifts in biogeochemical cycling (Riebesell 2004). For example, increased carbon dioxide concentrations are predicted to decrease the carbonate saturation state and cause a reduction in biogenic calcification of corals and some plankton, including coccolithophorids and foraminifera; however, increasing carbon dioxide concentrations could increase the rates of photosynthetic carbon fixation of some calcifying phytoplankton (Riebesell 2004).

Changes in community and ecosystem structure

The geographic distributions of species may expand, contract, or otherwise adjust to changing oceanic temperatures, creating new combinations of species that could interact in unpredictable ways. Fish communities are likely to change. For example, warming oceans may cause the southern range of northern species, such as Atlantic cod, American plaice (*Hippoglossoides platessoides*), haddock (*Melanogrammus aeglefinus*), and Atlantic halibut (*Hippoglossus hippoglossus*), to shift north as will the northern range limit of southern species, such as butterfish (*Peprilus triacanthus*) and menhaden (*Brevoortia tyrannus*) (Nedeau 2004; Frumhoff et al. 2007). Mountain and Murawski (1992) reported changes in the distribution of selected fish stocks in the northeast continental shelf that were attributed to changes in surface and bottom water temperatures. Distributional changes attributed to increased water temperatures were observed in 26 out of the 30 species examined and resulted in fish moving northward or shallower towards cooler water (Mountain and Murawski 1992). Temperature induced shifts in the distribution of fish have implications for stock recruitment success and abundance. Short-lived fish species may show the most rapid demographic responses to temperature changes, resulting in stronger distributional responses to warming (Perry et al. 2005). Range shifts could create new competitive interactions between species that had not evolved in sympatry, causing further losses of competitively inferior or poorly adapted species.

Because of changes in the atmospheric and oceanic circulation patterns in the Arctic Ocean, the Northwest Atlantic shelf waters became fresher during the 1990s relative to the 1980s (Greene and Pershing 2007). This freshening was believed to have enhanced stratification of shelf waters and led to greater phytoplankton and zooplankton production and abundance during the autumn, a period when primary production would otherwise be expected to decline (Greene and Pershing 2007). Although it is uncertain as to whether the increased abundances of plankton during the 1990s were solely attributed to enhanced stratification caused by greater inflow of freshwater (bottom-up control), overfishing of large predators, such as Atlantic cod (top-down control) or some combined effect, it is clear that changes in climate and oceanic circulation patterns can have profound effects on ecosystem functions and productivity (Greene and Pershing 2007). Mountain (2002) proposed several possible effects to fish stocks in the mid-Atlantic region in response to increased water temperatures, increased seasonal stratification of the water column, and changes in regional ocean circulation patterns. Direct effects included northward shift in stock distributions and reduced reproductive success for some cold water species because of increased water temperatures; indirect effects included changes in phytoplankton productivity and species composition that can impact the lower trophic levels affecting recruitment success of fish stocks (Mountain 2002).

Migratory and anadromous fish such as salmon and shad may be affected by climate change because they depend on the timing of seasonal temperature-related events as cues for migration. Ideal river and ocean temperatures may be out of synch as climate changes, making the saltwater-to-freshwater transition difficult for spawning adults or the freshwater-to-saltwater transition difficult for ocean-bound juveniles. Migration routes, timing of migration, and ocean growth and survival of fish may also be affected by altered sea-surface temperatures (Nedeau 2004).

Invasive species may flourish in a changing climate when shifting environmental conditions give certain species a foothold in a community and a competitive advantage over native species. Species inhabiting northern latitude islands may be particularly vulnerable as nonnative organisms adapted to warmer climates take advantage of changing climatic conditions (Scavia et al. 2002; IPCC 2007b).

Increases in the severity and frequency of coastal storms may result in cumulative losses of coastal marshes by eroding the seaward edge, causing flooding further inland, changing salinity regimes and marsh hydrology, and causing vegetation patterns to change. Healthy salt marshes can buffer upland areas (including human structures) from storm damage, and this ecosystem function will be impaired if marshes are destroyed or degraded. Increased sea-surface temperatures, sea-level rise, and intensity of storms and associated surge and swells, combined with more localized effects such as nutrients and increased loading of sediments, have had demonstrable impacts on SAV beds worldwide (Orth et al. 2006). The loss or degradation of freshwater, brackish, and salt marsh wetlands, SAV and shellfish beds, and other coastal habitats will affect critical habitat for many species of wildlife, which may ultimately affect biodiversity, coastal ecosystem productivity, fisheries, and water quality.

Changes in ocean/coastal uses

Commercial fisheries could be impacted by the cumulative effects of climate change, including rising sea levels and water temperatures and habitat degradation in estuaries, rivers, and coastal wetlands. Approximately 32% of species important to fisheries in New England are dependent upon estuaries during some portion of their life histories (Nedeau 2004). Climate change could also affect human health and the use of ocean resources if the frequency and intensity of harmful algal blooms, fish and shellfish diseases, coastal storms, and impacts to coastal wetlands increase. These effects, combined with sea-level rise, may result in a loss or inability to utilize coastal resources. Climate-induced changes to marine ecosystems will require consideration of longer time-scale effects in fisheries and coastal management strategies.

The IPCC Working Group II Report (IPCC 2007b) concluded there is “high confidence” that climate change will cause regional changes in the distribution and production of particular fish species, with adverse effects projected for aquaculture and fisheries. Conservative predictions of impacts to fisheries resources from sea-level rise and habitat loss from climate change would likely dwarf those impacts now attributed to direct human activities, like water quality degradation, coastal development, and dredging (Bigford 1991). It is possible that nonclimate stresses will increase the vulnerability to climate change impacts by reducing resilience and adaptive capacity (IPCC 2007b). However, it is likely that sustainable development, along with implementing strategies of climate change mitigation and adaptation, technological development (to enhance adaptation and mitigation), and research (on climate science, impacts, adaptation, and mitigation) can minimize some of the risks associated with climate change (IPCC 2007b).

The development of strategic mitigation and adaptation measures to address global climate change are beyond the scope of this report. However, conservation measures and best management practices that are consistent with sound coastal management and sustainable development may help mitigate some of the effects of global warming.

Conservation measures and best management practices for climate change impacts to aquatic habitat

1. Promote soft shore protection techniques, such as salt marsh restoration and creation and beach dune restoration, as alternatives to hard-armoring approaches.
2. Consider vertical structures such as concrete bulkheads for shoreline stabilization only as a last resort.
3. Establish setback lines for coastal development and rolling easements based on sea-level rise and subsidence projections that include local land movement.
4. Avoid development projects that involve wetland filling and increase impervious surfaces.
5. Improve land use practices, such as more efficient nutrient management and more extensive restoration and protection of riparian zones and wetlands.
6. Encourage the development and use of renewable, nongreenhouse gas emitting energy technologies, whenever practicable and feasible.
7. Encourage local, regional, and federal agencies to consider implications of climate change in their decision-support analysis and documents (e.g., National Environmental Policy Act) regarding permit decisions and funding programs.
8. Encourage the use of energy efficient technologies to be integrated into commercial and residential construction, including renewable energy and energy efficient heating and cooling systems and insulation.
9. Encourage the use of fuel-efficient vehicles and mass transportation systems.
10. Encourage communities and states to develop and implement strategies for sustainable development and greenhouse gas reduction initiatives, such as through the International Council for Local Environmental Initiatives (ICLEI).

Ocean Noise

Introduction

Sound is the result of energy created by a mechanical action dispersed from a source at a particular velocity and causes two types of actions: an oscillation of pressure in the surrounding environment and an oscillation of particles in the medium (Stocker 2002). Because water is 3500 times denser than air, sound travels five times faster in water (Stocker 2002). The openness of the ocean and relative density of the ocean medium allow for the transmission of sound energy over long distances. Factors that affect density include temperature, salinity, and pressure. These factors are relatively predictable in the open ocean but highly variable in coastal and estuarine waters. As a result of these factors along with water depth and variable nearshore bathymetry, sound attenuates more rapidly with distance in shallow compared to deep water (Rogers and Cox 1988).

Noise in the ocean environment can be categorized as natural and anthropogenic sources. Naturally generated sounds come from wind, waves, ice, seismic activity, tides and currents, and thunder, among other sources. Many sea animals use sound in a variety of ways; some use sound passively and others actively. Passive use of sound occurs when the animal does not create the sound that it senses but responds to environmental and ambient sounds. These uses include detection of predators, location and detection of prey, proximity perception of cospecies in schools or colonies, navigation, and perception of changing environmental conditions such as seismic movement, tides, and currents. Animals also create sounds to interact with their environment or other animals in it. Such active uses include sonic communication with cospecies for feeding and spawning (e.g., oyster toadfish [*Opsanus tau*]), territorial and social interactions, echolocation (e.g., marine mammals), stunning and apprehending prey, long distance navigation and mapping (e.g., sharks and marine mammals), and the use of sound as a defense against predators (e.g., croakers) (Stocker 2002).

The degree to which an individual fish exposed to noise will be affected is dependent upon a number of variables, including: (1) species of fish; (2) fish size; (3) presence of a swimbladder; (4) physical condition of the fish; (5) peak sound pressure and frequency; (6) shape of the sound wave (rise time); (7) depth of the water; (8) depth of the fish in the water column; (9) amount of air in the water; (10) size and number of waves on the water surface; (11) bottom substrate composition and texture; (12) tidal currents; and (13) presence of predators (Hanson et al. 2003).

Anthropogenic sources of noise include commercial shipping, seismic exploration, sonar, acoustic deterrent devices, and industrial activities and construction. The ambient noises in an average shipping channel are a combination of propeller, engine, hull, and navigation noises. In coastal areas the sounds of cargo and tanker traffic are multiplied by complex reflected paths – scattering and reverberating because of littoral geography. These cargo vessels are also accompanied by all other manner of vessels and watercraft: commercial and private fishing boats, pleasure craft, personal watercraft (e.g., jet skis) as well as coastal industrial vessels, public transport ferries, and shipping safety and security services such as tugs boats, pilot boats, US Coast Guard and coastal agency support craft, and of course all varieties of US Navy ships – from submarines to aircraft carriers. In large part, anthropogenic activities creating ocean noise are concentrated in coastal and nearshore areas. The most pervasive anthropogenic ocean noise is caused by transoceanic shipping traffic (Stocker 2002). The average shipping channel noise levels are 70-90 dB, which is as much as 45 dB over the natural ocean ambient noise in surface regions (Stocker 2002). Ships generate noise primarily by propeller action, propulsion machinery, and hydraulic flow over the hull (Hildebrand 2004). Considering all of these noises together, noise generated from a large container vessel can exceed 190 dB at the source (Jasny et al. 1999). Refer to the Marine Transportation chapter for additional information on ocean noises generated from vessels.

The loudest noises may be the sounds of marine extraction industries such as oil drilling and mineral mining (Stocker 2002). The most prevalent sources of these sounds are from "air guns" used to create and read seismic disturbances. Air guns are used in seismic exploration to create a sound pressure wave that aids in reflection profiling of underlying substrates for oil and gas. These devices generate and direct huge impact noises into the ocean substrate. Offshore oil and gas exploration generally occurs along the continental margins; however, a recent study indicated that air gun activity in these areas propagates into the deep ocean and is a significant component of low frequency noise (Hildebrand 2004). Peak source levels of air guns typically are 250-255 dB. Following the exploration stage, drilling, coring, and dredging are performed during extraction which also generates loud noises. Acoustic telemetry is also associated with positioning, locating, equipment steering, and remotely operated vessel control to support extraction operations (Stocker 2002).

Sonar systems are used for a wide variety of civilian and military operations. Active sonar systems send acoustic energy into the water column and receive reflected and scattered energy. Sonar systems can be classified into low (<1 kHz), mid (1-20 kHz), and high frequency (>20 kHz). Most vessels have sonar systems for navigation, depth sounding, and "fish finding." Some commercial fishing boats also deploy various acoustic aversion devices to keep dolphins, seals, and turtles from running afoul of the nets (Stocker 2002).

