



New England Fishery Management Council

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OMNIBUS ESSENTIAL FISH HABITAT AMENDMENT 2
FINAL ENVIRONMENTAL IMPACT STATEMENT

Appendix G: Non-fishing impacts to Essential Fish Habitat

Adapted from NOAA Technical Memorandum NMFS-NE-209, Impacts to Marine Fisheries Habitat from Non-fishing activities in the Northeastern United States

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2 Introduction

The purpose of this appendix to Omnibus Essential Fish Habitat Amendment 2 is to summarize adverse impacts associated with human activities, other than fishing, which could potentially affect habitats of species managed by the New England Fishery Management Council. This document relies heavily on NOAA Technical Memorandum NMFS-NE-209, Impacts to Marine Fisheries Habitat from Non-fishing Activities in the Northeastern United States (2008) and has been up-dated to include more recent information. In particular, more recent information related to the impacts of climate change, offshore wind development, offshore mineral mining, aquaculture, and liquefied natural gas facilities are included here to update the conclusions of the 2008 report.

The categories of activities and relative severity of impacts described in the 2008 Technical Memorandum were determined and scored at a 2005 workshop, and then additional research and references were reviewed after the workshop during preparation of the document. The workshop categorized fish habitats according to the Jury et al. (1994) scheme, adopted by NOAA's Estuarine Living Marine Resource program, dividing them into riverine, estuarine/nearshore, and marine/offshore. The Jury et al. classification considers areas with salinity values of 5-25 parts per thousand as estuarine/nearshore, and areas with salinity values above 25 ppt as marine/offshore. Non-fishing impacts on riverine habitats are ignored for the purpose of this appendix, as the only New England Council managed species with ties to riverine habitats is Atlantic salmon.

At the 2005 workshop, scoring of the severity of each type of impact on each habitat type was based on the professional judgment of participants. Impacts were scored from 0-5, with 5 representing the most severe impacts, and participants could score an impact as unknown if they were uncertain. The numeric scores were then averaged and converted to high/medium/low index scores as follows:

- Mean or median 4.0 or greater = high impact
- Mean between 2.1 and 3.9 = medium impact
- Mean 2.0 or less = low impact

The summary tables include the estuarine/nearshore and marine/offshore high impacts only, dividing these effects into benthic (affecting the seabed) or pelagic (affecting the water column). Section 2 classifies NEFMC-managed species as pelagic/nearshore, benthic/nearshore, pelagic/offshore, or benthic/offshore, by lifestage. This allows the reader to identify which species may be affected by particular types of impacts.

3 Habitat use of New England Fishery Management Council-managed species

In the following table, individual life stages for each of the 28 species managed by the New England Fishery Management Council (NEFMC) are listed according to the type (benthic or pelagic) and location (estuarine/nearshore vs marine/offshore) where they are most commonly found. Using this table, the high impact types and their potential adverse effects identified in sections 3.1 to 3.10 of this appendix can be linked to the species and life stages that could be affected. The assignments of species and life stages to habitat types and locations in this table are based on the EFH text descriptions and maps approved by the NEFMC in June 2007 and on supplementary information in Appendix B of the DEIS.

Table 1 - NEFMC-managed species and life stages that are commonly found in each habitat type and location.

	Benthic/Seabed	Pelagic/Water Column
Estuarine/Nearshore	American plaice juveniles/adults Atlantic cod juveniles/adults Atlantic herring eggs Atlantic sea scallop all life stages Clearnose skate juveniles/adults Haddock juveniles Little skate juveniles/adults Ocean pout all life stages Redfish juveniles Red hake juveniles Silver hake juveniles/adults Smooth skate juveniles Thorny skate juveniles White hake juveniles/adults Windowpane juveniles/adults Winter flounder eggs/juveniles/adults Winter skate juveniles/adults Yellowtail flounder juveniles/adults	American plaice eggs/larvae Atlantic cod eggs/larvae Atlantic herring larvae/juveniles/adults Atlantic salmon juveniles/adults Atlantic sea scallop larvae Haddock eggs/larvae Monkfish eggs/larvae Pollock eggs/larvae/juveniles Redfish larvae Silver hake all life stages White hake eggs/larvae Windowpane eggs/larvae Winter flounder larvae Witch flounder eggs/larvae Yellowtail flounder eggs/larvae
Marine/Offshore	American plaice juveniles/adults Atlantic cod juveniles/adults Atlantic halibut juveniles/adults Atlantic herring eggs Atlantic sea scallop all life stages Atlantic wolffish all life stages Barndoor skate juveniles/adults Clearnose skate juveniles/adults Deep-sea red crab eggs/juveniles/adults Haddock juveniles/adults Little skate juveniles/adults Monkfish juveniles/adults Ocean pout all life stages Offshore hake juveniles/adults Redfish juveniles/adults Red hake juveniles/adults Rosette skate juveniles/adults	American plaice eggs/larvae Atlantic cod eggs/larvae Atlantic halibut eggs/larvae Atlantic herring larvae/juveniles/adults Atlantic salmon juveniles/adults Atlantic sea scallop larvae Atlantic wolffish larvae Deep-sea red crab larvae Haddock eggs/larvae Monkfish eggs/larvae Offshore hake all life stages Pollock all life stages Redfish larvae Silver hake all life stages White hake eggs/larvae Windowpane eggs/larvae Winter flounder larvae

Appendix G: Non-fishing impacts to habitat

	Benthic/Seabed	Pelagic/Water Column
	Silver hake juveniles/adults Smooth skate juveniles/adults Thorny skate juveniles/adults White hake juveniles/adults Windowpane juveniles/adults Winter flounder eggs/juveniles/adults Winter skate juveniles/adults Witch flounder juveniles/adults Yellowtail flounder juveniles/adults	Witch flounder eggs/larvae Yellowtail flounder eggs/larvae

4 Summary of non-fishing impacts

The following tables summarize those non-fishing activities that potentially have a high impact on the estuarine/nearshore benthic, estuarine/nearshore pelagic, marine/offshore benthic, and marine/offshore pelagic environments.

Table 2 – Non-fishing activities that potentially have a high impact on the benthic estuarine/nearshore environment

Category	Impact type	Potential effects
Coastal development	Flood Control/Shoreline Protection	Altered hydrological regimes
		Altered sediment transport
		Alteration/loss of benthic habitat
		Loss of intertidal habitat
		Reduced ability to counter sea level rise
		Increased erosion/accretion
	Nonpoint Source Pollution and Urban Runoff	Nutrient loading/eutrophication
		Release of heavy metals
		Release of pesticides
		Loss/alteration of aquatic vegetation
		Sedimentation/turbidity
	Overwater Structures	Changes in predator/prey interactions
	Road Construction and Operation	Increased sedimentation/turbidity
		Altered hydrological regimes
		Reduced dissolved oxygen
		Loss/alteration of aquatic vegetation
		Altered tidal regimes
	Wetland Dredging and Filling	Alteration/loss of habitat
		Loss of submerged aquatic vegetation
		Altered hydrological regimes
Loss of wetlands		
Loss of fishery productivity		
Loss of flood storage capacity		
Energy-related	Cables and Pipelines	Habitat conversion
		Siltation/sedimentation/turbidity
		Resuspension of contaminants
		Spills associated with service structure
		Impacts from construction activities
		Loss of benthic habitat
		Physical barriers to habitat
		Impacts to submerged aquatic vegetation
		Impacts to migration
	Liquified Natural Gas	Discharge of contaminants
		Habitat conversion
		Siltation/sedimentation/turbidity
		Release of contaminants (i.e. spills)
		Introduction of invasive species
Benthic impacts from pipelines		
Loss of benthic habitat		

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Category	Impact type	Potential effects	
		Resuspension of contaminants	
		Vessel impacts (e.g. need to dredge)	
	Offshore Wind Energy Facilities	Loss of benthic habitat	
		Habitat conversion	
		Alteration of community structure	
		Spills associated with service structure	
	Petroleum Exploration, Production and Transportation	Oil spills	
		Habitat conversion	
		Loss of benthic habitat	
		Contaminant discharge (e.g. bilge/ballast)	
		Impacts from clean-up activities	
	Wave/Tidal Energy Facilities	Resuspension of contaminants	
		Habitat conversion	
Loss of benthic habitat			
Alteration of freshwater systems	Dam Construction/ Operation	Siltation/sedimentation/turbidity	
		Impaired fish passage	
		Altered hydrological regimes	
		Altered temperature regimes	
		Alteration of extent of tide	
	Dam Removal	Alteration of wetlands	
		Release of contaminated sediments	
	Dredging and Filling, Mining	Loss of submerged aquatic vegetation	
		Change in species communities	
	Water Withdrawal/ Diversion	Impaired fish passage	
Altered temperature regimes			
Marine transportation	Construction and Expansion of Ports and Marinas	Loss of benthic habitat	
		Siltation/sedimentation/turbidity	
		Contaminant releases	
		Loss of submerged aquatic vegetation	
		Conversion of substrate/habitat	
		Altered hydrological regimes	
		Altered tidal prism	
		Loss of wetlands	
		Loss of intertidal flats	
	Navigation Dredging	Contaminant releases	
		Conversion of substrate/habitat	
		Loss of submerged aquatic vegetation	
		Siltation/sedimentation/turbidity	
		Altered hydrological regimes	
		Altered temperature regimes	
		Loss of intertidal flats	
	Operation and Maintenance of Vessels	Loss of wetlands	
		Contaminant spills and discharges	
	Operations and Maintenance of Ports and Marinas	Impacts to benthic habitat	
		Contaminant releases	
			Storm water runoff

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Chemical effects - water discharge facilities	Combined Sewer Overflows	Potential for all of the above effects
	Industrial Discharge Facilities	Release of heavy metals
		Release of chlorine compounds
		Release of pesticides
		Release of organic compounds (e.g. PCBs)
		Release of petroleum products (PAH)
	Sewage Discharge Facilities	Release of inorganic compounds
		Release of nutrients/eutrophication
		Release of contaminants
		Impacts to submerged aquatic vegetation
		Reduced dissolved oxygen
		Siltation/sedimentation/turbidity
		Impacts to benthic habitat
		Changes in species composition
Trophic level alterations		
Introduction of pathogens		
Introduction of harmful algal blooms		
Contaminant bioaccumulation/biomagnification		
Behavioral avoidance		
Physical effects - water intake and discharge facilities	Discharge Facilities	Alteration of salinity regimes
		Alteration of temperature regimes
		Acute toxicity
		Attraction to flow
		Alteration of community structure
		Physical/chemical synergies
		Ballast water discharge
		Release of radioactive wastes
		Turbidity/sedimentation
		Alteration of sediment composition
		Reduced dissolved oxygen
		Habitat exclusion/avoidance
		Restrictions to migration
	Increased need for dredging	
	Intake Facilities	Entrainment/impingement
		Conversion/loss of habitat
		Ballast water uptake
		Alteration of hydrological regimes
		Flow restrictions
		Alteration of community structure
Increased need for dredging		
Agriculture and silviculture	Cropland, Rangelands, Livestock and Nursery Operations	Release of nutrients/eutrophication
		Bank/soil erosion
		Siltation/sedimentation/turbidity
		Release of pesticides, herbicides, fungicides
		Loss/Alteration of wetlands/riparian zone
		Endocrine disruptors

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	Silviculture and Timber Harvest Activities	Release of pesticides, herbicides, fungicides
		Release of nutrients/eutrophication
	Timber and Paper Mill Processing Activities	Chemical contamination release
Introduced and nuisance species and aquaculture*	Aquaculture*	Discharge of organic waste/contaminants
		Seafloor impacts
		Introduction exotic invasive species
		Food web impacts
		Gene pool alterations
		Impacts to water quality
		Changes in species diversity
		Introduction of diseases
		Habitat conversion
		Sediment deposition
	Habitat replacement/exclusion	
	Introduced/ Nuisance Species	Habitat alterations
		Trophic alterations
		Gene pool alterations
Alterations to communities/comp. w/ native spp.		
Introduced diseases		
Changes in species diversity		
Global effects	Atmospheric Deposition	Mercury loading/bioaccumulation
		Nutrient loading/eutrophication
		PCB's and other contaminants
	Climate Change	Alteration of hydrological regimes
		Alteration of temperature regimes
		Alteration of weather patterns
		Changes in community structure
		Changes in dissolved oxygen concentrations
		Nutrient loading/eutrophication
		Release of contaminants
		Alteration in salinity
		Changes in ecosystem structure
	Loss of wetlands	
	Military/Security Activities	Chemical releases
	Natural Disasters and Events	Loss/alteration of habitat
		Impacts to water quality
		Changes in community composition

*The Aquaculture section has been removed from the "Summary of non-fishing impacts" and included as Addendum I.