Because the ocean transfers sound over long distances so effectively, various technologies have been designed to make use of this feature (e.g., long distance communication, mapping, and surveillance). Since the early 1990s, it has been known that extremely loud sounds could be transmitted in the deep-ocean isotherm and could be coherently received throughout the seas. Early research in the use of deep-ocean noise was conducted to map and monitor deep-ocean water temperature regimes. Since the speed of sound in water is dependent on temperature, this characteristic was used to measure the temperature of the deep water throughout the sea. This technology has been used to study long-term trends in deep-ocean water temperature that could give a reliable confirmation of global warming. This program, Acoustic Thermometry of Ocean Climates (ATOC), uses receivers stationed throughout the Pacific Basin from the Aleutian Islands to Australia. ATOC is a long wavelength, low frequency sound in the 1-500 Hz band and is the first pervasive deep-water sound channel transmission, filling an acoustical niche previously only occupied by deep sounding whales and other deep water creatures (Stocker 2002). Concurrent with the development of ATOC, the US Navy and other North American Treaty Organization (NATO) navies have developed other low frequency communications and surveillance systems. Most notable of these is low frequency active sonar (LFAS) on a mobile platform, or towed array (Stocker 2002). Recently, the use of LFAS for military purposes has received considerable attention and controversy because of the concerns that this technology has resulted in injury and death to marine mammals, particularly threatened and endangered whales. Fernandez et al. (2005) found the occurrence of mass stranding events of beaked whales in the Canary Islands to have a temporal and spatial coincidence with military exercises using mid-frequency sonar. Beaked whales that died after stranding were found to have injuries to tissues consistent with acute decompression-like illness in humans and laboratory animals. Additional monitoring and research will need to be conducted to determine the degree of threat sonar has on marine organisms, particularly marine mammals. The effects of LFAS on bony fish and elasmobranches are unknown at this time.

Industrial and construction activities concentrated in nearshore areas contribute to ocean noise. Primary activities include pile driving, dredging, and resource extraction and production activities. Pile driving activities, which typically occur at frequencies below 1000 Hz, have led to mortality in fish (Hastings and Popper 2005). Intensity levels of pile driving have been measured up to 193 dB in certain studies (Hastings and Popper 2005). Refer to the chapter on Coastal Development for additional information on the affects of pile driving.

Underwater blasting with explosives is used for a number of development activities in coastal waters. Blasting is typically used for dredging new navigation channels in areas containing large boulders and ledges; decommissioning and removing bridge structures and dams; and construction of new in-water structures such as gas and oil pipelines, bridges, and dams. The potential for injury and mortality to fish from underwater explosives has been well-documented (Hubbs and Rechnitzer 1952; Teleki and Chamberlain 1978; Linton et al. 1985; and Keevin et al. 1999). Generally, aquatic organisms that possess air cavities (e.g., lungs, swim bladders) are more susceptible to underwater blasts than are those without. In addition, smaller fish are more likely to be impacted by the shock wave of underwater blasts than are larger fish, and the eggs and embryos tend to be particularly sensitive (Wright 1982). However, fish larvae tend to be less sensitive to blasts than are eggs or post-larval fish, probably because the larval stages do not yet possess air bladders (Wright 1982). Impacts to fishery habitat from underwater explosives may include sedimentation and turbidity in the water column and benthos and the release of contaminants (e.g., ammonia) in the water column with the use of certain types of explosives.

Noise generated from anthropogenic sources covers the full frequency of bandwidth used by marine animals (0.001-200 kHz), and most audiograms of fishes indicate a higher sensitivity to sound within the 0.100-2 kHz range (Stocker 2002). Evidence indicates that fish as a group have very complex and diverse relationships with sound and how they perceive it. It should be noted that relatively little direct research has been conducted on the impacts of noise to marine fish. However, some studies and formal observations have been conducted that elucidate general categories of impacts to fish species. Noise impacts to fish can generally be divided into four categories: (1) physiological; (2) acoustic; (3) behavioral; and (4) cumulative.

Physiological impacts to fish

Increased pressure from high noise levels may have impacts on other nonauditory biological structures such as swim bladders, the brain, eyes, and vascular systems (Hastings and Popper 2005). Any organ that reflects a pressure differential between internal and external conditions may be susceptible to pressure-related impacts. Some of the resulting affects on fish include a rupturing of organs and mortality (Hastings and Popper 2005). Sounds within autonomic response ranges of various organisms may trigger physiological responses that are not environmentally adapted in healthful ways (Stocker 2002).

The lethality of underwater blasts on fish is dependent upon the detonation velocity of the explosion; however, a number of other variables may play an important role, including the size, shape, species, and orientation of the organism to the shock wave, and the amount, type of explosive, detonation depth, water depth, and bottom type (Linton et al. 1985). Fish with swimbladders are the most susceptible to underwater blasts, owing to the effects of rapid changes in hydrostatic pressures on this gas-filled organ. The kidney, liver, spleen, and sinus venosus are other organs that are typically injured after underwater blasts (Linton et al. 1985).

Acoustic impacts to fish

Acoustic impacts include damage to auditory tissue that can lead to hearing loss or threshold shifts in hearing (Jasny et al. 1999; Heathershaw et al. 2001; Hastings and Popper 2005). Temporary threshold shifts and permanent threshold shifts may result from exposure to low levels of sound for a relatively long period of time or exposure to high levels of sound for shorter periods. Threshold shifts can impact a fish's ability to carry out its life functions.

Behavioral impacts to fish

While tissue damage would be a significant factor in compromising the health of fish, other effects of anthropogenic noise are more pervasive and potentially more damaging. For example, masking biologically significant sounds by anthropogenic interference could compromise acoustical interactions from feeding to breeding, to community bonding, to schooling synchronization, and all of the more subtle communications between these behaviors. Anthropogenic sounds that falsely trigger these responses may have animals expend energy without benefits (Stocker 2002). With respect to behavioral impacts on fish, studies in this area have been limited. Clupeid fish, including Atlantic herring (*Clupea harengus*) are extremely sensitive to noise, and schools have been shown to disperse when approached by fishing gear, such as trawls and seines (NOAA Fisheries 2005). Several studies indicate that catch rates of fish have decreased in areas exposed to seismic air gun blasts (Engås et al. 1996; Hastings and Popper 2005). These results imply that fish relocate to areas beyond the impact zone. One study indicated that catch rates increased 30-50 km away from the noise source (Hastings and Popper 2005). Several studies have indicated that increased background noise and sudden increases in sound pressure can lead to elevated levels of stress in many fish species (Hastings and Popper 2005). Elevated stress levels can increase a fish's vulnerability to predation and other environmental impacts. New studies are addressing the masking effects by background noise on the ability of fish to understand their surroundings. Because fish apparently rely so heavily on auditory cues to develop an "auditory scene," an increase in ambient background noise can potentially reduce a fish's ability to receive those cues and respond appropriately (Jasny et al. 1999; Scholik and Yan 2002; Hastings and Popper 2005). Furthermore, the auditory threshold shifts of fish exposed to noise may not recover even after termination of the noise exposure (Scholik and Yan 2002).

Cumulative impacts to fish

Few research efforts have focused on the cumulative effects of anthropogenic ocean noise on fish. Subtle and long-term effects on behavior or physiology could result from persistent exposure to certain noise levels leading to an impact on the survival of fish populations (Jasny et al. 1999; Hastings and Popper 2005).

Conservation measures and best management practices for ocean noise

1. Develop mitigation strategies for noise impacts to consider the frequency, intensity, and duration of exposure and evaluate possible reductions of each of these three factors. Mitigation strategies for ocean noise are challenged by the fact that a sound source may move in addition to the movement of affected fish in and out of the insonified region.
2. Assess the "acoustic footprint" of a given sound source and develop standoff ranges for various impact levels. Standoff ranges can be calculated by using damage risk criteria for species exposure, source levels, sound propagation conditions, and acoustic attenuation models. Development of standoff ranges implies that sound sources be relocated or reduced since the sound receptors (fish) are more difficult to control. Because the potential number of species affected and their location is most likely unknown, development of a generic approach for mitigation by using the species with the most sensitive hearing would produce a precautionary approach to reducing impacts on all animals (Heathershaw et al. 2001).
3. Recommend an assessment and designation of "acoustic hotspots" that are particularly susceptible to acoustic impacts and reducing sound sources around them. These hotspots may include seasonal areas for particularly susceptible life history activities like spawning or breeding (Jasny et al. 1999).
4. Recognize that reducing noise intensity at the source primarily relies on technological solutions. These options include the use of "quiet" technology in marine engines and using bubble curtains for activities such as pile driving.
5. Encourage the use of sound dampening technologies for vessels and port/marine infrastructure to reduce ocean noise impacts to aquatic organisms.
6. Manage the duration of sound when the source level of a sound cannot be reduced in order to reduce impacts. Underwater sounds should be avoided during sensitive times of year (e.g., upstream and downstream river migrations, spawning, and egg and larvae development).
7. Avoid using underwater explosives in areas supporting productive fishery habitats. The use of less destructive methods should be encouraged, whenever possible. In some cases, the use of mechanical devices (e.g., ram hoe, clamshell dredge) may reduce impacts associated with rock and ledge removal.
8. Investigate options to mitigate the impacts associated with underwater explosives. Avoiding use during sensitive periods (e.g., upstream and downstream river migrations, spawning, and egg and larvae development) may be one of the most effective means of minimizing impacts to fishery resources. Other methods may include the use of bubble curtains; stemming (back-filling charge holes with gravel); delayed charges (explosive charges broken down into a series of smaller charges); and the use of repelling charges (small explosive charges used to frighten and drive fish away from the blasting zone) (Keevin 1998).

Atmospheric Deposition

Introduction

Pollutants travel through the atmosphere for distances of up to thousands of miles, often times to be deposited into rivers, estuaries, and nearshore and offshore marine environments. Substances such as sulfur dioxide, nitrogen oxide, carbon monoxide, lead, volatile organic compounds, particulate matter, and other pollutants are returned to the earth through either wet or dry atmospheric deposition. Wet deposition removes gases and particles in the atmosphere and deposits them to the earth's surface by means of rain, sleet, snow, and fog. Dry deposition is the process through which particles and gases are deposited in the absence of precipitation. Deposition of nutrients (i.e., nitrogen and phosphorous) and contaminants (e.g., polychlorinated biphenyl [PCB] and mercury) into the aquatic system are of particular concern because of the resulting impacts to fisheries and health-risks to humans.

Atmospheric inputs of nutrients and contaminants differ from riverine inputs in the following ways: (1) riverine inputs are delivered to the coastal seas at their margins, whereas atmospheric inputs can be delivered directly to the surface of the central areas of coastal seas and hence exert an impact in regions less directly affected by riverine inputs; (2) atmospheric delivery occurs at all times, whereas riverine inputs are dominated by seasonal high-flows and coastal phytoplankton activity; (3) atmospheric inputs are capable of episodic, high deposition events associated with natural or manmade phenomena (e.g., volcanic eruptions, forest fires); and (4) atmospheric inputs of nitrogen are chemically different from river inputs in that rivers are dominated by nitrous oxides, phosphorus, and silica, while atmospheric inputs include reduced and oxidized nitrogen, but no significant phosphorus or silica (Jickells 1998). While there is little information on the direct effects of atmospheric deposition on marine ecosystems, management strategies must attempt to address these variations in inputs from terrestrial and atmospheric pathways.

Nutrient loading and eutrophication

Nutrient pollution is currently the largest pollution problem in the coastal rivers and bays of the United States (NRC 2000). Nitrogen inputs to estuaries on the Atlantic and Gulf Coasts of the United States are now 2-20 times greater than during preindustrialized times (Castro et al. 2003). Sources of nitrogen include emissions from automobiles, as well as urban, industrial, and agricultural sources. Atmospheric deposition is one means of nitrogen input into aquatic systems, with atmospheric inputs delivering 20 to greater than 50% of the total input of nitrogen oxide to coastal waters (Paerl 1995). One of the most rapidly increasing means of nutrient loading to both freshwater systems and the coastal zone is via atmospheric pathways (Anderson et al. 2002).

Precipitation readily removes most reactive nitrogen compounds, such as ammonia and nitrogen oxides, from the atmosphere. These compounds are subsequently available as nutrients to aquatic and terrestrial ecosystems. Because nitrogen is commonly a growth-limiting nutrient in streams, lakes, and coastal waters, increased concentrations can lead to eutrophication, a process involving excess algae production, followed by depletion of oxygen in bottom waters. Hypoxic and anoxic conditions are created as algae die off and decompose. Harmful algal blooms associated with unnatural nutrient levels have been known to stimulate fish disease and kills. In addition, phytoplankton production increases the turbidity of waters and may result in a reduced photic zone and subsequent loss of submerged aquatic vegetation. Anoxic conditions, increased turbidity, and fish mortality may result from increased nitrogen inputs into the aquatic system, potentially altering long-term community dynamics (NRC 2000; Castro et al. 2003). Refer to the chapters on

Agriculture and Silviculture, Coastal Development, Alteration of Freshwater Systems, and Chemical Effects: Water Discharge Facilities for further discussion on impacts to fisheries from eutrophication.

The atmospheric component of nitrogen flux into estuaries has often been underestimated, particularly with respect to deposition on the terrestrial landscape with subsequent export downstream to estuaries and coastal waters (Howarth et al. 2002). The deposition of nitrogen on land via atmospheric pathways impacts aquatic systems when terrestrial ecosystems become nitrogen saturated. Nitrogen saturation means that the inputs of nitrogen into the soil exceed the uptake ability by plants and soil microorganisms. Under conditions of nitrogen saturation, excess nitrogen leaches into soil water and subsequently into ground and surface waters. This leaching of excess nitrogen from the soils degrades water quality. Such conditions have been known to occur in some forested watersheds in the northeastern United States, and streams that drain these watersheds have shown increased levels of nitrogen from runoff (Williams et al. 1996).

In one study, quantifying nitrogen inputs for 34 estuaries on the Atlantic and Gulf Coasts of the United States, atmospheric deposition was the dominant nitrogen source for three estuaries, and six estuaries had atmospheric contributions greater than 30% of the total nitrogen inputs (Castro et al. 2003). In the northeastern United States, atmospheric deposition of oxidized nitrogen from fossil-fuel combustion may be the major source of nonpoint input. Evidence suggests a significant movement of nitrogen in the atmosphere from the eastern United States to coastal and offshore waters of the North Atlantic Ocean where it is deposited (Holland et al. 1999). Nitrogen fluxes in many rivers in the northeastern United States have increased 2- to 3-fold or more since 1960, with much of this increase occurring between 1965 and 1988. Most of this increase in nitrogen was attributed to increased atmospheric deposition originating from fossil-fuel combustion onto the landscape (Jaworski et al. 1997).

Mercury loading/bioaccumulation

Mercury is a hazardous environmental contaminant. Mercury bioaccumulates in the environment, which means it can collect in the tissues of a plant or animal over its lifetime and biomagnify (i.e., increases in concentration within organisms between successive trophic levels) within the food chain. Fish near the top of the food chain often contain high levels of mercury, prompting the United States and Canada to issue health advisories against consumption of certain fish species. The US Food and Drug Administration reports certain species, including sharks, swordfish (*Xiphias gladius*), king mackerel (*Scombermorus cavalla*), and tilefish (*Lopholatilus chamaeleonticeps*), to have typically high concentrations of mercury (USFDA 2004).

One of the most important anthropogenic sources of mercury pollution in aquatic systems is atmospheric deposition (Wang et al. 2004). The amount of mercury emitted into the atmosphere through natural and reemitted sources was estimated to be between 1500-2500 metric tons/year in the late 20th century (Nriagu 1990). Industrial activities have increased atmospheric mercury levels, with modern deposition flux estimated to be 3-24 times higher than preindustrial flux (Bindler 2003). More than half of the total global mercury emissions are from incineration of solid waste, municipal and medical wastes, and combustion of coal and oil (Pirrone et al. 1996).

Studies strongly support the theory that atmospheric deposition is an important (sometimes even the predominant) source of mercury contamination in aquatic systems (Wang et al. 2004). Mercury exists in the atmosphere predominately in the gaseous form, although particulate and aqueous forms also exist (Schroeder et al. 1991). Gaseous mercury is highly volatile, remaining in the atmosphere for more than one year, making long-range atmospheric transport a major environmental concern (Wang et al. 2004).

Concentrations of mercury in the atmosphere and flux of mercury deposition vary with the seasons, and studies suggest that atmospheric mercury deposition is greatest in summer and least in winter (Mason et al. 2000). Different, site-specific factors may influence the transport and transformation of mercury in the atmosphere. Wind influences the direction and distance of deposition from the source, while high moisture content may increase the oxidation of mercury, resulting in the rapid settlement of mercury into terrestrial or aquatic systems. Mercury that is deposited on land can be absorbed by plants through their foliage and ultimately be passed into watersheds by litterfall (Wang et al. 2004).

Mercury and other metal contaminants are found in the water column and persist in sediments (Buchholtz ten Brink et al. 1996). Mercury is toxic in any form according to some scientists, but when absorbed by certain bacteria such as those in marine sediments, it is converted to its most toxic form, methyl mercury. Methyl mercury can cause nerve and developmental damage in humans and animals. Mercury inhibits reproduction and development of aquatic organisms, with the early life-history stages of fish being the most susceptible to the toxic impacts associated with metals (Gould et al. 1994). Metals have also been implicated in disrupting endocrine secretions of aquatic organisms, potentially disrupting natural biotic properties (Brodeur et al. 1997). Direct mortality of fish and invertebrates by lethal concentrations of metals may occur in some instances. Refer to the Coastal Development and Chemical Effects: Water Discharge Facilities chapters for more information on impacts from mercury contamination.

PCB and other contaminants

PCB congeners are a group of organic chemicals which can be odorless or mildly aromatic and exist in solid or oily-liquid form. They were formerly used in the United States as hydraulic fluids, plasticizers, adhesives, fire retardants, way extenders, dedusting agents, pesticide extenders, inks, lubricants, cutting oils, manufacturing of heat transfer systems, and carbonless reproducing paper. Most uses of PCB were banned by the US Environmental Protection Agency in 1979; however this persistent contaminant continues to enter the atmosphere mainly by cycling from soil to air to soil again. PCB is also currently released from landfills, incineration of municipal refuse and sewage sludge, and improper (or illegal) disposal of PCB-contaminated materials, such as waste transformer fluid, to open areas (USEPA 2005a).

PCB compounds are a mixture of different congeners of chlorobiphenyl. In general, the persistence of PCB increases with an increase in the degree of chlorination. Mono-, di- and trichlorinated biphenyls biodegrade relatively rapidly, tetrachlorinated biphenyls biodegrade slowly, and higher chlorinated biphenyls are resistant to biodegradation. If released to the atmosphere, PCB will primarily exist in the vapor-phase and have a tendency to become associated with the particulate-phase as the degree of chlorination of the PCB increases. Physical removal of PCB from the atmosphere is accomplished by wet and dry deposition (USEPA 2005b).

Although restrictions were first placed on the use of PCBs in the United States during the 1970s, lipid-rich finfish and shellfish tissues have continued to accumulate PCBs, dichlorodiphenyl trichloroethane (DDT), and chlordane from the environment (Kennish 1998). PCB congeners are strongly lipophilic and accumulate in fatty tissues including egg masses, affecting the development of fish as well as posing a threat to human health through the consumption of contaminated seafood. Refer to the chapters on Coastal Development and Chemical Effects: Water Discharge Facilities for more additional information on PCB contamination.

Alteration of ocean alkalinity

The influx of acid to the aquatic environment occurs through the atmospheric precipitation of two predominant acids, sulfuric acid and nitric acid, making up acid rain (i.e., pH less than 5.0). Sulfur dioxide is produced naturally by volcanoes and decomposition of plants, while the main anthropogenic source is combustion, especially from coal-burning power plants. In eastern North America, acid rain is ubiquitous because of the presence of coal-burning power plants (Baird 1995). Other sources of sulfuric acid in the atmosphere include oil refinement, cleaning of natural gas, and nonferrous smelting. Affects on biological life depend strongly on soil composition. Granite and quartz have little capacity to neutralize acid, while limestone or chalk can efficiently neutralize acids. Under acidic conditions, aluminum is leached from rocks. Both acidity and high concentrations of dissolved aluminum are responsible for decreases in fish populations observed in many acidified water systems (Baird 1995).

The freshwater environment does not have the buffering capacity of marine ecosystems, so acidification has serious implications on riverine habitat. Low pH (below 5.0) has been implicated with osmoregulation problems (Staurnes et al. 1996), pathological changes in eggs (Peterson et al. 1980; Haines 1981), and reproduction failure in Atlantic salmon (Watt et al. 1983). Cumulative, long-term deposition of acid into the aquatic environment can hinder the survival and sustainability of fisheries by disrupting and degrading important fish and shellfish habitat. Refer to the Coastal Development and Chemical Effects: Water Discharge Facilities chapters for additional information on the affects of acidification of aquatic habitats.

Conservation measures and best management practices for atmospheric deposition

1. Install scrubbers for flue-gas desulfurization in electricity generating powerplants, oil refineries, nonferrous smelters, and other point sources of sulfur dioxide emissions.
2. Use integrated, gas-scrubbing systems on municipal waste combustion units.
3. Reduce sulfur dioxide emissions by substituting natural gas or low-sulfur coal for high-sulfur coal at power plants.
4. Encourage renewable energy generation using wind, solar, and geothermal technologies.
5. Encourage the use of fuel-efficient vehicles and mass transportation systems.
6. Encourage the separation of batteries from the waste stream to reduce the release of mercury vapors through waste incineration.
7. Lower volatization and/or erosion and resuspension of persistent compounds through remediation at waste sites.

Military/Security Activities

The operations of the US military span the globe and are carried out in coastal, estuarine, and marine habitats. Military operations have the potential to adversely impact fish habitat through training activities conducted on land bases as well as in coastal rivers and the open ocean. Military operations also impact fish habitat and larger ecological communities during wars (Literathy 1993).

Because many military bases and training activities are located in coastal areas and oftentimes directly on shorelines, they can cause impacts similar to those mentioned in other parts of this document (e.g., coastal development, dredging, sewage discharge, road construction, shoreline protection, over-water structures, pile driving, port and marina operations, and vessel operations). In addition to these conventional activities, the military often stockpiles and disposes of toxic chemicals on base grounds. Toxic dumping on base grounds has led to the contamination of groundwater at Otis Air National Guard Base on Cape Cod, MA, (NRDC 2003) and in Vieques, Puerto Rico.

The United States Navy also uses sonar systems that create large amounts of noise in ocean waters. The Surveillance Towed Array Sensor System (SURTASS) low frequency active sonar produces extremely loud low frequency sound that can be heard at 140 dB from 300 miles away from the source (NRDC 2004). Sixty percent of the US Navy's 294 ships are equipped with mid-frequency sonar devices that can produce noise above 215 dB (NRDC 2002). The intensity of these noises in the water column can cause a variety of impacts to fish, marine mammals, and other marine life such as behavior alterations, temporary and permanent impairments to hearing, and mortality. Other sources of underwater noise from military activities may include explosive devices and ordnances during training exercises and during wartime. Refer to the Ocean Noise section in this chapter for more information on impacts associated with sonar, as well as the Marine Transportation and Coastal Development chapters for information related to blasting impacts.

Natural Disasters and Events

Introduction

Natural events and natural disasters of greatest concern for the northeastern United States include hurricanes, floods, and drought. These events may impact water quality, alter or destroy habitat, alter hydrological regimes, and result in changes to biological communities. Natural disasters have the potential to impact fishery resources, such as displacing plankton and fish from preferred habitat and altering freshwater inputs and sediment patterns. While these effects may not themselves pose a threat to coastal ecosystems, they may have additive and synergistic effects when combined with anthropogenic influences such as the release of agricultural and industrial pollutants in storm water.