Table 3 – Non-fishing activities that potentially have a high impact on the pelagic estuarine/nearshore environment

Category	Impact type	Potential effects
Coastal development	Flood Control/Shoreline Protection	Altered sediment transport
		Loss of intertidal habitat
		Reduced ability to counter sea level rise
		Increased erosion/accretion

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Category	Impact type	Potential effects
	Nonpoint Source Pollution and Urban Runoff	Nutrient loading/eutrophication
		Release of heavy metals
		Release of pesticides
		Loss/alteration of aquatic vegetation
		Release of pharmaceuticals
		Sedimentation/turbidity
	Overwater Structures	Changes in predator/prey interactions
	Road Construction and Operation	Increased sedimentation/turbidity
		Impaired fish passage
		Altered hydrological regimes
		Reduced dissolved oxygen
		Loss/alteration of aquatic vegetation
		Fragmentation of habitat
	Wetland Dredging and Filling	Alteration/loss of habitat
		Loss of submerged aquatic vegetation
Altered hydrological regimes		
Loss of wetlands		
Loss of fishery productivity		
Loss of flood storage capacity		
Energy-related	Cables and Pipelines	Water withdrawal
	Liquefied Natural Gas	Discharge of contaminants
		Entrainment/Impingement
		Underwater noise
		Release of contaminants (i.e. spills)
		Resuspension of contaminants
	Offshore Wind Energy Facilities	Alteration of community structure
	Petroleum Exploration, Production and Transportation	Oil spills
		Habitat conversion
		Contaminant discharge (e.g. bilge/ballast)
		Impacts from clean-up activities
	Wave/Tidal Energy Facilities	Altered current patterns
Entrainment/Impingement (i.e. turbine)		
Alteration of hydrological regimes		
Alteration of freshwater systems	Dam Construction/ Operation	Impaired fish passage
		Alteration of extent of tide
		Alteration of wetlands
	Dredging and Filling, Mining	Release of nutrients/eutrophication
		Loss of submerged aquatic vegetation
	Water Withdrawal/ Diversion	Impaired fish passage
Change in species communities		
Marine transportation	Construction and Expansion of Ports and Marinas	Contaminant releases
		Loss of submerged aquatic vegetation
		Altered hydrological regimes
		Altered tidal prism
		Loss of wetlands
		Loss of water column
	Navigation Dredging	Loss of submerged aquatic vegetation

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Category	Impact type	Potential effects
		Loss of intertidal flats
		Loss of wetlands
	Operation and Maintenance of Vessels	Contaminant spills and discharges
Chemical effects - water discharge facilities	Combined Sewer Overflows	Potential for all of the above effects
	Industrial Discharge Facilities	Release of chlorine compounds
		Release of pesticides
		Release of organic compounds (e.g. PCBs)
		Release of petroleum products (PAH)
	Sewage Discharge Facilities	Release of inorganic compounds
		Release of nutrients/eutrophication
		Release of contaminants
		Impacts to submerged aquatic vegetation
		Reduced dissolved oxygen
		Siltation/sedimentation/turbidity
		Changes in species composition
		Trophic level alterations
		Introduction of pathogens
		Introduction of harmful algal blooms
Contaminant bioaccumulation/biomagnification		
Behavioral avoidance		
Physical effects - water intake and discharge facilities	Discharge Facilities	Alteration of salinity regimes
		Alteration of temperature regimes
		Acute toxicity
		Attraction to flow
		Alteration of community structure
		Release of radioactive wastes
		Reduced dissolved oxygen
		Habitat exclusion/avoidance
		Restrictions to migration
		Increased need for dredging
	Gas-bubble disease/mortality	
	Intake Facilities	Entrainment/impingement
		Conversion/loss of habitat
		Ballast water uptake
		Alteration of hydrological regimes
		Flow restrictions
		Alteration of community structure
Increased need for dredging		
Agriculture and silviculture	Cropland, Rangelands, Livestock and Nursery Operations	Release of nutrients/eutrophication
		Siltation/sedimentation/turbidity
		Endocrine disruptors
	Silviculture and Timber Harvest Activities	Release of pesticides, herbicides, fungicides
		Release of nutrients/eutrophication
	Timber and Paper Mill Processing Activities	Chemical contamination release

Appendix G: Non-fishing impacts to habitat

Introduced and nuisance species and aquaculture*	Aquaculture*	Introduction exotic invasive species
		Food web impacts
		Impacts to water column
		Impacts to water quality
		Changes in species diversity
	Introduced/ Nuisance Species	Habitat conversion
		Gene pool alterations
		Alterations to communities/comp. w/ native spp.
		Introduced diseases
		Changes in species diversity
Global effects	Atmospheric Deposition	Mercury loading/bioaccumulation
		Nutrient loading/eutrophication
		PCB's and other contaminants
	Climate Change	Alteration of hydrological regimes
		Alteration of temperature regimes
		Alteration of weather patterns
		Changes in community structure
		Changes in dissolved oxygen concentrations
		Alteration in salinity
		Changes in ecosystem structure
	Natural Disasters and Events	Loss of wetlands
		Loss/alteration of habitat

*The Aquaculture section has been removed from the “Summary of non-fishing impacts” and included as Addendum I.

Table 4 – Non-fishing activities that potentially have a high impact on the benthic marine/ offshore environment

Category	Impact type	Potential effects
Energy-related	Cables and Pipelines	Impacts from construction activities, physical barriers to habitat, impacts to migration
	Liquefied Natural Gas	Discharge of contaminants
	Offshore Wind Energy Facilities	Loss of benthic habitat, habitat conversion
	Petroleum Exploration, Production and Transportation	Oil spills, habitat conversion
Marine transportation	Construction and Expansion of Ports and Marinas	Loss of benthic habitat
Offshore dredging and disposal	Fish Waste Disposal	Introduction of pathogens, release of nutrients/eutrophication, release of biosolids, loss of benthic habitat types
	Offshore Dredge Material Disposal	Burial/disturbance of benthic habitat, conversion of substrate/habitat, changes in sediment composition
	Offshore Mineral Mining	Loss of benthic habitat types, change in community structure, conversion of substrate/habitat, changes in sediment composition
	Petroleum Extraction	Contaminant releases, drilling mud impacts

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Category	Impact type	Potential effects
	Vessel Disposal	Conversion of substrate/habitat, changes in community structure
Chemical effects - water discharge facilities	Combined Sewer Overflows	Potential for all of the above effects
	Industrial Discharge Facilities	Release of organic compounds (e.g. PCBs)
	Sewage Discharge Facilities	Release of nutrients/eutrophication, release of contaminants, introduction of harmful algal blooms, contaminant bioaccumulation/biomagnification
Physical effects - water intake and discharge facilities	Intake Facilities	Entrainment/impingement
Introduced/nuisance species	Introduced/ Nuisance Species	Changes in species diversity
Global effects	Climate Change	Alteration of temperature regimes, changes in community structure
	Ocean Noise	Mechanical injury to marine organisms

Table 5 – Non-fishing activities that potentially have a high impact on the pelagic marine/offshore environment

Category	Impact type	Potential effects
Energy-related	Liquefied Natural Gas	Discharge of contaminants
	Offshore Wind Energy Facilities	Underwater noise
	Petroleum Exploration, Production and Transportation	Oil spills
Offshore dredging and disposal	Fish Waste Disposal	Introduction of pathogens, release of nutrients/eutrophication
	Petroleum Extraction	Contaminant releases, drilling mud impacts
Chemical effects - water discharge facilities	Combined Sewer Overflows	Potential for all of the above effects
	Sewage Discharge Facilities	Release of nutrients/eutrophication, release of contaminants
Physical effects - water intake and discharge facilities	Intake Facilities	Entrainment/impingement
Global effects	Atmospheric Deposition	Mercury loading/bioaccumulation
	Climate Change	Alteration of hydrological regimes, alteration of temperature regimes, alteration of weather patterns, changes in community structure
	Military/Security Activities	Noise impacts
	Ocean Noise	Mechanical injury to marine organisms

4.1 Coastal development

Coastal development activities may have high impacts on benthic and pelagic estuarine/nearshore environments.

Table 6 – Potential impacts of coastal development on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Flood Control/Shoreline Protection	Altered sediment transport; <i>and</i>	√	√
	Increased erosion/accretion	√	√
	Alteration and loss of benthic habitat; <i>and</i>		√
	Loss of intertidal habitat	√	√
	Reduced ability to counter sea level rise	√	√
	Altered hydrological regimes		√
Nonpoint Source Pollution and Urban Runoff	Nutrient loading/eutrophication; <i>including</i>	√	√
	Loss/alteration of aquatic vegetation	√	√
	Release of pesticides and herbicides; <i>including</i>	√	√
	Loss/alteration of aquatic vegetation	√	√
	Sedimentation and turbidity	√	√
	Release of metals	√	√
Road Construction and Operation	Sedimentation, siltation, and turbidity	√	√
	Altered hydrological regimes; <i>and</i>	√	√
	Fragmentation of habitat	√	√
	Reduced dissolved oxygen	√	√
	Loss and alteration of aquatic vegetation	√	√
	Impaired fish passage	√	√
Wetland Dredging and Filling	Alteration of habitat and loss of wetlands	√	√
	Altered hydrological regimes	√	√
	Loss of fishery productivity	√	√
	Loss of flood storage capacity	√	√
Overwater Structures	Changes in predator/prey interactions	√	√

4.1.1 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.

4.1.1.1 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.

4.1.1.2 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.

4.1.1.3 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 12-22 cm (5-9 inches) from 1901 to 2010 and

may rise another 26-82 cm (10-32 inches) by 2100 (IPCC 2013). 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, Deegan and Buchsbaum 2005). 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 alter precipitation patterns and cause more intense storms in the mid-high latitudes in the northern hemisphere (Nedea 2004, IPCC 2013). 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 section for more information on global climate change.

4.1.1.4 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.

4.1.2 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. 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 2005d). 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 2004 National Water Quality Inventory (USEPA 2004c) reported that runoff from urban areas is a leading source of impairment in surveyed estuaries, lakes, and rivers and streams. In a 2007 survey of 6,237 coastal beaches nationwide, runoff was the single most common reason for the issuance of beach advisories, accounting for 35% of the advisories issued (USEPA 2012). Urban areas can have a chronic and insidious pollution potential that one-time events such as oil spills do not. DiDonato et al. (2009) discuss the need and potential to create forecasting models of indicator concentrations under land use and urbanization changes based on microbial contamination levels in tidal creek headwaters.

It is important to note that the effects 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.

4.1.2.1 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, USEPA 2012). 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 2005d). 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 section 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 2012) 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 poor conditions (USEPA 2012).

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 Section 3.6 on Chemical Effects: Water Discharge Facilities for more information on harmful algal blooms.

4.1.2.2 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 Section 3.8 on Agriculture and Silviculture 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 sensitivity 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 disruptors, 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.

4.1.2.3 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 2005d; 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 2005d). For example, Breitbart (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).