Water quality impacts

Water quality degradation by hurricanes can be exacerbated by human activities. Hurricanes and posthurricane flooding have been known to result in large freshwater inputs and high concentrations of nutrients into river and estuarine waters, causing reductions in water quality and massive fish kills (Mallin et al. 1999). For example, when Hurricane Fran struck North Carolina in the Cape Fear River area in 1996, the following impacts were reported as a result of the hurricane: (1) power failures caused the diversion of millions of liters of raw and partially treated human waste into rivers when sewage treatment plants and pump stations were unable to operate; (2) dissolved oxygen concentrations decreased in parts of the Cape Fear River for more than three weeks following the hurricane; (3) ammonium and total phosphorous concentrations were the highest recorded in 27 years of monitoring in Northeast Cape Fear River following the hurricane and; (4) sediment-laden waters flowing into Cape Fear River increased turbidity levels (Mallin et al. 1999).

Generally, high rates of flushing and reduced water residence times will inhibit the formation of algal blooms in bays and estuaries. However, the input of large amounts of human and animal waste can greatly increase the biological oxygen demand and lead to hypoxic conditions in aquatic systems. In addition to the diversion of untreated waste from sewage treatment plants during Hurricane Fran, several swine waste lagoons were breached, overtopped, or inundated, discharging large quantities of concentrated organic waste into the aquatic environment (Mallin et al. 1999). Other sources of nutrient releases during storms and subsequent flooding events include septic systems on private residences built on river and coastal floodplains.

Natural disasters, such as hurricanes, may also put vessels (e.g., oil tankers) and coastal industrial facilities (e.g., liquefied natural gas [LNG] facilities, nuclear power plants) at risk of damage and contaminant spills. Tanker ship groundings generally occur during severe storms, when moorings are more susceptible to being broken and the control of a vessel may be lost or compromised. The release of toxic chemicals from damaged tanks, pipelines, and vessels threaten aquatic organisms and habitats.

Changes to community composition

Major storm events may impact benthic communities through a variety of mechanisms, including increased sedimentation, introduction of contaminants, reduction in dissolved oxygen, short-term changes in salinity, and disturbance from increased flow. Monitoring of environmental impacts following Hurricane Fran in 1996 indicated that significant declines in benthic organism abundance were observed up to three months after the storm. However, significant declines in benthic abundance generally did not occur in areas where levels of dissolved oxygen recovered quickly after the storm (Mallin et al. 1999). Poorly flushed bays and inland river floodplains are areas that typically exhibit greater magnitude and duration of storm-related impacts.

Loss/alteration of habitat

The rate of accretion and erosion of coastal areas is influenced by wave energy impacting the shoreline, and natural events such as hurricanes will accelerate this process. Erosion may occur as a function of hydraulic scour produced by hurricane overwash and offshore-directed wave energy. Accretion of materials resulting from overwash deposition may result in subsequent flood tidal delta development. Extreme climatic events, such as hurricanes and tsunamis, can have large-scale impacts on submerged aquatic vegetation communities (Orth et al. 2006). Loss or alteration of coastal habitat as a result of storms may be exacerbated by the effects of shoreline development and erosion control measures. For example, the creation of hardened shoreline structures (e.g., seawalls, jetties) and storm-water control systems can focus storm energy and redirect storm water to wetlands, resulting in increased erosion and habitat loss in productive fishery habitat.

Alteration of hydrological regimes

Hurricane and flood events result in large volumes of water delivered to the watershed in a relatively short period of time. These events can alter the hydrology of wetlands, streams, and rivers by increasing erosion and overwhelming flood control structures. Freshwater flows into rivers draining into Charleston Harbor in South Carolina increased as much as four times the historical average after Hurricane Hugo in 1989 (Van Dolah and Anderson 1991). Reduced dissolved oxygen concentrations were observed in all portions of the Charleston Harbor estuary following Hurricane Hugo, with hypoxic conditions in some of the rivers in the watershed. The decomposition of vegetation and the failure of septic and sewer systems overflowing into the watershed as a result of this hurricane was identified as the primary cause of the high organic loads (Van Dolah and Anderson 1991). At the other extreme, drought will result in reduced run-off and low flows in streams and rivers that drain into estuaries and bays. Low freshwater input resulted in dramatic reductions in phytoplankton and zooplankton in San Francisco Bay, CA, reducing pelagic food for fish populations (Bennett et al. 1995). Larval starvation may limit recruitment. During low-flow years, toxins from agricultural and urban runoff are less diluted which can also harm fish.

Conservation measures and best management practices for natural disasters and events

1. Require backup generating systems for publicly owned waste treatment facilities.
2. Prohibit development of high-risk facilities, such as animal waste lagoons, storage of hazardous chemicals within the 100-year floodplain.
3. Ensure that all industrial and municipal facilities involving potentially hazardous chemicals and materials have appropriate emergency spill response plans, including emergency notification systems and spill cleanup procedures, training, and equipment.
4. Encourage the protection and restoration of coastal wetlands and barrier islands, which buffer the affects of storm events by dissipating wave energy and retaining floodwaters.
5. Discourage new construction and development in or near coastal and riparian wetlands.
6. Discourage the use of "hard" shoreline stabilization, such as seawalls and bulkheads.
7. Limit emergency authorizations (e.g., federal Clean Water Act permits) for reconstruction projects to replacing structures that were in-place and functional at the time of the natural disaster/event and do not include the expansion of structures and facilities.

Electromagnetic Fields

Anthropogenic activities are responsible for the majority of the overall electromagnetic fields (EMF) emitted into the environment, with natural sources making up the remainder. Levels of EMF from anthropogenic sources have increased steadily over the past 50-100 years (WHO 2005). Anthropogenic sources of EMF include undersea power cables, high voltage power lines, radar, FM radio and TV transmitters, cell phones, high frequency transmitters for atmospheric research, and solar power satellites. The EMF created by undersea power cables may have some adverse affect on marine organisms. Undersea power cables transfer electric power across water, usually conducting very large direct currents (DC) of up to a thousand amperes or more. It has been inferred that undersea cables can interfere with the prey sensing or navigational abilities of animals in the immediate vicinity of the sea cables (See also the Cables and Pipelines section of the Energy-related Activities chapter). Few published, peer reviewed scientific articles on the environmental effects of electromagnetic fields on aquatic organisms exist. However, the World Health Organization cosponsored an international seminar in October 1999 entitled "Effect of Electromagnetic Fields on the Living Environment" to focus attention on this subject. A review of the information presented at the seminar was prepared by Foster and Repacholi (2000).

Electromagnetic fields are the product of both natural and artificial sources. Natural sources of EMF include radiation from the sun, the earth's magnetic fields, the atmosphere (e.g., lightning discharges), and geological processes (WHO 2005). Marine animals are also exposed to natural electric fields caused by sea currents moving through the geomagnetic field. Examples of anthropogenic sources of EMF include undersea power cables and US Navy submarine communication systems (Foster and Repacholi 2000). Mild electroreception by teleost (bony) fishes occurs through external pit organs that interpret minute electrical currents in the water (Moyle and Cech 1988). However, elasmobranches (i.e., sharks, skates, and rays) are unique in that they possess well-developed electroreceptive organs, called Ampullae of Lorenzini, that enable them to detect weak electric fields in the surrounding seawater as low as 0.01 μV/m (Kalmijn 1971). Elasmobranchs are able to receive information about the positions of their prey, the drift of ocean currents, and their magnetic compass headings from electric fields in their surrounding environment.

Most aquatic organisms emanate low-frequency electric fields that can be detected by fish, such as skates and rays, through a process known as "passive electrolocation" or "passive electroreception." Passive electroreception allows animals to sense electric fields generated in the environment, thereby allowing predators to detect prey by the electric fields that individual fauna emanate. Elasmobranchs have demonstrated during controlled experiments the ability to detect artificially created electric fields (1-5 μA) that are similar to those produced by prey (Kalmijn 1971). The other form of electroreception is "active electroreception" and occurs when an animal detects changes in their own electric field caused by the electric field produced by prey in the vicinity. This ability to detect disturbances to an individual's own electric field is rare, occurring only in a few families of weakly electric fish, none of which are found in the Northwest Atlantic Ocean.

There is evidence that elasmobranches also use their ability to detect electric fields for the purpose of navigation. For example, blue sharks (*Prionace glauca*) have been observed migrating in the North Atlantic Ocean maintaining straight courses for hundreds of kilometers over many days (Paulin 1995). The two modes of detection used for navigation are: (1) passive detection (when an animal estimates its drift from the electrical fields produced by interactions of tidal and wind-driven currents and the vertical component of the earth's magnetic field); or (2) active detection (when the animal derives its magnetic compass heading from the electrical field it generates by its interaction with the horizontal component of the earth's magnetic field) (Gill and Taylor 2001).

Changes in migration of marine organisms

Anthropogenic sources of EMFs may affect social behavior, communications, navigation, and orientation of those animals that rely on the earth's magnetic field. Certain fish rely on the natural (geomagnetic) static magnetic field as one of a number of parameters believed to be used as orientation and navigational cues. For example, stingrays have demonstrated their ability during training experiments to orient relative to uniform electric fields similar to those produced by ocean currents (Kalmijn 1982). In addition, the small-spotted catshark (*Scyliorhinus canicula*) and the thornback skate (*Raja clavata*) have shown a remarkable sensitivity to electric fields (Kalmijn 1982). However, studies demonstrating an impact on the ability of marine organisms to migrate because of anthropogenic sources of EMFs have not been found. Foster and Repacholi (2000) noted the sensitivity of sharks to low frequency electric fields and a potential mechanism for adverse effects from DC fields but made no mention of adverse effects from EMFs.

Changes to feeding behavior

Electric or magnetic fields near sea cables may affect prey sensing of electrically or magnetically sensitive species. Submarine cables may attract species when the field intensity approximates that of their natural prey. Smooth dogfish (*Mustelus canis*) and the blue shark have been observed to execute apparent feeding responses to dipole electric fields designed to mimic prey (Kalmijn 1982). Less is known about how elasmobranches respond in the presence of stronger EMFs that exist closer to the cable. Depending on the presence and strength of electric fields, the feeding behavior of elasmobranches could be altered by submarine cables.

The possible affects of exposure to EMF depend on a coupling between the external field and the body of the animal and the biological response mechanisms. The size of the animal, frequency of the field, and whether the pathway of exposure is via air or water will determine effects to the animal. It has been suggested that monopolar power links are more likely to affect aquatic animals than bipolar links do because they produce perceptible levels of fields over larger distances from the cables (Kalmijn 2000). Sea cables are isolated from the surrounding water by layers of insulation and metal sheathing, yet electric fields that can exceed natural ambient levels remain detectable (Foster and Repacholi 2000). The flow of seawater past the cables can create electric fields by magnetic induction. The resulting field strength in the seawater can exceed naturally occurring levels and depends on the flow velocity, whether or not the observer is moving with respect to the water, and on the electrical conductivity of nearby surfaces (Foster and Repacholi 2000).

Further directed research should be conducted to examine the effect of EMFs from underwater transmission lines on marine organisms. Increased understanding is needed about the effects of cable burial within different substrata and the range of frequencies and sensitivities of electric fields that marine species are capable of detecting.

Conservation recommendations and best management practices for electromagnetic fields

1. Map proposed submarine cable routes with marine resource utilization in a geographic information system database to provide information on potential interference with elasmobranch fishes and other organisms. Particular attention should be paid to known nursery and pupping grounds of coastal shark species.
2. Bury submarine cables below the seafloor to potentially reduce possible interference with the electroreception of fishes. However, the benefits of cable burial to minimize potential impacts to elasmobranchs should be weighed with the adverse effects associated with trenching on the seafloor.
3. Place new submarine electric transmission lines within existing transmission corridors to minimize the cumulative effect of transmission lines across the ocean bottom to the extent practicable.

References for Global Effects and Other Impacts

Anderson DM, Glibert PM, Burkholder JM. 2002. Harmful algal blooms and eutrophication: nutrient sources, composition, and consequences. Estuaries 25(4b):704-26.

Baird C. 1995. Ground-level air chemistry and air pollution. In: Environmental Chemistry. New York (NY): W. H. Freeman and Company. p 90-9.

Bennett WA, Ostrach DJ, Hinton DE. 1995. Larval striped bass condition in a drought-stricken estuary: evaluating pelagic food-web limitation. Ecological Applications 5(3):680-92.