4.1.2.4 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 2001b; 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). 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 section 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). 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 Section 3.10 on Global Effects and Other Impacts for mercury loading/bioaccumulation via the atmosphere.

4.1.3 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.

4.1.3.1 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).

4.1.3.2 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).

4.1.3.3 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.

4.1.3.4 Loss and alteration of vegetation

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).

4.1.3.5 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 section of this appendix.

4.1.4 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 section 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 sections 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.

4.1.4.1 Alteration of habitat and loss of wetlands

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).

4.1.4.2 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).

4.1.4.3 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).

4.1.4.4 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.

4.1.4.5 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).

4.1.4.6 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 2001a).

4.2 Energy-related activities

Energy development activities may have high impacts on both estuarine/nearshore and marine/offshore habitats.

Table 7 – Potential impacts of energy facilities and infrastructure on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Cables and Pipelines	Habitat conversion, <i>including</i> :		√
	Loss of benthic habitat		√
	Impacts to submerged aquatic vegetation		√
	Physical barriers to habitat		√
	Impacts to migration		√
	Impacts from construction activities		√
	Siltation/sedimentation/turbidity		√
	Resuspension of contaminants		√
Liquefied Natural Gas	Discharge of contaminants, <i>including</i> :	√	√
	Release of contaminants (i.e. spills)	√	√

Appendix G: Non-fishing impacts to habitat

	Habitat conversion, <i>including</i> :		√
	Loss of benthic habitat		√
	Benthic impacts from pipelines		√
	Siltation/sedimentation/turbidity, <i>including</i> :		√
	Vessel impacts (e.g. need to dredge)		√
	Introduction of invasive species		√
	Entrainment and impingement	√	
	Underwater noise	√	
	Siltation, sedimentation, and turbidity	√	√
Offshore Wind Energy Facilities	Habitat conversion, <i>and</i>		√
	Loss of benthic habitat		√
	Alteration of community structure	√	√
	Spills associated with service structure		√
Petroleum Exploration, Production and Transportation	Oil spills	√	√
	Habitat conversion, <i>and</i>	√	√
	Loss of benthic habitat		√
	Contaminant discharge (e.g. bilge/ballast), <i>including</i> :	√	√
	Resuspension of contaminants		√
Impacts from clean-up activities	√	√	
Wave/Tidal Energy Facilities	Habitat conversion, <i>and</i>		√
	Loss of benthic habitat		√
	Siltation/sedimentation/turbidity		√
	Entrainment and impingement (i.e., turbines)	√	
	Alteration of hydrological regimes, <i>including</i> :	√	
	Altered current patterns	√	

Table 8 – Potential impacts of energy development and infrastructure on marine/offshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Liquefied Natural Gas	Discharge of contaminants	√	√
Offshore Wind Energy Facilities	Underwater noise	√	
	Habitat conversion, <i>and</i>		√
	Loss of benthic habitat		√
Petroleum Exploration, Production and Transportation	Oil spills	√	
	Habitat conversion		√
Cables and Pipelines	<i>Included under Habitat conversion:</i>		
	Impacts from construction activities		√
	Physical barriers to habitat		√
	Impacts to migration		√

4.2.1 Cables and Pipelines

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.

4.2.1.1 Habitat conversion (estuarine/nearshore and marine/offshore impact)

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 escarpments.

This, in turn, can interfere with the migratory patterns of benthic species.

4.2.1.2 Siltation, sedimentation, and turbidity (estuarine/nearshore and marine/offshore)

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 appendix.

4.2.1.3 Resuspension of contaminants (estuarine/nearshore and marine/offshore)

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 appendix. 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).

4.2.2 Liquefied Natural Gas (LNG)

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.

4.2.2.1 Discharge of contaminants (estuarine/nearshore and marine/offshore)

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.

4.2.2.2 Habitat conversion (estuarine/nearshore only)

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 sections for more detailed information on impacts from dredging.

4.2.2.3 Siltation, sedimentation, and turbidity (estuarine/nearshore only)

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 section.

4.2.2.4 Introduction of invasive species (estuarine/nearshore only)

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 sections on Marine Transportation and Introduced/Nuisance Species and Aquaculture for more information on invasive species and shipping.

4.2.2.5 Entrainment and impingement (estuarine/nearshore only)

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).

4.2.2.6 Underwater noise (estuarine/nearshore only)

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 section on Global Effects and Other Impacts.

4.2.3 Offshore Wind Energy Facilities

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, leases have been sold in the Rhode Island/Massachusetts Wind Energy Area (July 2013), the Virginia Wind Energy Area (September 2013), for the Cape Wind project in Nantucket Sound (October 2010), the Bluewater Wind project off Delaware (November 2012), and the Deepwater Wind and Fishermen's Energy of New Jersey off New Jersey in October and November 2010 (for more information, see <http://www.boem.gov/Lease-and-Grant-Information/>). The construction and operation of offshore wind facilities has the potential to adversely affect fishery habitats.

4.2.3.1 Habitat conversion and loss of benthic habitat (estuarine/nearshore and marine/offshore)

The construction of offshore wind turbines and support structures can result in benthic habitat conversion and loss as a result 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 (Inger et al. 2009). 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.

4.2.3.2 Alteration of community structure (estuarine/nearshore only)

Offshore wind energy facilities have the potential to alter the local community structure of the marine ecosystem. The alteration of community structure is not simply a result of habitat conversion effects. In areas where wind farms have been placed in hard substrate dominated habitat types, alterations in community structure have been identified between the communities that develop on the bases of wind turbines and the adjacent hard substrate communities. In the Baltic Sea, over the three year period after installation of a wind farm, the benthic community changed from a dominant blue mussel community (75%) to an almost exclusively blue mussel community (97-99%) that resulted in altered local ecosystem dynamics (Maar et al. 2009). Wilhelmsson and Malm (2008) evaluated benthic community structure at the base of wind turbines and adjacent hard substrate communities. The community structure was significantly different between the wind turbine fouling community and the adjacent hard bottom substrates (Wilhelmsson and Malm 2008).

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 species to increased fishing. Recent studies on juvenile Atlantic cod and pouting at wind farms in the northeast Atlantic illustrated the bases of turbines were supporting aggregations of juveniles within wind farms and indicate that these farms may act as an ecological trap via fishing mortality without the implementation of thorough management restrictions to protect fish aggregations (Reubens et al. 2011, 2013a, 2013b, and 2014). It is likely that floating turbine platforms, typically proposed in deeper waters, will act essentially as floating aggregation devices (FADs) altering the community structure and increasing the susceptibility of aggregated fish to fishing mortality (Fayram and Risi 2007, Inger et al. 2009, Snyder and Kaiser 2009). 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) (Petersen and Malm 2006).

4.2.3.3 Spills associated with service structure (estuarine/nearshore only)

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 appendix, and the sections on Coastal Development and Chemical Affects: Water Discharge Facilities.

4.2.3.4 Underwater noise (marine/offshore only)

Underwater noise during construction of turbines may impact hearing in fish, and may cause fish to disperse with possible disruption to their feeding and spawning patterns. Noise from construction of wind farms (e.g., pile driving) could have significant effects on fish, but the degree to which fish will be impacted will vary (Hoffmann et al. 2000, Snyder and Kaiser 2009). Pile-driving noise associated with construction of wind farms has been recorded at maximum levels of 205 dB at the site of pile driving to a distance of 80m where generated noise diminished to ambient noise levels (104-119 dB) (Bailey et al. 2010). Based on existing records of noise generated during the operation of wind farms, there is a potential for ecological impacts to fish (Kikuchi 2010). Noise generated by the operation of wind turbines is not expected to be as intrusive as construction related noise, but research needs to be conducted to determine if chronic, long term effects may result (Inger et al. 2009). Operational noise 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). 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

section on Global Effects and Other Impacts.

4.2.4 Petroleum Exploration, Production, and Transportation (estuarine/nearshore)

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. Leases were sold and 51 test wells drilled on the outer continental shelf on the U.S. Atlantic coast in the late 1970s and early 1980s, but there has been no additional test well activity since then (see <http://www.boem.gov/Atlantic-Oil-and-Gas-Information/>). 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 U.S. Department of the Interior's Minerals Management Service (MMS), now the Bureau of Ocean Management (BOEM), 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. BOEM is currently in the 2012-2017 planning period for the development of the 2017-2022 Oil and Gas Leasing Program. The Atlantic OCS Region is divided into four planning areas: North Atlantic, Mid-Atlantic, South Atlantic, Straits of Florida, for administrative purposes under the Oil and Gas Leasing Program. At present, no active OCS oil and gas leases exist in any of these four planning areas, and no oil and gas lease sales are proposed under the current 2012-2017 leasing program.

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 absorbed 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).

4.2.4.1 Oil spills (estuarine/nearshore and marine/offshore)

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, Ko and Day 2004). 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). These effects may be short-term or long-term impacts in coastal systems (Ko and Day 2004). Spills contacting coastal vegetation and benthic species have significant adverse impacts to these resources (e.g. vegetation die-back, marsh erosion, decreased benthic diversity and abundance) due to oils physical effects and chemical toxicity (Ko and Day 2004). 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 Mendelsohn 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 (*Oncorhynchus 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).

4.2.4.2 Habitat conversion and loss of benthic habitat (estuarine/nearshore and marine/offshore)

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.

4.2.4.3 Resuspension of contaminants (estuarine/nearshore only)

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).

4.2.4.4 Oil spill clean-up activities (estuarine/nearshore only)

There are a number of oil spill response and cleanup methods available. Chemical dispersants are used primarily in open water environments. Dispersants contain surfactant chemicals 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. There are multiple types of oil dispersants and care should be taken to determine which dispersant is utilized. 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). The toxicity of dispersants to sea urchin embryos is dependent on the type of dispersant with toxicity levels for some dispersants an entire order of magnitude greater than others (Rial et al. 2014). 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. Over the last two decades, studies have revealed that many organisms are capable of degrading hydrocarbons in different salinity and oxygen regimes, but further research needs to be done to determine their placement in cleanup processes (Fathepure 2014). 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 spill itself.

4.2.5 Wave and Tidal Energy Facilities

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. A detailed review of current

literature, beyond the scope reportable here, on the potential adverse habitat and community structure alteration impacts, hydrological impacts, behavioral and reproductive impacts, noise impacts, and electromagnetic field impacts associated with different tidal and wave energy generating devices was published by Frid et al. (2012).

Wave energy can be utilized to generate power from the nearshore area in three ways (see also <http://www.boem.gov/Renewable-Energy-Program/Renewable-Energy-Guide/Ocean-Wave-Energy.aspx>):

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 pendolor, 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).
4. 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 section on Alteration of Freshwater Systems.

Various projects in Maine, New Hampshire, Massachusetts, Rhode Island, and Connecticut are in the siting/planning, site development, and device testing phases. There are no deployed projects in the New England region. Information about current projects can be found here: http://en.openei.org/wiki/Marine_and_Hydrokinetic_Technology_Database.

4.2.5.1 Habitat conversion and loss of benthic habitat (estuarine/nearshore only)

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. These structures may act as artificial reefs resulting in a loss of existing benthic habitat and conversion (Shields et al. 2009). Tidal turbines have been documented to increase tidal ranges and decrease tidal heights resulting in a loss of intertidal habitat (Wolf et al. 2009). Offshore

wave power foundations and buoys may also act as artificial reefs supporting successional biofouling communities (Langhamer et al. 2009). 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). Sedimentation patterns can be influenced by altered water flow resulting in conversion of habitat substrate type (Shields et al. 2009).

4.2.5.2 Siltation, sedimentation, and turbidity (estuarine/nearshore only)

Construction of tidal facilities in riverine and estuarine areas can result in increased sedimentation and altered regimes (Shields et al. 2009, Kadiri et al. 2012). Structures placed within riverine and estuarine habitats can reduce the natural transport of sediments and cause an accretion of silt and sediments within impoundments (Kadirie et al. 2012). 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 appendix.

4.2.5.3 Entrainment and impingement (estuarine/nearshore only)

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. Tidal energy devices have the potential to serve as fish aggregation devices influencing predator distribution and increasing the risk of collision (Shields et al. 2009). The physical presence of tidal power facilities can impact the return of diadromous fishes to natal rivers (Semple 1984).

4.2.5.4 Alteration of hydrological regimes (estuarine/nearshore only)

Water circulation patterns and tidal regimes can be altered during the operation of wave and tidal facilities (Shields et al. 2009, Kadiri et al. 2012). This can result in poor tidal flushing of the

headwaters of estuaries and rivers and can lead to decreased water quality, increased contaminant build-up, salinity alterations, and increases in water temperature (Rulifson and Dadswell 1987, Wolf et al. 2009, Kadiri et al. 2012). Local sedimentation patterns may also be impacted as a result of altered water flows from tidal facilities resulting in increased sediment deposition and sedimentation rates (Shields et al. 2009, Kadiri et al. 2012). 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.

4.3 Alteration of freshwater systems

Table 9 – Potential impacts of the alterations of freshwater systems on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Dam Construction/ Operation	Impaired fish passage	√	√
	Alteration of wetlands, <i>including:</i>	√	√
	Alteration of extent of tide	√	√
	Altered hydrological regimes, <i>and</i>		√
	Altered temperature regimes		√
Dam Removal	Release of contaminated sediments		√
Dredging, Filling, and Mining	Release of nutrients/eutrophication	√	
	Loss of submerged aquatic vegetation	√	√
	Change in species communities		√
Water Withdrawal/ Diversion	Impaired fish passage	√	√
	Change in species communities	√	
	Altered temperature regimes		√

4.3.1 Dam construction and operation (estuarine/nearshore only)

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.

4.3.1.1 Impaired fish passage (estuarine/nearshore only)

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).

4.3.1.2 Alteration of wetlands (estuarine/nearshore only)

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.

4.3.1.3 Altered hydrologic and temperature regimes (estuarine/nearshore only)

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.

4.3.2 Dam removal (estuarine/nearshore only)

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.

4.3.2.1 Release of contaminated sediments (estuarine/nearshore only)

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).

4.3.3 Dredging, filling, mining (estuarine/nearshore only)

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 2001b); (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 2001b).

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).

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 diadromous 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.

4.3.4 Water withdrawal and diversion (estuarine/nearshore only)

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 and forested wetlands (Day et al. 2012).

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.

4.3.4.1 Impaired fish passage (estuarine/nearshore only)

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. Modelling of the Russian River basin in CA for existing diversion demands

indicates that stream flow for anadromous fish passage may be reduced by 20% in one third of streams during spring, and may accelerate summer intermittence in 80% of studied streams (Deitch et al. 2009). Diversions may also impair fish passage by entraining fish in fast flowing structural diversions. Agricultural water diversions utilizing unscreened diversion pipes entrained green sturgeon during migratory spawning periods; up to 52% of sturgeon seeking passage were entrained after passing within 1.5m of an active diversion pipe (Mussen et al. 2014).

4.3.4.2 Changes in species communities (estuarine/nearshore only)

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 (Day et al. 2012). Wetlands may become isolated as a result of even small diversions, significantly impacting wetland growth and accretion rates resulting in a loss of wetland species communities over time (Day et al. 2012). 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. Water diversions can alter salinity regimes resulting in significant changes to estuarine community structures (Mutsert and Cowan 2012, Das et al. 2012). In conjunction with anthropogenic watershed changes, water diversions and associated riparian impacts have been associated with the increase in some harmful algal blooms (HABs) (Boesch et al. 1997), which further impact an array of aquatic habitat characteristics. However, the intensity of the diversion, turbulence, environmental fluctuations, and nutrient composition may mediate the potential for HABs (Roy et al. 2013).

4.3.4.3 Altered temperature regimes (estuarine/nearshore only)

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 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 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).

4.4 Marine transportation

Marine transportation activities may have high impacts on estuarine/nearshore habitats.

Table 10 – Potential impacts of marine transportation on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Construction and Expansion of Ports and Marinas	Loss of benthic habitat, <i>and</i>		√*
	Conversion of substrate/habitat, <i>including:</i>		√
	Loss of wetlands	√	√
	Loss of intertidal flats		√
	Loss of water column	√	
	Siltation/sedimentation/turbidity		√
	Contaminant releases	√	√
	Loss of submerged aquatic vegetation	√	√
	Altered hydrological regimes, <i>including:</i>	√	√
	Altered tidal prism	√	√
Navigation Dredging	Contaminant releases		√
	Conversion of substrate/habitat		√
	Loss of submerged aquatic vegetation	√	√
	Siltation, sedimentation, and turbidity		√
	Altered hydrological regimes		√
	Altered temperature regimes		√
	Loss of intertidal flats, <i>and</i>	√	√
Loss of wetlands	√	√	
Operation and Maintenance of Vessels	Contaminant spills and discharges	√	√
	Impacts to benthic habitat		√
Operations and Maintenance of Ports and Marinas	Contaminant releases, and		√
	Storm water runoff		√

* = Construction and expansion of ports and marinas is also highly likely to cause a loss of marine benthic habitat: no other potential effects of marine transportation activities were identified for marine habitats

4.4.1 Construction and expansion of ports and marinas (estuarine/nearshore only)

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.

4.4.1.1 *Loss of benthic habitat and conversion of substrate/habitat (estuarine/nearshore only)*

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.

4.4.1.2 Siltation, sedimentation, and turbidity (estuarine/nearshore only)

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).

4.4.1.3 Contaminant releases (estuarine/nearshore only)

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 effects 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.

4.4.1.4 Loss of submerged aquatic vegetation (estuarine/nearshore only)

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).

4.4.1.5 Altered hydrologic regime (estuarine/nearshore only)

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.

4.4.2 Navigation dredging (estuarine/nearshore only)

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).

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.

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).

4.4.2.1 Contaminant releases (estuarine/nearshore only)

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, Su et al. 2002). 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.

4.4.2.2 Loss or conversion of substrate/habitat (estuarine/nearshore only)

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).

4.4.2.3 Loss of submerged aquatic vegetation (estuarine/nearshore only)

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, Duarte 2002). 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, Erftemeijer and Lewis 2006). 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, Erftemeijer and Lewis 2006). The extent of damage to SAV can not be simplified to the extent and scale of the dredging operation, but also depends on the proximity to SAV beds, sediment type and composition, dredge methodology, mitigation measures, and other factors (Erftemeijer and Lewis 2006).

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). The extent of predicted SAV impacts and cumulative effects are issues of concern in permit processes (Erftemeijer and Lewis 2006). Erftemeijer and Lewis (2006) provide a recent review of research on impacts to SAV from dredging activities, mitigation measures, and regulatory processes utilized in reviewing dredging projects world-wide.

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).

4.4.2.4 Siltation, sedimentation, and turbidity (estuarine/nearshore only)

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). A recent study by Suedel et al. (2014) did not detect effects of short duration exposure (seven day trial) to increased turbidity levels up to 500mg/L in oysters from a riverine environment, but did detect a difference of turbidity response in weight change based on attachment position (vertical versus horizontal). In other studies, settlement of suspended sediments in a layer as little as 1-2mm thick significantly reduced oyster spat settlement (Wilbur and Clark 2001). 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).

4.4.2.5 Altered hydrologic regimes (estuarine/nearshore only)

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).

4.4.2.6 Altered temperature regimes (estuarine/nearshore only)

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.

4.4.2.7 Loss of intertidal habitat and wetlands (estuarine/nearshore only)

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).

4.4.3 Operation and maintenance of vessels (estuarine/nearshore only)

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 vanHofen 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).

4.4.3.1 Contaminant spills and discharges (estuarine/nearshore only)

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. While these events are relatively rare and usually involve small geographic areas, oil spills in marine protected areas (MPAs) are of the greatest concern for fisheries resources. From 2002 through 2006, the number of large spills (>100,000 gal) was greater inside MPAs with 71% of the total volume of vessel oil spills occurring in

federal fisheries closures (Dalton and Jin 2010).

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).

4.4.3.2 Impacts to benthic habitat (estuarine/nearshore only)

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).

4.4.4 Operation and maintenance of ports and marinas (estuarine/nearshore only)

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.

4.4.4.1 Contaminant release and storm water runoff (estuarine/nearshore only)

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 2001b). 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.

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 2001b). 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).

4.5 Offshore dredging and disposal

Offshore dredging and disposal may have high impacts on marine/offshore habitats.

Table 11 – Potential impacts of offshore dredging and disposal on marine/offshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Fish Waste Disposal	Introduction of pathogens	√	√
	Release of nutrients/eutrophication	√	√
	Release of bio-solids		√
	Alteration/Loss of benthic habitat types		√
Offshore Dredge Material Disposal	Conversion of substrate/habitat, and Changes in sediment composition		√
			√
Offshore Mineral Mining	Loss of benthic habitat types		√
	Conversion of substrate/habitat, and Changes in sediment composition, including:		√
	Change in community structure		√
	Burial/disturbance of benthic habitat		√
Petroleum Extraction	Contaminant releases	√	√
	Drilling mud impacts	√	√
Vessel Disposal	Conversion of substrate/habitat, and Changes in community structure		√
			√

4.5.1 Fish waste disposal

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 2005). 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 2005). 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 2005).

4.5.1.1 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 2005). 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.

4.5.1.2 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. Effluent releases in nearshore habitats have increased potential for adverse impacts to resources from releases. The effect of these releases to fish is variable by species and can result in acute toxicity to fish in the vicinity of a release (Jamieson et al. 2010). In ocean disposal, these effects may be seen with mounding of wastes, subsequent increases in BOD and contamination with bacteria associated with partly degraded organic wastes (IMO 2005). However, disposal guidelines require that dumpsite selection criteria maximizes waste dispersion and consumption of the wastes by marine organisms.

4.5.1.3 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 2005). Biosolid waste at fish disposal sites could result in nutrient over-enrichment and reduced dissolved oxygen concentration. Releases in nearshore habitats have increased potential for adverse impacts to resources from releases. As mentioned above, the effect of these releases to fish is variable by species and can result in acute toxicity (Jamieson et al. 2010). 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.

4.5.1.4 Alteration/loss 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 anaerobic conditions and release of hazardous and toxic chemical compounds into the marine environment (Islam et al. 2004). This can lead to reductions of small consumer organisms that 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 2005).

4.5.2 Offshore dredged material disposal

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 further 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).

4.5.2.1 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 notwithstanding, 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).

4.5.2.2 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. A recent review of disposal sites around England and Wales illustrated that the burial of benthic habitat resulted in significantly decreased production and functional values, and significant differences in structural parameters (Bolam 2012).

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). Similar results were found for opportunistic species by O'Donnell et al. (2007) where different responses from ecologically similar species following disposal of dredge materials were identified in Penobscot Bay, Maine. No significant differences were observed in lobster abundance, attributable to the time of year the disposal took place, but an increase in opportunistic rock crab abundance was observed following disposal, attributed to the increased availability of invertebrates and other food resources in the deposited sediment mounds (O'Donnell et al. 2007).

4.5.3 Offshore mineral mining

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.

4.5.3.1 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).

4.5.3.2 Conversion of substrate/habitat and changes in community structure

Overspill of sediments during mining operations can alter habitat type, functions, and values. The alteration of these habitat parameters will impact benthic community structure and rates of recovery (Cooper et al. 2007a, 2007b, 2011). Disturbance of the seafloor during mining operations will alter benthic community structure through direct removal of species and cause indirect impacts to adjacent habitats as a result of increased turbidity and deposition of suspended sediments (Scarrat 1987, Cooper et al. 2007a). The natural composition of benthic communities may be stochastic, such as is expected in successional communities, or dynamic where community structure continually changes overtime. Persistent temporal changes in benthic communities following the cessation of mining operations may be a result of dynamic community processes, transition through successional stages, or an inability to reach a stable state as a result of unstable remnant sediments from mining operations (Barrio Froján et al. 2008). In laboratory experiments, benthic dwelling flatfishes (Johnson et al. 1998a) and crabs (Johnson et al. 1998b) persistently avoided sediments comprised of mine tailings.