Bigford TE. 1991. Sea-level rise, nearshore fisheries, and the fishing industry. Coastal Management 19:417-37.

Bindler R. 2003. Estimating the natural background atmospheric deposition rate of mercury utilizing ombrotrophic bogs in southern Sweden. Environmental Science and Technology 37:40-6.

Brodeur JC, Sherwood G, Rasmussen JB, Hontela A. 1997. Impaired cortisol secretion in yellow perch (*Perca flavescens*) from lakes contaminated by heavy metals: *in vivo* and *in vitro* assessment. Canadian Journal of Fisheries and Aquatic Sciences 54 (12):2752-8.

Buchholtz ten Brink MR, Manheim FT, Bothner MH. 1996. Contaminants in the Gulf of Maine: what's there and should we worry? In: Dow D, Braasch E, editors. The health of the Gulf of Maine ecosystem: cumulative impacts of multiple stressors. Hanover (NH): Dartmouth College, Regional Association for Research on the Gulf of Maine (RARGOM) Report 96-1. p 91-5.

Castro MS, Driscoll CT, Jorden TE, Reay WG, Boynton WR. 2003. Sources of nitrogen to estuaries in the United States. Estuaries 26(3):803-14.

Cloern J. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210:223-52.

Edwards M, Richardson A. 2004. Impact of climate change on marine pelagic phenology and trophic mismatch. Nature 430(7002):881-4.

Engås A, Lokkeborg S, Ona E, Soldal AV. 1996. Effects of seismic shooting on local abundance and catch rates of cod (*Gadus morhua*) and haddock (*Melanogrammus aeglefinus*). Canadian Journal of Fisheries and Aquatic Sciences 53:2238-49.

Fernandez A, Edwards JF, Rodriguez F, Espinosa De Los Monteros A, Herraez P, Castro P, Jaber JR, Martin V, Arbelo M. 2005. ‘‘Gas and fat embolic syndrome’’ involving a mass stranding of beaked whales (Family Ziphiidae) exposed to anthropogenic sonar signals. Veterinary Pathology 42:446-57.

Foster KR, Repacholi MH. 2000. Environmental impacts of electromagnetic fields from major electrical technologies. In: Matthes R, Bernhardt JH, Repacholi MH, editors. Proceedings of the international seminar on effects of electromagnetic fields on the living environment; 1999 Oct. 4-5; Ismaning, Germany. Munich (Germany): International Commission on Non-Ionizing Radiation Protection. World Health Organization (WHO). 17 p.

Frumhoff PC, McCarthy JJ, Melillo, SC, Moser SC, Wuebbles DJ. 2007. Confronting climate change in the U.S. Northeast: science, impacts, and solutions. Synthesis report of the Northeast Climate Impacts Assessment (NECIA). Cambridge (MA): Union of Concerned Scientists (UCS). 146 p.

Gill AB, Taylor H. 2001. The potential effects of electromagnetic fields generated by cabling between offshore wind turbines upon elasmobranch fishes. University of Liverpool: Research project for Countryside Council of Wales. CCW Science report No. 488. 73 p.

Gould E, Clark PE, Thurberg FP. 1994. Pollutant effects on dermersal fishes. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 30-41.

Greene CH, Pershing AJ. 2007. Oceans: climate drives sea change. Science 315:1084-5.

Gutierrez BT, Williams SJ, Thieler ER. 2007. Potential for shoreline changes due to sea-level rise along the U.S. mid-Atlantic region. [Internet]. Reston (VA): US Geological Survey Open-File Report 2007-1278. [cited 2008 July 28]. 26 p. Available from: http://pubs.er.usgs.gov/usgspubs/ofr/ofr20071278.

Haines TA. 1981. Acidic precipitation and its consequences for aquatic ecosystems: a review. Transactions of the American Fisheries Society 110(6):669-707.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Hastings MC, Popper AN. 2005. Effects of sound on fish. Sacramento (CA): Jones & Stokes. CADOT Contract No. 43A0139, Task Order 1. 82 p.

Heathershaw AD, Ward PD, David AM. 2001. The environmental impact of underwater sound. Proceedings of the Institute of Acoustics 23(4).

Hildebrand J. 2004. Sources of anthropogenic sound in the marine environment. [Internet]. In: Vos E, Reeves RR, editors. Report of an international workshop: policy on sound and marine mammals; 2004 Sep 28-30; London, England. Bethesda (MD): Marine Mammal Commission. [cited 2008 Jul 21]. 129 p. Available from: http://www.mmc.gov/sound/internationalwrkshp/pdf/hildebrand.pdf.

Hodgkins GA, Dudley RW. 2005. Changes in the magnitude of annual and monthly streamflows in New England, 1902-2002. Denver (CO): US Department of the Interior, US Geological Survey. Scientific Investigations Report 2005-5135. 23 p. plus appendices.

Holland TA, Dentener FJ, Braswell BH, Sulzman JM. 1999. Contemporary and pre-industrial global reactive nitrogen budgets. Biogeochemistry 46:7-43.

Howarth RW, Sharpley A, Walker D. 2002. Sources of nutrient pollution to coastal waters in the United States: implications for achieving coastal water quality goals. Estuaries 25(4b):656-76.

Hubbs CL, Rechnitzer AB. 1952. Report on experiments designed to determine effects of underwater explosions on fish life. California Fish and Game 38:333-66.

[IPCC] Intergovernmental Panel on Climate Change. 2007a. Climate change 2007: the physical science basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. [Internet]. In: Solomon S, Qin D, Manning M, Chen Z, Marquis M, Averyt KB, Tignor M, Miller HL, editors. New York (NY): Cambridge University Press. [cited 2008 Jul 9]. 996 p. Available from: http://www.ipcc.ch.

[IPCC] Intergovernmental Panel on Climate Change (IPCC). 2007b. Climate change 2007: impacts, adaptation and vulnerability. Contribution of Working Group II to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. [Internet]. In: Parry ML, Canziani OF, Palutikof JP, van der Linden PJ, Hanson CE, editors. New York (NY): Cambridge University Press. [cited 2008 Jul 9]. 976 p. Available from: http://www.ipcc.ch.

Jasny M, Reynolds J, Horowitz C, Wetzler A. 1999. Sounding the depths: supertankers, sonar, and rise of undersea noise. Los Angeles (CA): Natural Resources Defense Council. Marine Mammal Protection Project. 38 p.

Jaworski NA, Howarth RW, Hetling LJ. 1997. Atmospheric deposition of nitrogen oxides onto the landscape contributes to coastal eutrophication in the northeast United States. Environmental Science and Technology 31:1995-2004.

Jickells TD. 1998. Nutrient biogeochemistry of the coastal zone. Science 281(5374):217-22.

Kalmijn AJ. 1971. The electric sense of sharks and rays. Journal of Experimental Biology 55:371-83.

Kalmijn AJ. 1982. The electric and magnetic field detection in elasmobranch fishes. Science 218(4575):916-18.

Kalmijn AJ. 2000. Detection and processing of electromagnetic and near-field acoustic signals in elasmobranch fishes. Biological Sciences 335(1401):1135-41.

Karl TR, Knight RW. 1998. Secular trends in precipitation amount, frequency, and intensity in the United States. Bulletin of the American Meteorological Society 79(2):231-41.

Keevin TM. 1998. A review of natural resource agency recommendations for mitigating the impacts of underwater blasting. Reviews in Fisheries Science 6(4):281-313.

Keevin TM, Gaspin JB, Gitschlag GR, Hempen GL, Linton TL, Smith M, Wright DG. 1999. Underwater explosions: natural resource concerns, uncertainty of effects, and data needs. In: Proceedings of the twenty-fifth annual conference on explosives and blasting technique; 1999 Feb 7-10; Nashville, TN. Cleveland (OH): International Society of Explosive Engineers. p 105-16.

Kennedy VS, Twilley RR, Kleypas JA, Cowan JH, Hare SR. 2002. Coastal and marine ecosystems and global climate change: potential effects on U.S. resources. Arlington (VA): Pew Center on Global Climate Change. 51 p.

Kennish MJ, Ruppel BE. 1998. Organochlorine contamination in selected estuarine and coastal marine finfish and shellfish of New Jersey. Water, Air, and Soil Pollution 101(1-4):123-36.

Linton TL, Landry AM Jr, Buckner JE Jr, Berry RL. 1985. Effects upon selected marine organisms of explosives used for sound production in geophysical exploration. The Texas Journal of Science 37(4):341-53.

Literathy P. 1993. Considerations for the assessement of environmental consequences of the 1991 Gulf War. Marine Pollution Bulletin 27:349-56.

Mallin MA, Posey MH, Shank GC, McIver MR, Ensign SH, Alphin TD. 1999. Hurricane effects on water quality and benthos in the Cape Fear watershed: natural and anthropogenic impacts. Ecological Applications 9(1):350-62.

Mason RP, Lawson NM, Sheu GR. 2000. Annual and seasonal trends in mercury deposition in Maryland. Atmospheric Environment 34:1691-701.

Mountain DG. 2002. Potential consequences of climate change for the fish resources in the Mid-Atlantic region. In: McGinn NA, editor. Fisheries in a changing climate. Bethesda (MD): American Fisheries Society, Symposium 32.

Mountain DG, Murawski SA. 1992. Variation in the distribution of fish stocks on the northeast continental shelf in relation to their environment, 1980-1989. ICES Marine Science Symposia 195:424-432.

Moyle P, Cech J. 1988. Fishes: an introduction to ichthyology. Englewood Cliffs (NJ): Prentice-Hall. 559 p.

[NOAA] National Oceanic and Atmospheric Administration. 2005. Final environmental impact statement for minimizing impacts of the Atlantic herring fishery on essential fish habitat. Gloucester (MA): National Marine Fisheries Service, Northeast Regional Office. 274 p.

[NRC] National Research Council. 2000. Clean coastal waters: understanding and reducing the effects of nutrient pollution. Washington (DC): National Academy Press. 405 p.

[NRDC] Natural Resource Defense Council. 2002. Federal court blocks full deployment of navy low frequency active sonar system. [Internet]. NRDC Press Archive, October 31, 2002. [cited 2007 Dec 21]. Available from: http://www.nrdc.org/media/pressreleases/021031.asp.

[NRDC] Natural Resource Defense Council. 2003. Above the law? The Pentagon is taking aim at America's public health and environmental protections. [Internet]. NRDC Backgrounder, March 12, 2003. [cited 2007 Dec 21]. Available from: http://www.nrdc.org/media/pressreleases/030312.asp.

[NRDC] Natural Resources Defense Council. 2004. Coalition warns navy over destructive use of mid-frequency sonar. [Internet]. NRDC Press Archive, July 15, 2004. [cited 2007 Dec 21]. Available from: http://www.nrdc.org/media/pressreleases/040715.asp.

Nedeau E. 2004. Effects of climate change on the Gulf of Maine region. [Internet]. Gulf of Maine Council on the Marine Environment. [cited 2008 Jul 9]. 12 p. Available from: http://www.gulfofmaine.org/summit/ClimateChangeKbase.pdf.

Nriagu JO. 1990. Global metal pollution. Poisoning the biosphere? Environment 32:7-33.

Orr JC, Fabry VJ, Aumont O, Bopp L, Doney SC, Feely RA, Gnanadesikan A, Gruber N, Ishida A, Joos F, and others. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. Nature 437:681-6.

Orth RJ, Carruthers TJB, Dennison WC, Duarte CM, Fourqurean JW, Heck KL Jr, Hughes AR, Kendrick GA, Kenworthy WJ, Olyarnik S, and others. 2006. A global crisis for seagrass ecosystems. Bioscience 56(12):987-96.

Paerl HW. 1995. Coastal eutrophication in relation to atmospheric nitrogen deposition: current perspectives. Ophelia 41:237-59.

Paulin MG. 1995. Electroreception and the compass sense of sharks. Journal of Theoretical Biology 174(3):325-39.

Perry AL, Low PJ, Ellis JR, Reynolds JD. 2005. Climate change and distribution shifts in marine fisheries. Science 308:1912-5.

Peterson RH, Daye PG, Metcalfe JL. 1980. Inhibition of Atlantic salmon (*Salmo salar*) hatching at low pH. Canadian Journal of Fisheries and Aquatic Sciences 37:770-4.

Pirrone N, Keeler GJ, Nriagu JO. 1996. Regional differences in worldwide emissions of mercury to the atmosphere. Atmospheric Environment 30:2981-7.

Richardson A, Schoeman D. 2004. Climate impact on plankton ecosystems in the Northeast Atlantic. Science 305(5690):1609-12.

Riebesell U. 2004. Effects of CO_2 enrichment on marine phytoplankton. Journal of Oceanography 60:719-29.

Rogers PH, Cox M. 1988. Underwater sound as a biological stimulus. In: Atema J, Fay RR, Popper AN, Tavolga WN, editors. Sensory biology of aquatic animals. New York (NY): Springer-Verlag. p 131-49.

Scavia D, Field JC, Boesch DF, Buddemeier RW, Burkett V, Cayan DR, Fogarty M, Harwell MA, Howart RW, Mason C, and others. 2002. Climate change impacts on U.S. coastal and marine ecosystems. Estuaries 25(2):149-64.

Schneider D. 1998. The rising seas. Oceans; Scientific American Quarterly 9(3):28-35.

Scholik AR, Yan HY. 2002. Effects of boat engine noise on the auditory sensitivity of the fathead minnow, *Pimephales promelas*. Environmental Biology of Fishes 63:203-9.

Schroeder WH, Yarwood G, Niki H. 1991. Transformation processes involving mercury species in the atmosphere-results from a literature survey. Water, Air, and Soil Pollution 56:653-66.

Short FT, Neckles HA. 1999. The effects of global climate change on seagrasses. Aquatic Botany 63 (3-4): 169-96.

Staurnes M, Hansen LP, Fugelli K, Haraldstad O. 1996. Short-term exposure to acid water impairs osmoregulation, seawater tolerance, and subsequent marine survival of smolts of Atlantic salmon (*Salmo salar* L.). Canadian Journal of Fisheries and Aquatic Sciences 53(8):1695-704.

Stocker M. 2002. Fish, mollusks and other sea animals' use of sound, and the impact of anthropogenic noise in the marine acoustic environment. Journal of Acoustical Society of America 112(5):2431.

Teleki GC, Chamberlain AJ. 1978. Acute effects of underwater construction blasting on fishes in Long Point Bay, Lake Erie. Journal of the Fisheries Research Board of Canada 35:1191-8.

[USEPA] US Environmental Protection Agency. 2005a. Technical factsheet on polychlorinated biphenyls (PCBs). [Internet]. Washington (DC): US EPA. [updated 2006 Nov 28; cited 2007 Dec 21]. Available from: http://www.epa.gov/safewater/dwh/t-soc/pcbs.html.

[USEPA] US Environmental Protection Agency (USEPA). 2005b. Consumer factsheet on polychlorinated biphenyls (PCBs). [Internet]. Washington (DC): US EPA Office of Groundwater and Drinking Water. [updated 2006 Nov 28; cited 2007 Dec 21]. Available from: http://www.epa.gov/OGWDW/dwh/c-soc/pcbs.html.

[USFDA] US Food and Drug Adminstration. 2004. What you need to know about mercury in fish and shellfish. [Internet]. Washington (DC): US Department of Health and Human Services and US Environmental Protection Agency. EPA-823-R-04-005. [cited 2008 Jul 28]. Available from: http://www.cfsan.fda.gov/~dms/admehg3.html.

[USGS] US Geological Survey. 2005. Streamflow trends in the United States. Reston (VA): US Department of the Interior, US Geological Survey. USGS Fact Sheet 2005-3017. 3 p.

Van Dolah RF, Anderson GS. 1991. Effects of Hurricane Hugo on salinity and dissolved oxygen conditions in the Charleston Harbor estuary. Journal of Coastal Research Special Issue 8:83-94.

Wang Q, Kim D, Doinysiou DD, Sorial GA, Timberlake D. 2004. Sources and remediation for mercury contamination in aquatic systems-a literature review. Environmental Pollution 131(2004):323-36.

Watt WD, Scott CD, White WJ. 1983. Evidence of acidification of some Nova Scotia rivers and its impact on Atlantic salmon, *Salmo salar*. Canadian Journal of Fisheries and Aquatic Sciences 40:462-73.

Williams MW, Baron JS, Caine N, Sommerfield R, Sanford RJ. 1996. Nitrogen saturation in the Rocky Mountains. Environmental Science and Technology 30:640-6.

[WHO] World Health Organization. 2005. Electromagnetic fields and public health: Effects of EMF on the environment. [Internet]. Munich (Germany): WHO. Fact sheet No. 296. 2005 Dec. [cited 2008 Jul 28]. Available from: http://www.who.int/mediacentre/factsheets/fs296/en/.

Wright DG. 1982. A discussion paper on the effects of explosives on fish and marine mammals in the waters of the Northwest Territories. Winnipeg (Manitoba): Western Region, Department of Fisheries and Oceans. Canadian Technical Report of Fisheries and Aquatic Sciences No. 1052. 16 p.

Zervas C. 2001. Sea level variations of the United States 1854-1999. [Internet]. Silver Spring (MD): National Ocean Service. NOAA Technical Report NOS CO-OPS 36. [cited 2008 Aug 12]. 66 p. Available from: http://tidesandcurrents.noaa.gov/publications/techrpt36doc.pdf.

CHAPTER TWELVE: COMPENSATORY MITIGATION

Introduction

The purpose of this chapter is to describe the need for and use of compensatory mitigation within the context of regulatory review of proposed coastal development activities. This topic has purposefully been included in a separate chapter of this report to reflect NOAA National Marine Fisheries Services' view that compensatory mitigation is a process that is distinct and separate from impact avoidance and minimization. Only a cursory discussion of compensatory mitigation has been attempted in this report because of the complexity and depth that would be required to cover this topic. We have provided a list of websites and publications that the reader may want to refer to for more detailed discussion of compensatory mitigation.

Compensatory mitigation is a means of offsetting unavoidable impacts to natural resources. It cannot be stressed strongly enough that compensatory mitigation should not be considered until a thorough and exhaustive assessment of project alternatives that may be less environmentally damaging and options for avoiding and minimizing impacts has been completed, and all remaining impacts are "unavoidable." The term "unavoidable impacts" is used ubiquitously in environmental impact assessments developed to meet various requirements of the National Environmental Policy Act (NEPA), Clean Water Act (CWA), Magnuson-Stevens Fishery Conservation and Management Act (MSA), Fish and Wildlife Coordination Act, and other laws and regulations.

The MSA identified the continuing loss of marine, estuarine, and other aquatic habitats to be one of the greatest long-term threats to the viability of commercial and recreational fisheries. The consultation requirements of §305(b)(4)(A) of the MSA require that NOAA National Marine Fisheries Service provide recommendations, which may include measures to avoid, minimize, mitigate, or otherwise offset adverse effects on essential fish habitat (EFH), to federal or state agencies for activities that would adversely effect EFH.

According to NEPA regulations, environmental assessments and environmental impact statements must include a discussion of the means to mitigate adverse environmental impacts. However, according to NEPA guidance, the term "mitigation" includes avoidance and minimization in addition to compensatory mitigation, and NEPA does not strictly require agencies to first avoid and minimize before utilizing compensatory mitigation to offset adverse effects. NEPA regulations do, however, require agencies to assess and discuss the environmental effects of all reasonable alternatives, including the means to mitigate any adverse effects.

The Federal CWA 404(b)(1) guidelines prohibit the discharge of dredge or fill material in waters of the United States if there is a practicable alternative. The 404(b)(1) guidelines also require that all waters of the United States will be accorded the full measure of protection under the CWA, including the requirements for appropriate and practicable mitigation. "Appropriate" is based on the values and functions of the aquatic resource that will be impacted, and "practicable" is defined as that which is available and capable of being done after taking into consideration the cost, existing technology, and logistics in light of overall project purposes. The Memorandum of Agreement (MOA) between the US Environmental Protection Agency and the Department of the Army Concerning the Determination of Mitigation under the Clean Water Act Section 404(b)(1) Guidelines states, "Appropriate and practicable compensatory mitigation is required for unavoidable adverse impacts which remain after all appropriate and practicable minimization has been required." This MOA established a three-part sequential process to help guide mitigation decisions, which includes: (1) avoidance – adverse impacts are to be avoided and no discharge shall be permitted if there is a practicable alternative with less adverse impact; (2) minimization – if impacts cannot be avoided, appropriate and practicable steps to minimize adverse impacts must be taken; and (3) compensation – appropriate and practicable compensatory mitigation is required for unavoidable adverse impacts which remain (USDOA and USEPA 1989).

The need for exhausting all practicable alternatives to avoid and minimize adverse impacts prior to consideration of compensatory mitigation is necessary because of the inherent risks associated with compensatory mitigation. Establishing (creating), reestablishing (restoring), and rehabilitating (enhancing) degraded wetlands and/or aquatic habitats have inherent risks. Replicating or restoring the physical and chemical characteristics of fishery habitat, including soil/sediment hydrology and chemistry, hydrologic connections, and water quality are complex undertakings and can require years to achieve desired results. Replicating and restoring the full ecological functions and values of fishery habitat may not occur without additional effort and cost, and there are no assurances of success. In addition, evaluating mitigation performance and success can require considerable pre- and postconstruction monitoring and assessment, which can be time consuming and costly. For these and other reasons, compensatory mitigation should be viewed as a "last resort" option to achieve effective mitigation, with avoidance and minimization of impacts being the initial focus during the impact assessment process.

Once all practicable alternatives have been considered satisfactorily and a least damaging practicable alternative has been selected that effectively avoids and minimizes adverse effects to the maximum extent practicable, measures to offset unavoidable impacts should be assessed and utilized. Compensatory mitigation can be accomplished on-site or off-site (i.e., in relation to the area being impacted) and can either be in-kind or out-of-kind (i.e., compensation with the same or different ecological functions and values). Generally, in order to achieve the functional replacement of the same or similar ecological resources, in-kind should be considered over out-of-kind compensatory mitigation. However, compensatory mitigation decisions are often made in the context of landscape and watershed implications, as well as logistical and technological limitations. Out-of-kind mitigation, should it be considered, should provide services of equal or greater ecological value and should only be employed if in-kind mitigation is deemed impracticable, unfeasible, or less desirable in the watershed context. However, replacing lost or degraded tidal wetlands or other intertidal/subtidal habitats with nontidal (e.g., freshwater) wetlands should not occur.

Compensatory mitigation can be broadly categorized as restoration, creation, enhancement, and preservation (USACE 2002). Restoration includes reestablishment of a wetland or other aquatic resource with the goal of returning natural or historic functions and characteristics to a former or degraded habitat. Restoration may result in a net gain in ecological function and area. Creation or establishment consists of the development of a wetland or other aquatic resource through manipulation of the physical, chemical, or biological characteristics where a wetland did not previously exist. Creation results in a net gain in ecological function and area. Enhancement or rehabilitation includes activities within existing wetlands that heighten, intensify, or improve one or more ecological functions. Enhancement may result in improved ecological function(s), but does not result in a gain in area. Preservation is designed to protect important wetland or other aquatic resources into perpetuity through implementation of appropriate legal and physical mechanisms (i.e., conservation easements, title transfers). Preservation may include protection of upland areas adjacent to wetlands or other aquatic resources. Preservation does not result in a net gain of wetland acres or other aquatic habitats and should only be used in exceptional circumstances. Preservation is best applied in conjunction with restoration and/or enhancement of ecological functions and values and rarely as the sole means of compensation.

Compensatory mitigation can be provided in the form of project-specific mitigation, mitigation banking, or in-lieu fee mitigation (USEPA 2003). Project-specific mitigation is generally undertaken by a permittee or agency in order to compensate for resource impacts resulting from a specific action or permit. The permittee or agency performs the mitigation and is ultimately responsible for implementation and success of the mitigation. Mitigation banking is a wetland area that has been restored, created, or enhanced, which is then set aside ("banked") to compensate for future impacts to wetlands or other aquatic resources. The value of a bank is determined by quantifying the resource functions restored or created in terms of "credits," which can be acquired, upon the approval of regulatory agencies, to meet a project's requirements for compensatory mitigation. The bank sponsor is ultimately responsible for the success of the project. In-lieu fee mitigation involves a program where funds are paid to a natural resource management entity by a permittee or agency to meet their requirements of compensatory mitigation. The fees are used to fund the implementation of either specific or general wetland or other aquatic resource conservation projects. The management entity may be a third party (e.g., nongovernmental organizations, land trusts) or a public agency that specializes in resource conservation, restoration, and enhancement programs.