Seabed alteration can fragment habitat, reduce habitat availability, and disrupt predator/prey interactions, resulting in negative impacts to fish and shellfish populations. Hitchcock and Bell (2004) studied physical impacts of an actively dredged shallow water, small scale mining operation that does not conduct onsite screening of mined materials and found significant physical impacts extending 300m downtide of the dredge area. Significant composition differences in sediment fractions were also identified within the excursion tract of the plume with zones of coarser materials extending 1500-2000m from the dredge location (Hitchcock and Bell 2004). Newell et al. (2004) studied the benthic community response to mining activities at the same dredge location and identified significant impacts to the benthic community structure within the dredge location when anchor dredging occurred with suppression of the benthic community extending up to 100m from the dredge site. Benthic community enhancement, possibly due to organic enrichment from dredge activities, was identified up to 2km in either direction of tidal streams extending from the dredge site (Newell et al. 2004). However, where less intensive mining occurred by trailer dredge, no significant impacts were identified (Newell et al. 2004). At an offshore shallow water aggregate mining operation, Despresz et al. (2009) found that benthic community structure dynamics in depositional locations are dominated by changes in the physical environment versus biological interactions with impacts to both substrate characteristics and benthic communities extended up to 2km from the dredge location.

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 Dalssen 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. Boyd and Rees (2003) found clear gradients of change in the benthic community related to both dredging intensity and the physical characteristics of the sediments. Differences in extraction methods also impact benthic community structure and sediment composition alterations. Suction hopper dredging creates dredge pits while trailer dredging creates elongated furrows (Birchenough et al. 2010). The excavated dredge pits quickly filling with fines and the benthic community becoming dominated by opportunistic species (Birchenough et al. 2010). Smith et al. (2006) also identified an increase of opportunistic mobile species at mining locations compared to reference, undisturbed sites.

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). As discussed previously, significant composition differences in sediment composition were identified in a shallow water, small scale mining operation extending 1500-2000m from the dredge location (Hitchcock and Bell 2004). Desprez et al. (2010) studied a shallow-water offshore mining operation and also found the sediment deposition from tidal stream plumes to extend up to 2km from the dredge site.

4.5.3.3 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). Mining operations may also lead to increased erosion in some areas. Long term sand mining operations off the coast of California ended in 1990 due to concerns of increased shoreline erosion impacts (Thornton et al. 2006). Following the cessation, erosion rates along the southern portion of the mining operations have decreased (Thornton et al. 2006).

4.5.4 Petroleum extraction

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). In coastal areas where extraction is prevalent, significant direct impacts (from spills) and secondary indirect impacts to coastal ecosystems (hydrological impacts, wetland loss, fault activation) are well documented (Ko and Day 2004). 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 section of this appendix.

4.5.5 Vessel disposal

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.

4.5.5.1 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 affect the marine environment.

4.6 Chemical effects of water discharge facilities

Water discharge from various sources can have high impacts related to chemical effects on both estuarine/nearshore and marine/offshore habitats.

Table 12 – Potential chemical effects of water discharge on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Industrial Discharge Facilities	Release of organic compounds (e.g. PCBs), <i>including</i>	√	√
	Release of chlorine compounds	√	√
	Release of petroleum products (PAH)	√	√
	Release of metals		√
	Release of pesticides	√	√
Sewage Discharge Facilities	Release of nutrients/eutrophication	√	√
	Release of contaminants	√	√
	Impacts to submerged aquatic vegetation	√	√
	Reduced dissolved oxygen	√	√
	Siltation, sedimentation, and turbidity	√	√
	Changes in species composition, <i>including:</i>	√	√
	Trophic level alterations	√	√
	Introduction of pathogens	√	√
	Introduction of harmful algal blooms	√	√
	Contaminant bioaccumulation and biomagnification	√	√
	Impacts to benthic habitat		√
	Behavioral responses	√	√
Combined Sewer Overflows	Potential for all of the above effects	√	√

Table 13 – Potential chemical effects of water discharge on marine/offshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Combined Sewer Overflows	Potential for all of the above effects	√	√
Industrial Discharge Facilities	Release of organic compounds (e.g. PCBs)		√
Sewage Discharge Facilities	Release of nutrients/eutrophication	√	√
	Release of contaminants	√	√
	Introduction of harmful algal blooms		√
	Contaminant bioaccumulation and biomagnification		√

4.6.1 Industrial discharge facilities (estuarine/nearshore and marine/offshore)

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).

4.6.1.1 Release of organic compounds (estuarine/nearshore only)

Industrial facilities that process animal or plant by-products can release effluent with high BOD which may have deleterious effects on 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

2002a). Approximately 96% of all bleached pulp production uses chlorine-free bleaching technologies (USEPA 2002a).

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.

4.6.1.2 Release of petroleum products (estuarine/nearshore only)

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 Mendelsohn 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).

4.6.1.3 Release of metals (estuarine/nearshore only)

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).

4.6.2 Sewage discharge facilities (estuarine/nearshore and marine/offshore)

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.

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 2002a; USEPA 2004b). In 2007, 127 beaches were issued advisories due to CSO discharges, with 46 of the affected beaches located on the Northeast coast (USEPA 2012).

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.

4.6.2.1 Release of nutrients and eutrophication (estuarine/nearshore and marine/offshore)

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 effects 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.

4.6.2.2 Release of contaminants (estuarine/nearshore and marine/offshore)

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). Studies on fish larvae response to wastewater discharge do not indicate that larvae actively avoid depositional zones for contaminants from plume waters. Fish larvae assemblages were shown to differ between control waters and within surface water sewage plumes, but at a depth of 20m beneath the surface no differences in assemblages was detected between the subsurface plume waters and control sites (Gray 1997).

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. 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 –CH₃ 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).

4.6.2.3 Impacts to submerged aquatic vegetation (estuarine/nearshore only)

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).

4.6.2.4 Reduced dissolved oxygen (estuarine/nearshore only)

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).

4.6.2.5 Siltation, sedimentation, and turbidity (estuarine/nearshore only)

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).

4.6.2.6 Changes in species composition (estuarine/nearshore only)

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).

4.6.2.7 Introduction of pathogens (estuarine/nearshore only)

Pathogens are generally a concern to human health because of consumption of contaminated shellfish and finfish and exposure at beaches and swimming areas (USEPA 2005d). 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 sections on Coastal Development and Introduced/Nuisance Species and Aquaculture for more information on the introduction of pathogens.

4.6.2.8 Introduction of harmful algal blooms (estuarine/nearshore and marine/offshore)

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 section on Introduced/Nuisance Species and Aquaculture for more information on harmful algal blooms.

4.6.2.9 Contaminant bioaccumulation and biomagnification (estuarine/nearshore and marine/offshore)

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 the USEPA rated the Northeast Coast with an overall good rating for sediment quality in 2012, sediment toxicity levels, elevated levels of metals, PCBs, and DDT, and TOC levels were primarily responsible for 12% of the coastal areas obtaining a poor sediment quality rating (USEPA 2012). 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 2012) 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 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.

4.6.2.10 Impacts to benthic habitat (estuarine/nearshore only)

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.

4.6.2.11 Behavioral responses (estuarine/nearshore only)

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.

4.6.3 Combined sewer overflow (CSO, estuarine/nearshore and marine/offshore)

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.

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 Industrial Discharge Facility and Sewage Discharge Facilities subsections of this chapter for additional information on this topic.

4.7 Physical effects of water intake and discharge facilities

Water intake and discharge facilities may have high impacts on estuarine/nearshore habitats associated with water intake.

Table 14 – Physical effects of water intake and discharge facilities on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Discharge Facilities	Alteration of salinity regimes	√	√
	Alteration of temperature regimes, <i>including:</i>	√	√
	Alteration of community structure	√	√
	Toxicity	√	√
	Attraction to flow, <i>including:</i>	√	√
	Physical/chemical synergies		√
	Restrictions to migration	√	√
	Mortality	√	
	Ballast water discharge		√
	Release of radioactive wastes	√	√
	Turbidity/sedimentation		√
	Alteration of sediment composition		√
	Reduced dissolved oxygen	√	√
	Habitat exclusion/avoidance	√	√
	Increased need for dredging	√	√
Intake Facilities	Entrainment/impingement	√*	√*
	Conversion/loss of habitat, <i>and</i>	√	√
	Alteration of community structure	√	√
	Ballast water uptake	√	√
	Alteration of hydrological regimes, <i>and</i>	√	√
	Flow restrictions	√	√
	Increased need for dredging	√	√

* = Water intake and discharge facilities are also highly likely to have entrainment and impingement effects in marine pelagic and benthic habitats: no other potential effects of these facilities were identified for marine habitats

4.7.1 Discharge facilities (estuarine/nearshore only)

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.

4.7.1.1 Alteration of salinity regimes (estuarine/nearshore only)

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).

4.7.1.2 Alteration of temperature regimes (estuarine/nearshore only)

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 2003b). 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).

4.7.1.3 Attraction to flow (estuarine/nearshore only)

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).

4.7.1.4 Ballast water discharges (estuarine/nearshore only)

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 sections on Introduced/Nuisance Species

and Aquaculture and Marine Transportation.

4.7.1.5 Release of radioactive wastes (estuarine/nearshore only)

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).

4.7.1.6 Turbidity and sedimentation effects (estuarine/nearshore only)

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).

4.7.1.7 Alteration of sediment composition (estuarine/nearshore only)

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 coarse 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.

4.7.1.8 Reduced dissolved oxygen (estuarine/nearshore only)

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.

4.7.1.9 Habitat conversion and exclusion (estuarine/nearshore only)

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.

4.7.1.10 Increased need for dredging (estuarine/nearshore only)

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 sections on Marine Transportation and Offshore Dredging and Disposal Activities.

4.7.2 Intake facilities (estuarine/nearshore only)

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.

4.7.2.1 Entrainment and impingement (estuarine/nearshore only)

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 m³/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 impingement 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). Another study in Britain estimated 5.66×10^7 fish were killed on cooling water intake screens during a two year monitoring study at Longannet Power Station with an estimated loss of 353.1 tons of whiting, cod, and plaice to the fishing industry (Greenwood 2008).

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).

4.7.2.2 Conversion/loss of habitat and alteration of community structure (estuarine/nearshore only)

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).

4.7.2.3 Ballast water and vessel operations intake (estuarine only/nearshore)

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 above, 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-m³ 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 sections on Energy-related Activities for additional information on vessel entrainment and impingement impacts.

4.7.2.4 Alteration of hydrological regimes and flow restrictions (estuarine/nearshore only)

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 (*Tautoglabrus 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).

4.7.2.5 Increased need for dredging (estuarine/nearshore only)

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 sections on Marine Transportation and Offshore Dredging and Disposal Activities.

4.8 Agriculture and silviculture

Agriculture and silviculture may have high impacts on estuarine/nearshore habitats.

Table 15 – Potential impacts of agriculture and silviculture on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Cropland, Rangelands, Livestock and Nursery Operations	Release of nutrients/eutrophication	√	√
	Siltation/sedimentation/turbidity	√	√
	Endocrine disruptors	√	√
	Bank/soil erosion		√
	Release of pesticides, herbicides, fungicides		√
	Loss/Alteration of wetlands/riparian zone		√
Silviculture and Timber Harvest Activities	Release of pesticides, herbicides, fungicides	√	√
	Release of nutrients/eutrophication	√	√
Timber and Paper Mill Processing Activities	Chemical contamination release	√	√

4.8.1 Croplands, rangelands, livestock, and nursery operation

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 2002b). 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 2002b). 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 2002b).