Below are some general topics and recommendations regarding the assessment and implementation of compensatory mitigation for actions that may adversely affect fishery resources. It may be necessary to include some of these measures as permit conditions or in decision documents in order to ensure that compensatory mitigation is completed satisfactorily and within the agreed upon timeframes.

Baseline information

The primary purpose of providing effective compensatory mitigation should be to restore or replace the ecological functions and values of resources. In order to assess the effectiveness of compensatory mitigation, the baseline or existing functions and values of the project impact site must be known, as well as the target functions and values for the completed compensatory mitigation site. This can only be accomplished through site-specific monitoring and resource assessments. There are a number of assessment methodologies available to accomplish this, and it is important to determine the method(s) that should be used in advance because it will be necessary for the performance evaluation of the completed mitigation site.

Generally, compensatory mitigation should be provided for direct and indirect impacts, as well as short-term, long-term, and cumulative impacts to fishery resources. Indirect, long-term, and cumulative impacts of a development project may be more difficult to identify and quantify than short-term impacts, but they are no less important. In some cases, the adverse effects on aquatic resources from indirect, long-term, and cumulative impacts may be greater than the direct, short-term construction-related impacts. For example, the direct construction-related impacts of deepening a navigation channel for the purpose of expanding a commercial marina may only involve the removal of bottom sediments in the existing channel. Even so, the dredging project may also result in other short-term impacts to benthic resources from sedimentation and turbidity and anchor damage from vessels. Expansion of a marina operation may result in long-term and cumulative impacts to seagrass and riparian vegetation from vessel wakes and prop scour and in chronic turbidity and sedimentation from larger and more frequent vessel activity. Long-term and cumulative impacts from a development project may also determine whether compensatory mitigation is more appropriately located on-site or off-site.

Compensatory mitigation plan

A clear and concise description of the specific habitats and the functions and values that are intended to be restored should be provided in the mitigation plan. Wetlands and other aquatic habitats provide numerous functions and values within an ecosystem, so it is important to identify the specific functions and values that the compensatory mitigation is intended to restore or replace. Performance criteria should be established (e.g., 80% vegetation cover by target species by the end of the second growing season), and specific monitoring and analytical methods to assess the success of the mitigation should be stipulated in advance.

Adaptive management should be incorporated into mitigation plans, when appropriate. While clear and concise performance criteria are important in all compensatory mitigation plans, monitoring data and predetermined ecological indicators should be used to guide the progress of the mitigation and ensure mitigation objectives are met. Effective compensatory mitigation plans should recognize the importance of adaptive management and allow for corrective action when performance measures are not being met.

A compensatory mitigation plan should include requirements for monitoring and performance reporting, including the content and frequency of reports and who should receive the reports. Generally, the reports should be provided concurrently with the completion of performance monitoring to allow for corrective actions to be taken should success criteria not be met. Other features of a mitigation plan may include measures to ensure mitigation site protection, financial assurances, and a description of long-term maintenance requirements, if necessary, and the party or parties responsible for completing the mitigation requirements.

Contingency plans

Contingency plans for the mitigation plan may be necessary to ensure that adequate compensation is provided, particularly for mitigation that is considered a high-risk endeavor, such as restoration of eelgrass beds. The contingency plan may be necessary to extend the completion of the mitigation plan, and it may require supplemental effort (e.g., planting) or call for alternative mitigations (e.g., out-of-kind). If it is determined that mitigation contingencies are necessary, they should be specified in the permit or decision documents.

Mitigation timing

To minimize the time lag between the loss of wetlands or other aquatic resources and the completion of the compensatory mitigation project, implementation of mitigation construction should begin as soon as possible. For example, if mitigation construction must begin during a specific time of year or the ecological functions and values at the mitigation site require multiple years before being realized, it may be desirable for the compensatory mitigation project to begin before the resource impacts occur.

Interim losses

In situations where there will be delays in implementation of compensatory mitigation or a compensatory mitigation project requires several years to complete, interim or temporal losses of ecological functions and values may be substantial. In these cases, compensation of the interim losses of ecological functions and values should be included in the compensatory mitigation plan. There are a number of ways in which compensation of interim losses can be assessed, such as increasing the ratio of acreage lost to acreage replaced. However, “loss of services” analyses, such as the Habitat Equivalency Analysis (HEA), have been used successfully in a number of restoration projects (NOAA 2006). The HEA assumes there is a one-to-one tradeoff between the resource services at the compensatory restoration site and the resource impact site. In other words, it assumes that the resources can be compensated for past losses through habitat replacement projects providing the replacement resources are the same type as the lost or damaged resources (i.e., in-kind mitigation).

For more information and a more detailed discussion about compensatory mitigation, the reader may refer to the following resources.

General compensatory mitigation guidelines

http://www.epa.gov/wetlandsmitigation
http://www.epa.gov/owow/wetlands/guidance
http://www.nap.usace.army.mil/cenap-op/regulatory/draft_mit_guidelines.pdf
http://www.mitigationactionplan.gov/Preservation_8-27-04.htm

Mitigation banking and in-lieu fee programs

http://www2.eli.org/wmb/backgroundb.htm
http://www.gao.gov/new.items/d01325.pdf

Habitat equivalency analysis

http://www.csc.noaa.gov/coastal/economics/habitatequ.htm
http://www.darrp.noaa.gov/library/pdf/heaoverv.pdf

References for Compensatory Mitigation

[NOAA] National Oceanic and Atmospheric Administration. 2006. Habitat equivalency analysis: an overview. [Internet]. Washington (DC): US Department of Commerce, NOAA, Damage Assessment and Restoration Program.[cited 2008 Jul 28]. 24 pp. Available from: http://www.darrp.noaa.gov/library/pdf/heaoverv.pdf

[USACE] US Army Corps of Engineers. 2002. Guidance on compensatory mitigation projects for aquatic resource impacts under the Corps regulatory program pursuant to Section 404(a) of the Clean Water Act and Section 10 of the Rivers and Harbors Act of 1899. [Internet]. Washington DC): USACE. Regulatory Guidance Letter No. 02-2, 2002 Dec 24. [cited 2008 Jul 28]. Available from: http://www.usace.army.mil/cw/cecwo/reg/rgls/RGL2-02.pdf.

[US Army], [USEPA] US Department of the Army, US Environmental Protection Agency. 1989. Memorandum of agreement between the Department of the Army and the Environmental Protection Agency concerning the determination of the Section 404 Program and the application of the exemptions under Section 404(F) of the Clean Water Act. [Internet]. Washington (DC): US Army and US EPA. [cited 2008 Jul 28]. Available from: http://www.usace.army.mil/cw/cecwo/reg/mou/404f.htm.

[USEPA] US Environmental Protection Agency. 2003. Wetlands compensatory mitigation. [Internet]. Washington (DC): US EPA, Office of Wetlands, Oceans, and Watersheds. EPA Report EPA-843-F-03-002. [cited 2008 Jul 28]. Available from: http://www.epa.gov/owow/wetlands/pdf/CMitigation.pdf.

CONCLUSIONS AND RECOMMENDATIONS

The purpose of this chapter is to synthesize the information discussed in the previous chapters of this report and to identify topics for future research and focus. In addition, the participants of the technical workshop on nonfishing impacts identified activities that are known or suspected to have adverse impacts on fisheries habitat, and we have attempted to draw some conclusions (based upon the effects scores) concerning those activities and effects that deserve further scrutiny and discussion. While many of these activities clearly have known direct, adverse impacts on the quantity and quality of fisheries habitat, their effects at the population and ecosystem level are not well known or understood. For example, there are a number of ports and harbors in the northeast region that have been identified as the most contaminated sites in US coastal waters for polycyclic aromatic hydrocarbons, chlorinated hydrocarbons, and trace metals (USEPA 2004; Buchsbaum 2005). Although many of the effects of these pollutants at the cellular, physiological, and whole organism level are known, information on the effects at the population and ecosystem level is less understood.

There were some noteworthy results from the technical workshop, particularly regarding the geographic areas that scored high for some of the activity types and effects. As one might expect, the workshop participants considered impacts on fisheries habitats to be generally focused in nearshore coastal areas. These results are not particularly surprising considering the proximity of riverine and nearshore habitats to industrial facilities, shipping, and other coastal development. Rivers, estuaries, and coastal embayments are essential for fisheries because they serve as nurseries for the juvenile stages of species harvested offshore or as habitats for the prey of commercially important species (Deegan and Buchsbaum 2005). Estuarine and wetland dependent fish and shellfish species account for about 75% of the total annual seafood harvest of the United States (Dahl 2006). In the workshop session on alteration of freshwater systems, several effects scored high in the estuarine/nearshore ecosystem in addition to the riverine ecosystem. For example, impaired fish passage and altered temperature regimes scored high for the riverine and estuarine/nearshore ecosystems in the dam construction/operation and water withdrawal activity types, suggesting that the participants viewed these activities to have broad ecosystem impacts.

Most effects in both the chemical and physical effects workshop sessions scored high in the riverine and estuarine/nearshore ecosystems. In addition, a few of these effects also scored high in the marine/offshore ecosystem. For the chemical effects session, the release of nutrients/eutrophication, release of contaminants, development of harmful algal blooms, contaminant bioaccumulation/biomagnification, and all effects under the combined sewer overflows impact type scored high in all ecosystem types. The concern of the workshop participants regarding these impacts seems to reflect recently published assessments on threats to coastal habitats (USEPA 2004; Deegan and Buchsbaum 2005; Lotze et al. 2006). For example, the 2004 National Coastal Condition Report (USEPA 2004) assessed the condition of estuaries in the northeast to be poor, with 27% of estuarine area as impaired for aquatic life, 31% impaired for human use, and an additional 49% as threatened for aquatic life use. One of the primary factors contributing to poor estuarine conditions in the northeast region is poor water quality, which is typically caused by high total nitrogen loading, low dissolved oxygen concentrations, and poor water clarity. In the northeast region, the contributing factors associated with nutrient enrichment are principally high human population density and, in the mid-Atlantic states, agriculture (USEPA 2004). In addition, harmful algal blooms (HABs) have been associated with eutrophication of coastal waters, which can deplete oxygen in the water, result in hypoxia or anoxia, and lead to large-scale fish kills (Deegan and Buchsbaum 2005). HABs may also contain species of algae that produce toxins, such as red tides, that can kill or otherwise negatively affect large numbers of fish and shellfish, contaminate shellfish beds, and cause health problems in humans. The extent and severity of coastal eutrophication and HABs will likely continue and may worsen as coastal human population density increases. Considerable attention should be focused on the effects of eutrophication on habitat and water quality, the populations of fish and shellfish, and the role of natural versus anthropogenic sources of nutrients in the occurrence of HABs.

For the workshop session on physical effects, entrainment and impingement effects scored high in all ecosystem types. Entrainment and impingement of eggs, larvae, and juvenile fish and shellfish are increasingly being identified as potential threats to fishery populations from a wide variety of activities, including industrial and municipal water intake facilities, electric power generating facilities, shipping, and liquefied natural gas facilities (Hanson et al. 1977; Travnichek et al. 1993; Richkus and McLean 2000; Deegan and Buchsbaum 2005). Future research is needed to assess the long-term and cumulative effects that entrainment and impingement from these activities have on fish stocks, their prey, and higher trophic levels of the marine ecosystem.

The participants of the workshop session on global effects and other impacts scored most effects in the estuarine/nearshore ecosystem as high. However, several effects of climate change scored high for all ecosystems, including alteration of temperature and hydrological regimes, alteration of weather patterns, and changes in community structure. The effects of climate change related to commercial and recreational fisheries have not as of yet been the focus of extensive research. However, greater emphasis on this topic will likely be necessary as the effects of global warming become more pronounced (Bigford 1991; Frumhoff et al. 2007).