4.8.1.1 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 2002b). In addition, agricultural lands 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 into the Chesapeake Bay can be attributed to nonpoint sources, and agriculture accounted for the

majority of those (USEPA 2003c). Agriculture accounts for approximately 40% and 48% of nitrogen and phosphorus loads, respectively, to the Chesapeake Bay (USEPA 2003c). 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 2003b). 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 2003b). 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 2003b). 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 2003b). 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.

4.8.1.2 Siltation, sedimentation, and turbidity

As discussed below, 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 2003b; 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 2003c) and results in devastating impacts to shellfish and SAV. Wide-spread agricultural deforestation during the 18th and 19th 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 section, 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 2003b). Consequently, the impacts to aquatic organisms associated with siltation and sedimentation may be combined with the affects of pollution originating from the agricultural lands.

4.8.1.3 Endocrine disruptors

Studies have recently focused on a group of chemicals, called “endocrine disruptors,” that when present at extremely low concentrations 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.

4.8.1.4 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 2003b). 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 2003b). One of the consequences of erosional runoff from agricultural land is that it necessitates more frequent dredging of navigational channels (USEPA 2003b), 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 2003b).

4.8.1.5 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 2003b). Pesticides can be divided into four categories according to the target pest: insecticides, herbicides, fungicides, and nematicides (USEPA 2003b). 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 2003b). 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 2003b). 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 effects 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).

4.8.1.6 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 2003b). 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 2003b). 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.

4.8.2 Silviculture and timber harvest activities

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 2003c). 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.

4.8.2.1 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). Other possible effects to Atlantic salmon from pesticides may include altered chemical perception of home stream odor and osmoregulatory ability (USFWS and NMFS 1999).

4.8.2.2 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 2003c).

4.8.3 Timber and paper mill processing activities

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.

4.8.3.1 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 microsporidian 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 effects and effluent discharged by the pulp and paper mills (Khan 2004).

4.9 Introduced/nuisance species*

Introduced species may have high impacts on estuarine/nearshore habitats.

*Impacts from Aquaculture have been revised and included as Addendum I.

Table 16 – Potential impacts of introduced species on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	Pelagic	Benthic
Introduced/ Nuisance Species	Habitat alterations		√
	Trophic alterations		√
	Gene pool alterations	√	√
	Alterations to communities/comp. w/ native spp.	√	√
	Introduced diseases	√	√
	Changes in species diversity	√	√*

* = Introduced species can also highly affect species diversity in benthic marine habitats

4.9.1 Introduced/nuisance 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 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.

4.9.1.1 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 vegetation 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).

4.9.1.2 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).

4.9.1.3 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 section for a more detailed discussion on impacts from aquaculture operations.

4.9.1.4 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). Introduced species, like native species, have the potential to modify habitat value and influence community structure. Structure forming species increase habitat complexity which can increase species diversity and abundance. Structure forming introduced species were examined to evaluate the potential for increased species diversity in response to added habitat complexity, but only increases in mobile species diversity were identified, no differences in species abundance or richness for either mobile or sessile epifauna were identified (Sellheim et al. 2009).

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 Balleto 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).

Introduced species are common in bays and estuaries as dominant taxa in fouling communities (Tyrrell and Byers 2007, Ruiz et al. 2009). The relative abundance of native versus introduced species on artificial substrates compared to natural substrates was found to be higher for introduced species and may be a factor in the prevalence of introduced species in altered environments with a high prevalence of artificial substrate availability along coastal bays and estuaries (Tyrrell and Byers 2007, Ruiz et al. 2009).

4.9.1.5 *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 section for more information

regarding diseases introduced through aquaculture operations.

4.9.1.6 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 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 section for more information regarding diseases introduced through aquaculture operations.

4.10 Global effects and other impacts

Global effects may have high impacts on both estuarine/nearshore and marine/offshore environments.

Table 17 – Potential global effects on estuarine/nearshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Atmospheric Deposition	Mercury loading/bioaccumulation	√	√
	Nutrient loading/eutrophication	√	√
	PCB's and other contaminants	√	√
Climate Change	Alteration of hydrological regimes	√	√
	Alteration of temperature regimes	√	√
	Alteration of weather patterns	√	√
	Changes in community structure, <i>and</i>	√	√
	Changes in ecosystem structure	√	√
	Changes in dissolved oxygen concentrations	√	√
	Nutrient loading/eutrophication		√
	Release of contaminants		√
	Alteration in salinity	√	√
	Loss of wetlands	√	√
Military/Security Activities	Chemical releases		√
Natural Disasters and Events	Loss/alteration of habitat	√	√
	Impacts to water quality	√	√
	Changes in community composition		√

Table 18 – Potential global effects on marine/offshore habitats

IMPACT TYPE	POTENTIAL EFFECTS	P	B
Atmospheric Deposition	Mercury loading/bioaccumulation	√	
Climate Change	Alteration of hydrological regimes	√	
	Alteration of temperature regimes	√	√
	Alteration of weather patterns	√	
	Changes in community structure	√	√
Military/Security Activities	Noise impacts	√	
Ocean Noise	Mechanical injury to marine organisms	√	√

4.10.1 Atmospheric deposition (estuarine/nearshore and marine/offshore)

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.

4.10.1.1 Mercury loading/bioaccumulation (estuarine/nearshore and marine/offshore)

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 (*Scomberomorus 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.

4.10.1.2 Nutrient loading and eutrophication (estuarine/nearshore only)

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).

4.10.1.3 PCB and other contaminants (estuarine/nearshore only)

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 2005e).

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.

4.10.2 Climate change (estuarine/nearshore and marine/offshore impacts)

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 0.8°C since 1900, although this warming trend has increased to approximately 0.3°C 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 (Neddeau 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 precipitation 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 effects 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 appendix reflect the general topics identified by participants of the Technical Workshop on Impacts to Coastal Fishery Habitat from Non-fishing 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.

4.10.2.1 Alteration of hydrological regimes (estuarine/nearshore and marine/offshore)

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 precipitation 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 Niño-Southern Oscillation and the North Atlantic Oscillation/Northern 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 (Nedea 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 (Nedea 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 (Nedea 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 (Nedea 2004).

4.10.2.2 Alteration of temperature regimes (estuarine/nearshore and marine/offshore)

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 to 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 affect 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 30°C threshold of thermally-suitable habitat and practical limit of Atlantic cod distribution. The 26°C 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) (Nedea 2004).

4.10.2.3 Alteration of weather patterns (estuarine/nearshore and marine/offshore)

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.

4.10.2.4 Changes in community and ecosystem structure (estuarine/nearshore and marine/offshore)

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*) (Nedea 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 (Nedea 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.

4.10.2.5 Changes in dissolved oxygen concentrations (estuarine/nearshore only)

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).

4.10.2.6 Nutrient loading and eutrophication (estuarine/nearshore only)

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).

4.10.2.7 Release of contaminants (estuarine/nearshore only)

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).

4.10.2.8 Alteration of salinity regimes (estuarine/nearshore only)

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 (Nedea 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 effects 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.

4.10.2.9 Loss of wetlands and other fishery habitat (estuarine/nearshore only)

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 inundate 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.

4.10.3 Military/security activities (estuarine/nearshore and marine/offshore)

The operations of the U.S. 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 (Popper and Hastings 2009). Other sources of underwater noise from military activities may include explosive devices and ordnances during training exercises and during wartime. Refer to the summary of ocean noise effects in this section for more information on impacts associated with sonar, as well as the Marine Transportation section of the appendix for information related to blasting impacts.

4.10.4 Ocean noise (marine/offshore only)

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 conspecifics 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 conspecifics 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 anthropogenic generated 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, Popper and Hastings 2009).

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. 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 section 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 full effects of LFAS on bony fish and elasmobranchs 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).

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, Govoni et al. 2008). However, impacts to fish larvae vary by species. Some species larval stages have been found to be less sensitive to blasts than eggs or post-larval fish, but some species larval stages have been found to be just as sensitive as post-larval early juvenile stages, likely due to differences in larval physiology (Wright 1982, Govoni et al. 2008). Govoni et al. (2008) found differences in the sensitivity of two fish species larvae during blasting experiments. One species larvae was highly sensitive to blasting impacts with 100% mortality at a distance of 3.6 meters from the blasting event while the second species larvae were less sensitive with a range of mortality of 33% to 100% (Govoni et al. 2008). 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 to 500-1500 kHz), and most audiograms of fishes indicate a higher sensitivity to sound within the 0.100-2 kHz range (Stocker 2002, Popper and Hastings 2009). Evidence indicates that fish as a group have very complex and diverse relationships with sound and how they perceive it. Noise impacts to fish can generally be divided into four categories: (1) physiological; (2) acoustic; (3) behavioral; and (4) cumulative. The sensitivity of fish to noise impacts is variable by species and dependent on the source, duration, and environmental factors. Due to this variability, it is not possible at this time to extrapolate from existing research the effects of particular sounds to other sounds, to other effects, or to other species (Popper and Hastings 2009). Atlantic cod and haddock exposed to seismic shooting over a five day period varied in local abundance pattern responses, but both species exhibited avoidance behavioral responses (Engas et al. 1996). Both cod and haddock significantly decreased in abundance over the entire study area (18 nm from the location of shooting activity) from the time shooting was initiated through the end of the study (five days post-shooting) (Engas et al. 1996). The response of both species extended three times farther from the distance from the seismic shooting area than was anticipated based on prior research and knowledge of cod hearing and response thresholds (Engas et al. 1996). Physiological responses to similar range noises vary greatly between species. Atlantic cod exposed to 180 dB at varying frequencies (50-400Hz) for a duration of one to five hours exhibited destroyed ciliary bundles, while oscar exposed to the same decibels at 60 and 300 Hz frequencies did not exhibit physical damage until the termination of the study, four days post exposure, and then only oscar exposed at the 300Hz frequency for a continuous duration experienced physical damage (Enger 1981, Hastings et al. 1996). The research by Hasting et al. (1996) and McCauley et al. (2003) indicate that detection of sub-lethal physiological effects of noise in fish species may be delayed in excess of 58 days post exposure. The observed post exposure effects may have been a visual manifestation of a much greater undetected effect (Potter and Hastings 2009).

4.10.5 Natural disasters and events (estuarine/nearshore only)

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.

4.10.5.1 Loss/alteration of habitat (estuarine/nearshore only)

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.

4.10.5.2 Water quality impacts (estuarine/nearshore only)

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.

4.10.5.3 Changes to community composition (estuarine/nearshore only)

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.

4.11 Aquaculture

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http://coastalscience.noaa.gov/research/scem/marine_aquaculture Environmental Sustainability Program for providing guidance and policy recommendations for this document. For more information on this program, see

http://coastalscience.noaa.gov/research/scem/marine_aquaculture.

Table 199 - Potential impacts of aquaculture on estuarine/nearshore habitats.

IMPACT TYPE	POTENTIAL EFFECTS	Pelagic	Benthic
Aquaculture	Discharge of organic waste and contaminants		√
	Seafloor impacts		√
	Introduction exotic invasive species	√	√
	Food web impacts	√	√
	Gene pool alterations		√
	Impacts to water quality, <i>and</i>	√	√
	Impacts to water column	√	
	Changes in species diversity	√	√
	Introduction of diseases		√
	Habitat conversion, <i>and</i>	√	√
	Habitat replacement/exclusion		√
	Sediment deposition		√

For the purposes of policy development, aquaculture is defined as the propagation and rearing of aquatic marine organisms for commercial, recreational, or public purposes. This definition covers all authorized production of marine finfish, shellfish, plants, algae, and other aquatic organisms for 1) food and other commercial products; 2) wild stock replenishment and enhancement for commercial and recreational fisheries; 3) rebuilding populations of threatened or endangered species under species recovery and conservation plans; and 4) restoration and conservation of aquatic habitat (DOC Aquaculture Policy 2011; NOAA Aquaculture Policy 2011). This guidance addresses concerns related to the production of seafood and other non-seafood related products (e.g., biofuels, ornamentals, bait and pharmaceuticals) by aquaculture. The findings assess potential impacts, negative and positive, to EFH and EFH- HAPCs posed by activities related to marine aquaculture in offshore and coastal waters, riverine systems and adjacent wetland habitats, and the processes that could improve or place those resources at risk.

Overview of Marine Aquaculture and EFH Interactions

The environmental effects of marine aquaculture can vary widely depending on the species selected for culture, the location and scale of the aquaculture operation, the experience level of the operators, and the production methods. The use of modern production technologies, proper siting protocols, standardized operating procedures, and best management practices (BMPs) can help reduce or eliminate the risk of environmental degradation from aquaculture activities. In recent years, marine aquaculture has been used to bolster EFH (e.g., oyster cultch planting to

rebuild oyster reefs) and in some instances, aquaculture has been used to mitigate eutrophication by sequestering nutrients in coastal waters (e.g., shellfish and algae culture).

The following summary provides information on the types of environmental effects resulting from marine aquaculture activities that have been documented and includes references to various BMPs and other existing regulatory frameworks used to safeguard coastal resources. This summary is not an exhaustive literature review of scientific information on this complex topic, rather it is a synthesis of relevant information intended to provide managers with a better understanding of the environmental impacts of marine aquaculture.

4.11.1 Escapement

Unintentional introductions and accidental releases of cultured organisms may have wide ranging positive or negative effects on EFH. Ecological damage caused by organisms that have escaped or been displaced, in the case of shellfish or algae, from aquaculture may occur in riverine, estuarine, and marine habitats (Waples et al. 2012). The potential for adverse effects on the biological and physical properties of EFH include: (1) introduction of invasive species, (2) habitat alteration, (3) trophic alteration, (4) gene pool alteration, (5) spatial alteration, and (6) introduction of pathogens and parasites that cause disease. The use of local, native species can result in little to no impacts on EFH in the event that escapement does occur.

Aquaculture is recognized as a pathway for both purposeful and inadvertent introduction of non-native species in aquatic ecosystems. Most introduced species do not become invasive; however, naturalization of introduced non-native species that results in invasion and competition with native fauna and flora has emerged as one of the major threats to natural biodiversity (Wilcove et al. 1998; Bax et al. 2001; D'Antonio et al. 2001; Olenin et al. 2007). Some non-native species alter the physical characteristics of coastal habitats and constitute a force of change affecting population, community, and ecosystem processes (Grosholz 2002).

Even through use of native species, escapees have the potential to alter community structure, disrupt important ecosystem processes, and affect biodiversity. Environmental impacts are augmented by competition for food and space, introduction or spread of pathogens, and breeding or interbreeding with wild populations. Excessive colonization by shellfish or other sessile organisms may lead to alterations of physical habitat and preclude the growth of less abundant species with ecological significance. Similarly, escapees that colonize specific habitats and exhibit territorial behavior may compete with and displace local species to segregated habitats.

Culture of native species presents genetic risk from escapees interbreeding with individuals in the wild. The magnitude of the genetic impact on the fitness of wild stock is somewhat unclear. Genetic introgression of cultured escapees into wild populations is strongly density- dependent and appears linked to the population size and health of native populations relative to the magnitude of the escapes. To make a genetic impact, escapees must survive and reproduce successfully in the wild and contribute offspring with sufficient reproductive fitness to contribute to the gene pool. The capability of escaped fish to do so can vary widely based on a multitude of environmental and biological factors (e.g., predation, competition, disease). In general, fitness of captive-reared individuals in the wild decreases with domestication (i.e., the

number of generations in captivity). Some genetic risks are inversely correlated, such that reducing one risk simultaneously increases another. For example, creating an aquaculture population that is genetically divergent from the wild stock may reduce the chances that escapees can survive and reproduce. Still, under this scenario aquacultured organisms that do survive could potentially pass on maladapted genes to the wild population.

The likelihood of escapes from aquaculture operations will vary depending on the species being cultured, siting guidelines, structural engineering and operational design, management practices (including probability for human error), frequency of extreme weather events, and direct interactions with predators such as sharks, marine mammals, and birds. While a certain level of escapes may not be avoidable in all cases, risk assessments should be used to make informed regulatory decisions in an effort to account for potential impacts on EFH. Risk assessment tools are available and have been used to identify and evaluate risks of farmed escapes on wild populations (Waples et al. 2012). Many empirical models have been used to inform policy (ICF 2012; RIST 2009), and are readily available for use in permitting and project planning.

Good practices for monitoring, surveillance, and maintenance of the aquaculture operation are critical to preventing the possibility of escapes. An escape prevention and mitigation plan should be developed for each farm. Plans should contain a rationale for approaches taken and any recapture or mitigation activities that should be initiated when an escape occurs.

4.11.2 Disease in aquaculture

As with all animal production systems, disease is a considerable risk for production, development, and expansion of the aquaculture industry. The industry has experienced diseases caused by both infectious (bacteria, virus, fungi, parasites) and non-infectious (nutritional, environmental, pollution, stress) agents. In addition to mortality and morbidity, disease causes reduced market value, growth performance, and feed conversion. An accredited health professional should regularly inspect crops and perform detailed diagnostic procedures to determine if disease presents a risk. Veterinarians with expertise in fish culture, or qualified aquatic animal health experts, can assist with development of a biosecurity plan to prevent or control the spread of pathogens within a farm site, between aquaculture operations, or to wild populations.

The spread of pathogens from cultured organisms to wild populations is a risk to fisheries and EFH conservation. There are documented cases of mortality in wild populations caused by both endemic and exotic diseases (NAAHP 2008). The prevalence of disease in intensive aquaculture operations is influenced by many factors, including immune status, stress level, pathogen load, environmental conditions, nutritional health, and feeding management. The type and level of husbandry practices and disease surveillance will also influence the potential spread of pathogens to wild stocks. International trade in live fish and shellfish has led to the introduction of diseases to new areas. Once a pathogen or disease is introduced and becomes established in the natural environment, there is little possibility of eradication. However, increased awareness of disease risks, health control legislation, and better diagnostic methods, which have increased the ability to detect diseases and pathogens, are helping to reduce the frequency of introduction and the spread

of diseases (NAAHP 2008).

In some cases, the expansion and diversification of the marine aquaculture industry has resulted in parasite translocations (Shumway 2011). Because of this, many countries and regions have created compacts and agreements to include pathogen screening guidelines and certification programs for movement of germplasm, embryos, larvae, juveniles, and broodstock associated with marine aquaculture operations. In the United States, import and export certifications and testing for certain types of diseases falls under the jurisdiction of the USDA Animal and Plant and Health Inspection Service (APHIS). Most states have specific protocols that must be followed when transplanting cultured species into wild environments to minimize the incidence of disease transfer. In the case of aquaculture operations in federal waters, the Gulf of Mexico Fishery Management Council specified in their Fishery Management Plan for Regulating Offshore Marine Aquaculture that prior to stocking animals in an aquaculture system in federal waters of the Gulf, the permittee must provide NOAA Fisheries a copy of a health certificate signed by an aquatic animal health expert certifying cultured animals were inspected and determined to be free of World Organization of Animal Health reportable pathogens (OIE 2003,) or additional pathogens that are identified as reportable pathogens in the National Aquatic Animal Health Plan (GMFMC 2012).

Climate change has been implicated in increasing the prevalence and severity of infectious pathogens that may cause disease originating from cultured or transplanted aquaculture stocks (Hoegu-Guldberg and Bruno 2010). The emergence of these diseases is likely a consequence of several factors, including shifting of pathogen ranges in response to warming, changes to host susceptibility as a result of increasing environmental stress, and the expansion of potential vectors. Classical examples are outbreaks of oysters infected with MSX (*Haplosporidium nelsoni*), Dermo (*Perkinsus marinus*), and *Bonamia* spp. (Ford and Smolowitz 2007, Soniat et al. 2009, Shumway 2011). In most cases, pathogens have undergone rapid ecological and genetic adaptation in response to climate change. Guidelines for management of these diseases are well-developed for shellfish and other aquatic species. Managing for disease outbreaks is a key aspect of climate adaptation to prevent adverse impact to EFH. Management guidelines include record keeping and strict regulations on stocking or transplanting species from infected areas. Following these management recommendations should yield protection and conservation benefits for EFH.

4.11.3 Use of drugs, biologics, and other chemicals

Disease control by prevention is preferable to prophylactic measures and curative medical treatment. Aquaculture drugs, biologics, and other chemicals play an important role in the integrated management of aquatic animal health. Aquaculture operations in the United States use these products for: (1) disinfectants as part of biosecurity protocols, (2) herbicides and pesticides used in pond maintenance, (3) spawning aids, (4) vaccines used in disease prevention, or (5) marking agents used in resource management (AFS 2011). Despite the best efforts of aquaculture producers to avoid pathogen introductions, therapeutic drugs are occasionally needed to control mortality, infestations, or infections. The availability and use of legally approved pharmaceutical drugs, biologics and other chemicals is quite limited in marine aquaculture (FDA 2012).

While antibiotics are a commonly cited chemical therapeutant, the use of antibiotics in U.S.

aquaculture is not common and strictly limited, and global use in aquaculture of antibiotics has declined in recent years, up to 95% in the culture of salmon and other species, largely attributed to improved husbandry and use of vaccines (Asche and Bjørndal 2011; Forster 2010; Rico et al. 2012). Antibiotics are characterized by low toxicity to vertebrates. The environmental risks of antibiotic use are minimal, especially with regards to impacts to fisheries and EFH. The transference of antimicrobial drug resistance among marine fish and shellfish is theoretically possible yet an unproven concern. In a comprehensive review of the salmon aquaculture industry, no direct evidence of negative impact to wild fish health resulting from antibiotic use in salmon farming has been found (BurrIDGE et al. 2010). With farms that use medicated feeds, some antibiotic compounds can persist in sediments around fish farms and therefore affect the microbial community. Laboratory and field studies have found that antibiotic persistence in sediment ranges from a few days to years depending on the drug in question and the geophysical properties of the water or sediment (Scott 2004, Armstrong et al. 2005, Rigos and Troisi 2005). A limited number of broad spectrum antibiotics and feed additives (*i.e.*, florfenicol and oxytetracycline) are allowed as part of the National Investigational New Animal Drug Program, which is regulated by FDA and managed through partnership with the U.S. Fish and Wildlife Service. Antibiotics like other medicines should be used sparingly with prescription and in accordance with approved protocol to minimize environmental interactions.

Cultured fish are susceptible to parasitic diseases. Sea lice are natural ectoparasites of marine fish and the most prevalent parasites of cultured marine finfish. Effective mitigation, management, and control of parasitic infestations requires good husbandry. Chemicals used in the treatment of most parasitic infestations with netpen operations are subsequently released to the aquatic environment. These compounds have varying degrees of environmental impact, but many are lethal to non-targeted aquatic invertebrates. Research suggests that environmental impacts from parasiticide treatments are minor and restricted to the spatiotemporal scale of infestation and treatment (BurrIDGE et al. 2010). The use of large quantities of drugs and chemicals for parasite control has the potential to be detrimental to fish health and EFH. Excessive use of paraciticides is of concern to the aquaculture industry and its regulators.