A number of activities and effects were identified during the workshop and in the preparation of this report that may pose substantial threats to fisheries habitat, but the extent of the problems they represent and their implications to aquatic ecosystems are not well understood. Some of these activities and effects have only recently been recognized as potential threats, such as the effects of endocrine disrupting chemicals on aquatic organisms and the threats to fisheries from global warming and will require additional research to have a clearer understanding of the mechanism and scope of these problems. However, other effects such as sedimentation on benthic habitats and biota have been the focus of considerable research and attention, but questions remain as to the lethal and sublethal thresholds of sedimentation effects on individual species and its effects on populations. For example, although sedimentation caused by navigation channel dredging is known to adversely affect the demersal eggs of winter flounder (*Pseudopleuronectes americanus*) (Berry et al. 2004; Klein-MacPhee et al. 2004; Wilber et al. 2005) a better understanding of how the intensity and duration of egg burial effects mortality is needed (i.e., lower lethal thresholds). In addition, how do grain size, the type and amount of contamination, and background suspended sediment concentrations affect egg and larvae survival rates, and what are the implications at the population level?

A number of energy-related activities were assessed for adverse effects on fisheries habitat in the technical workshop and in the corresponding report chapter, including offshore liquefied natural gas platforms, wind turbines, and wave and tidal energy facilities. Although various impacts were discussed, there have not been any facilities of this type constructed in the northeast region of the United States at the time of this report. Although we believe the resource assessments for these types of facilities have been based upon the best available information, further monitoring and assessments will be necessary once they are constructed.

The workshop participants identified a number of chemical effects in several sessions that may have a high degree of impact on fisheries, such as endocrine disrupting chemicals and pharmaceuticals in treated wastewater. Pharmaceuticals and personal care products (PPCP) can persist in treated wastewater and have been found in natural surface waters at concentrations of parts per thousand to parts per billion (Daughton and Ternes 1999). Although the range of concentrations of PPCPs may not pose an acute risk, because aquatic organisms may be exposed continually and for multi-generations, the effects on coastal aquatic communities are a major concern (USEPA 2007). Some of these PPCPs include steroid compounds, which may also be endocrine disruptors. Endocrine disruptors can mimic the functions of sex hormones, androgen and estrogen, and can interfere with reproductive functions and potentially result in population-level impacts. Some chemicals shown to be estrogenic include polychlorinated biphenyl (PCB) congeners, pesticides (e.g., dieldrin, dichlorodiphenyl trichloroethane [DDT]), and compounds used in some industrial manufacturing (e.g., phthalates, alkylphenols) (Thurberg and Gould 2005). In addition, some metal compounds have also been implicated in disrupting endocrine secretions of marine organisms (Brodeur et al. 1997). Additional investigation into the effects of PPCPs and endocrine disruptors on aquatic organisms and their potential impacts at the population and ecosystem level is needed.

In addition, the workshop participants identified a number of adverse effects on aquatic ecosystems from introduced/nuisance species, particularly in the estuarine/nearshore ecosystem. Introduction of nonnative invasive species into marine and estuarine waters poses a significant threat to living marine resources in the United States (Carlton 2001). Nonnative species introductions occur through a wide range of activities, including hull fouling and ballast water releases from ships, aquaculture operations, fish stocking and pest control programs, and aquarium discharges (Hanson et al. 2003; Niimi 2004). The rate of introductions has increased exponentially over the past 200 years, and it does not appear that this rate will level off in the near future (Carlton 2001). Increased research focused towards reducing the rate of nonnative species introductions is needed, in addition to a better understanding of the effects of nonnative species on fisheries in the United States.

Overfishing, including fishing effects on habitat, is likely the greatest factor in the decline of groundfish species in New England (Buchsbaum 2005) and is responsible for the majority of fish and shellfish species depletions and extinctions worldwide (Lotze et al. 2006). However, habitat loss and degradation through nonfishing activities (including pollution, eutrophication, and sedimentation) closely follow exploitation as a causative agent in fishery declines and may be equally or more important for some species such as Atlantic salmon (*Salmo salar*) (Buchsbaum 2005; Lotze et al. 2006). Cumulative effects likely play a role in a large majority of historic changes in fish stocks. Worldwide, nearly half of all marine and estuarine species depletions and extinctions involve multiple human impacts, most notably exploitation and habitat loss (Lotze et al. 2006). It is imperative that management measures intended to reduce exploitation, increase habitat protection, and improve water quality be applied holistically and that the cumulative effects of multiple human interactions be considered in both management and conservation strategies (Lotze et al. 2006). The challenges of quantifying the cumulative effects of nonfishing impacts are vast and complex. Nonetheless, the importance of nonfishing impacts on the coastal ecosystem will likely become greater in the future, and we believe fishery managers would be well served by beginning to collaborate with coastal resource managers and integrate signals from nonfishing effects and stresses on the ecosystem with traditional stock assessment models.

References for Conclusions and Recommendations

Berry WJ, Hinchey EK, Rubinstein NI, Klein-MacPhee G. 2004. Winter flounder, *Pseudopleuronectes americanus*, hatching success as a function of burial depth in the laboratory. In: Ninth f latfish biology conference-poster presentation; 2004 Dec 1-2; Westbrook, CT. Woods Hole (MA): Northeast Fisheries Science Center Reference Document 04-13.

Bigford TE. 1991. Sea-level rise, nearshore fisheries, and the fishing industry. Coastal Management 19:417-37.

Brodeur JC, Sherwood G, Rasmussen JB, Hontela A. 1997. Impaired cortisol secretion in yellow perch (*Perca flavescens*) from lakes contaminated by heavy metals: *in vivo* and *in vitro* assessment. Canadian Journal of Fisheries and Aquatic Sciences 54 (12):2752-8.

Buchsbaum RN. 2005. The role of overfishing, pollution, and habitat degradation on marine fish and shellfish populations of New England: summary and conclusions. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. 175 p.

Carlton JT. 2001. Introduced species in U.S. coastal waters: environmental impacts and management priorities. Arlington (VA): Pew Oceans Commission. 28 p.

Dahl TE. 2006. Status and trends of wetlands in the conterminous United States 1998 to 2004. [Internet]. Washington (DC): US Department of the Interior; Fish and Wildlife Service. [cited 2008 Jul 21]. 112 p. Available from: http://wetlandsfws.er.usgs.gov/status_trends/National_Reports/trends_2005_report.pdf.

Daughton CG, Ternes TA. 1999. Pharmaceuticals and personal care products in the environment: agents of subtle change? [Internet]. Environmental Health Perspectives 107(S6): 907-38. [cited 2008 Aug 5]. Available from: http://www.epa.gov/ppcp/pdf/errata.pdf.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Frumhoff PC, McCarthy JJ, Melillo, SC, Moser SC, Wuebbles DJ. 2007. Confronting climate change in the U.S. Northeast: science, impacts, and solutions. Synthesis report of the Northeast Climate Impacts Assessment (NECIA). Cambridge (MA): Union of Concerned Scientists (UCS). 146 p.

Hanson CH, White JR, Li HW. 1977. Entrapment and impingement of fishes by power plant cooling water intakes: an overview. Marine Fisheries Review 39:7-17.

Hanson J, Helvey M, Strach R. editors. 2003. Non-fishing impacts to essential fish habitat and recommended conservation measures. Long Beach (CA): National Marine Fisheries Service (NOAA Fisheries) Southwest Region. Version 1. 75 p.

Klein-MacPhee G, Macy WK, Berry W. 2004. *In situ* effects of suspended particulate loads produced by dredging on eggs of winter flounder (*Pseudopleuronectes americanus*). In: Ninth flatfish biology conference-oral presentation; 2004 Dec 1-2; Water's Edge Resort, Westbrook, CT. Woods Hole (MA): Northeast Fisheries Science Center Reference Document 04-13.

Lotze HK, Lenihan HS, Bourque BJ, Bradbury RH, Cooke RG, Kay MC, Kidwell SM, Kirby MX, Peterson CH, Jackson JBC. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science 312:1806-9.

Niimi AJ. 2004. Role of container vessels in the introduction of exotic species. Marine Pollution Bulletin 49(9-10):778-82.

Richkus WA, McLean R. 2000. Historical overview of the efficacy of two decades of power plant fisheries impact assessment activities in Chesapeake Bay. Environmental Science and Policy 3(Supplement 1):283-93.

Thurberg FP, Gould E. 2005. Pollutant effects upon cod, haddock, pollock, and flounder of the inshore fisheries of Massachusetts and Cape Cod Bays. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline of fisheries resources in New England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 43-66.

Travnichek VH, Zale AV, Fisher WL. 1993. Entrainment of ichthyoplankton by a warmwater hydroelectric facility. Transactions of the American Fisheries Society 122(5):709-16.

[USEPA] US Environmental Protection Agency. 2004. National coastal condition report II. [Internet]. Washington (DC): US EPA Office of Research and Development/Office of Water. EPA-620/R-03/002. [cited 2008 Jul 9]. Available from: http://www.epa.gov/owow/oceans/nccr2.

[USEPA] US Environmental Protection Agency. 2007. Pharmaceuticals and personal care products (PPCP): frequent questions. [Internet]. Washington (DC): US EPA Office of Research and Development. [updated 2007 Dec 14; cited 2008 Aug 5]. Available from: http://www.epa.gov/ppcp/faq.html.

Wilber D, Brostoff W, Clarke D, Ray G. 2005. Sedimentation: potential biological effects of dredging operations in estuarine and marine environments. [Internet]. DOER Technical Notes Collection. Vicksburg (MS): US Army Engineer Research and Development Center. ERDC TN-DOER-E20. [cited 2008 Jul 22]. 14 p. Available from: http://el.erdc.usace.army.mil/dots/doer/pdf/doere20.pdf.

APPENDIX

Technical Workshop on Impacts to Coastal Fishery Habitat from Nonfishing Activities, January 10-12, 2005 in Mystic, CT

Attendee List

Name	Organization/Affiliation	City and State
Michael Johnson	National Marine Fisheries Service	Gloucester, MA
Sean McDermott	National Marine Fisheries Service	Gloucester, MA
Chris Boelke	National Marine Fisheries Service	Gloucester, MA
Marcy Scott	National Marine Fisheries Service	Gloucester, MA
Lou Chiarella	National Marine Fisheries Service	Gloucester, MA
David Tomey	National Marine Fisheries Service	Gloucester, MA
Jennifer Anderson	National Marine Fisheries Service	Gloucester, MA
Mike Ludwig	National Marine Fisheries Service	Milford, CT
Diane Rusanowsky	National Marine Fisheries Service	Milford, CT
Anita Riportella	National Marine Fisheries Service	Highlands, NJ
Stan Gorski	National Marine Fisheries Service	Highlands, NJ
Andy Draxler	National Marine Fisheries Service	Highlands, NJ
Ric Ruebsamen	National Marine Fisheries Service	St. Petersburg, FL
Jeanne Hanson	National Marine Fisheries Service	Anchorage, AK
Heather Ludemann	National Marine Fisheries Service	Silver Spring, MD
Kimberly Lellis	National Marine Fisheries Service	Silver Spring, MD
David Wiley	Stellwagen Bank National Marine Sanctuary	Situate, MA
Leslie-Ann McGee	New England Fishery Management Council	Woods Hole, MA
Sally McGee	New England Fishery Management Council	Mystic, CT
Eric Nelson	US Environmental Protection Agency	Boston, MA
Phil Colarusso	US Environmental Protection Agency	Boston, MA
Cathy Rogers	US Army Corps of Engineers	Concord, MA
Michael Hayduk	US Army Corps of Engineers	Philadelphia, PA
Brenda Schrecengost	US Army Corps of Engineers	Philadelphia, PA
Steven Mars	US Fish and Wildlife Service	Trenton, NJ
Michelle Dione	Wells National Estuarine Research Reserve	Wells, ME
John Sowles	Maine Dept. of Marine Resources	W. Boothbay Harbor, ME
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Trevor Kenchington	Gadus Associates/Fisheries Survival Fund	Nova Scotia, Canada
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