The most common biologics used for aquatic organisms are vaccines. A vaccine is any biologically based preparation intended to establish or improve immunity to a particular disease or group of diseases. Vaccines have been used for many years in humans and agricultural livestock. They are considered the safest prophylactic approach to management of aquatic animal health and pose no risk to the environment or EFH. In aquaculture, the use of vaccines for disease prevention has expanded both with regard to the number of aquatic species and number of microbial diseases. Vaccination has become a basis for good health for most finfish operations. Commercial vaccines can be administered by injection or immersion. Oral vaccines remain experimental. Vaccines have been successfully used to prevent a variety of bacterial diseases in finfish. Few viral vaccines are commercially available and vaccines for fungal and parasite diseases do not exist. The efficacy and safety of a vaccine is species specific and requires detailed knowledge of pathogenesis of the disease, antigens for protection, and immune response. All vaccines for use on fish destined for human consumption must be approved by the USDA APHIS, the federal agency responsible for regulating all veterinary biologics, including vaccines, bacterins, antisera, and other products of biological origin.

4.11.4 Water quality impacts

Water quality is a key factor in any aquaculture operation, affecting both success and environmental sustainability. Aquaculture operations should be sited in areas with an abundant and reliable supply of good water quality. The primary risks to water quality from marine aquaculture operations are increased organic loading and nutrient enrichment. Excess nutrients, organic matter, and suspended solids in finfish aquaculture effluents can cause eutrophication in receiving water bodies when nutrient inputs exceed the capacity of natural dispersal and assimilative processes. Elevated nutrients and declines in dissolved oxygen are sometimes observed following feeding high-density operations. These conditions rarely persist or present long-term risk to water quality.

At some farm sites, a phytoplankton response to nutrient loading has been reported, but generally this is a low risk and causal linkages to algal blooms are not evident. Because a change in primary productivity linked to fish farm effluents would have to be detected against the background of natural variability, it is difficult to discern effects unless they are of great magnitude and duration. At large scales, the occurrence of many anthropogenically derived nutrients in coastal marine waters makes it difficult to attribute increased primary productivity directly to aquaculture.

Environmental impacts will vary by location (*i.e.*, on-shore, near-shore, and offshore); therefore, careful selection of sites is the most important tool for risk management. Operations appropriately sited in well-flushed, non-depositional areas may have little to no impact on water quality. The approach to limiting impacts to water quality will also vary by production format. For example, closed systems located onshore are able to directly control their discharges while production systems located offshore rely on best management practices, including siting aquaculture operations outside of nutrient sensitive habitats (*e.g.*, EHF), responsible cleaning practices, integration of feed management strategies, use of optimally formulated diets, and other management measures to minimize nutrient discharge.

Aquaculture operations are regulated under the Clean Water Act, by the National Pollutant Discharge Elimination System (NPDES), a permitting system administered by the EPA for wastewater discharges into navigable waters.¹ NPDES permits contain industry-specific, technology-based, and water-quality-based limits, and establish pollutant monitoring and reporting requirements.² Aquaculture operations that qualify as concentrated aquatic animal production facilities (*i.e.*, produce more than 45,454 harvest weight kilograms of fish and feed) must obtain a permit before discharging wastes. A permit applicant must provide quantitative analytical data identifying the types of pollutants present in wastewater effluents. The permit will set forth the conditions and effluent limitations under which an aquaculture operation may make a discharge. NPDES permit limitations are based on best professional judgment when national effluent limitations guidelines have not been issued pertaining to an industrial category

¹ Pursuant to the provisions of Section 402(a)(1); 40 CFR 122.44(k) of the Federal Water Pollution Control Act (Clean Water Act).

² EPA issues effluent guidelines for categories of existing sources and sources under Title III of the Clean Water Act. The standards are technology-based (*i.e.*, they are based on the performance of treatment and control technologies); they are not based on risk or impacts upon receiving waters.

or process.

4.11.5 Benthic sediment and community impacts

Benthic impacts can result from deposition of organic wastes from aquaculture operations. These impacts can affect EFH if aquaculture operations are not properly sited. Excess feed and feces are the predominant sources of particulate wastes from fish farms. Shellfish operations release pseudofeces, a byproduct of mollusks filtering food from the water column. If allowed to accumulate, particulate waste products may alter biogeochemical processes of decomposition and nutrient assimilation. At sites with poor circulation, waste accumulation can alter the bottom sediment and perturbate infaunal communities if wastes are released in excess of the aerobic assimilative capacity of the bottom. Under such conditions, sediments will turn anoxic and the benthic community will decline in species diversity. Benthic impacts are generally localized and ephemeral in nature.

Common indicators used to assess benthic condition include total organic carbon, redox potential, total sulfides, and abundance and diversity of marine life. Electro-chemical and image analysis methods are used to quantify video-recorded observations of benthic condition. These indicators guide BMPs for grading and stocking fish, fallowing, or adjusting feed rates. Fallowing is the practice of temporarily relocating or suspending aquaculture operations to allow the benthic community and sediments to undergo natural recovery from the impacts of nutrient loading. Under ideal conditions, farms should not require a fallowing period for the purpose of sediment recovery; however, this practice is widely and successfully implemented around the world as a management practice for preventing damage to the benthic environment and EFH (Tucker and Hargreaves 2008). Fallowing times range from a few months to several years depending on local hydrology, circulation at a site, and the level of accumulation (Brooks et al. 2003, Brooks et al. 2004).

Benthic accumulation of organic wastes can be reduced by siting aquaculture operations in well-flushed areas, or in areas where net erosional sediments can decrease or eliminate accumulation of wastes, thereby minimizing benthic effects. In some cases, moderate discharge has been shown to enhance local productivity of marine species including algae and fish (Machias et al. 2004; Dempster et al. 2006; Wang et al. 2012). Benthic monitoring plans should be designed to allow for early detection of enrichment and deterioration of benthic community structure. Additionally, nearby control sites should be established in order to collect data to differentiate between aquaculture effects and natural and seasonal variability, or non-aquaculture factors.

4.11.6 Location specific interactions with EFH

Onshore Aquaculture

Onshore aquaculture activities occur on-land in ponds, raceways, and tank-based systems. These systems can be used for multiple phases of aquaculture including broodstock holding, hatchery production, nursery production, grow-out, and quarantine. Water demand and usage varies from conventional pond systems to intensive recirculating aquaculture systems, which may employ sophisticated filtration components for water reuse. Onshore marine aquaculture operations have the potential to impact a variety of EFHs including:

- a) waters and benthic habitats in or near marine aquaculture sites;
- b) exposed hardbottom in shallow and deep waters;
- c) submerged aquatic vegetation beds;
- d) shellfish beds;
- e) spawning and nursery areas;
- f) coastal wetlands, and
- g) riverine systems and associated wetlands.

The greatest impacts to EFH by onshore aquaculture involve escape of non-native species and nutrient discharge and its impact on water quality and bottom sediments. Onshore aquaculture activities affecting EFH are regulated by existing state and federal laws and requirements specified by EPA's National Pollutant Discharge Elimination System and coastal habitat protection plans.

Nearshore aquaculture

Nearshore aquaculture activities are those that occur in rivers, sounds, estuaries and other areas that extend through the coastal zone.³ Currently, all aquaculture in the Northeast region is in nearshore areas. The dockside value is split between shellfish and finfish. The two most common cultured species are the American oyster (*Crassostrea virginica*) and the hard clam (*Mercenaria mercenaria*), with oysters being the valuable. Finfish culture is currently limited to Atlantic salmon.

While the relative risk of nearshore shellfish aquaculture to various EFHs is uncertain, the ranges of possible interactions include:

- a) marine and estuarine waters;
- b) estuarine wetlands, including mangroves and marshes;
- c) submerged aquatic vegetation;
- d) waters that support diadromous fishes, and their spawning and nursery habitats, and
- e) waters hydrologically and ecologically connected to waters that support EFH.

The environmental effects of shellfish and finfish aquaculture in coastal waters are well-documented (Naylor et al. 2006; Nash 2005; Tucker and Hargreaves 2008). Poorly sited and managed aquaculture activities can have significant impact on benthic communities, water quality, and associated marine life. While there are case studies documenting environmental impacts of practices used several decades ago, regulatory and management practices are reducing the likelihood of negative environmental effects (Price and Morris 2013).

In the case of cage culture, water quality and benthic effects are sometimes observed; however, these are typically episodic and restricted to within 30 m of the cages (Nash 2003). Long-term risks to water quality from offshore aquaculture activities are unlikely when operations are sited

³ The term "coastal zone" means the coastal waters strongly influenced by each other and in proximity to the shorelines of several coastal states, and includes islands, transitional and intertidal areas, salt marshes, wetlands, and beaches. The zone extends seaward to the outer limit of State title and ownership under the Submerged Lands Act (43 U.S.C. 1301 et seq.).

in well-flushed waters. Belle and Nash (2008) recommend the siting of cages in water at least twice as deep as the cage with minimum flows of 7cm/second. It is not common for increases in chlorophyll or algal production to be measureable near aquaculture operations, especially in well flushed areas. Therefore, algal blooms are not expected to result from nutrient enrichment from fish aquaculture operations where properly sited.

The most studied benefit from marine aquaculture operations is as fish attractants as wild fish use aquaculture cages for shelter, foraging on biofouling organisms, and consumption of uneaten feed. Wild fish can help distribute organic waste away from the cages and re-suspend organic compounds in sediments. As a result, overall fish abundance may increase in areas with aquaculture operations. Recreational and commercial fishers may benefit from increased fishing opportunities around marine aquaculture operations. Conversely, interactions with marine mammals that are attracted to the forage fish around cages are identified as potential long-term concern for management of protected species.

Moderate nutrient loads discharged from aquaculture operations can also increase productivity of some marine environments. This is especially true in waters with low levels of nitrogen and phosphorus, where nutrients are quickly assimilated into the food web. The actual environmental interactions of these nutrient loads are difficult to study due to the high rate of nutrient flushing and assimilation by phytoplankton.

Potential interactions of nearshore shellfish aquaculture with EFH are changes to benthic habitat as a result of pseudofeces, the effects of mechanical harvesting, conversion of soft sediment habitat to hard bottom shellfish reef, displacement of cultured organisms, potential genetic transfer, sedimentation and loading of organic waste to the water column and benthic sediments, and disruption of the benthic community. Some changes could potentially impact SAV located near shellfish aquaculture operations, although this impact likely varies with species and production type.

In general, shellfish and algae aquaculture has positive impacts on EFH, providing ecosystem services and habitat related benefits in the estuary including mitigation of land-based nutrients and increased habitat for fish, shellfish, and crustaceans (Shumway 2011). Therefore, the positive and negative effects of shellfish culture activities to EFH need to be considered. The risk of nearshore aquaculture impacts to EFH can be minimized by including terms and conditions designed to protect sensitive habitats in permits issued under state and federal laws and regulations. Best management practices are now in place for shellfish aquaculture along the U.S. East Coast (Flimlin 2010).

Offshore aquaculture

Offshore aquaculture activities occur in areas of the open ocean that extend from the seaward edge of the coastal zone through the exclusive economic zone.⁴ In the region, offshore aquaculture may include the cultivation of macrophytic algae, molluscan shellfish or finfish. Currently there are applications for two offshore mussel farms in the region. It is feasible that

⁴ The term 'offshore aquaculture' is often used to refer to aquaculture in waters under federal jurisdiction, which typically extend from 3-200 nautical miles from the shoreline.

co-siting aquaculture facilities with other offshore industries such as wind energy could facilitate offshore aquaculture development. Over twenty-five laws exist to provide regulatory oversight of aquaculture in federal waters. Some examples include the Clean Water Act and the Coastal Zone Management Act.

While the relative threat of offshore aquaculture to EFHs varies widely depending on siting and management considerations, the ranges of possible interactions include:

- a) marine and estuarine waters;
- b) waters that support diadromous fishes, and their spawning and nursery habitats, and
- c) waters hydrologically and ecologically connected to waters that support EFH.

The environmental effects of offshore shellfish and finfish aquaculture are not well-documented because few operations exist in the United States. The information gleaned from coastal production sites, especially those with conditions similar to federal waters, provides some indications as to the potential effects of offshore aquaculture (see section on near shore aquaculture).

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Appendix G: Non-fishing impacts to habitat

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