CHAPTER SIX: OFFSHORE DREDGING AND DISPOSAL ACTIVITIES

Introduction

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

Offshore Mineral Mining

Introduction

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

Mineral extraction is usually conducted with hydraulic dredges by vacuuming or, in some cases, by mechanical dredging with clamshell buckets in shallow water mining sites. Mechanical dredges can have a more severe but localized impact on the seabed and benthic biota, whereas hydraulic dredges may result in less intense but more widespread impact (Pearce 1994). The impacts of offshore mineral mining on living marine resources and their habitats include: (1) the removal of substrates that serve as habitat for fish and invertebrates; (2) creation of (or conversion to) less productive or uninhabitable sites such as anoxic depressions or highly hydrated clay/silt substrates; (3) release of harmful or toxic materials either in association with actual mining, or from incidental or accidental releases from machinery and materials used for mining; (4) burial of productive habitats during beach nourishment or other shoreline stabilization activities; (5) creation of harmful suspended sediment levels; and (6) modification of hydrologic conditions causing adverse impacts to desirable habitats (Pearce 1994; Wilber et al. 2003).

In addition, mineral extraction can potentially have secondary and indirect adverse effects on fishery habitat at the mining site and surrounding areas. These impacts may include accidental or intentional discharges of mining equipment and processing wastes and degradation or elimination of marine habitats from structures constructed to process or transport mined materials. These secondary effects can sometimes exceed the initial, direct consequences of the offshore mining.

Loss of benthic habitat types

Offshore benthic habitats occurring on or over target aggregates may be adversely affected by mining. The mineral extraction process can disrupt or eliminate existing biological communities within the mining or borrow areas for several years following the excavation. Filling in of the borrow areas and reestablishment of a stable sediment structure is dependent upon the ability of bottom currents to transport similar sediments from surrounding areas to the mining site (ICES 1992). The principal concern noted by the International Council for the Exploration of the Sea (ICES) Working Group on the Effects of Extraction of Marine Sediments on Fisheries was dredging in spawning areas of commercial fish species (ICES 1992). Of particular concern to the ICES Working Group are fishery resources with demersal eggs (e.g., Atlantic herring [*Clupea harengus*] and sand lance [*Ammodytes marinus*]). They report that when aggregates are removed, Atlantic herring eggs are taken with them, resulting in lost production to the stock. Stewart and Arnold (1994) list the impacts on Atlantic herring from offshore mining to include the entrainment of eggs, larvae, and adults; burial of eggs; and effects of the turbidity plume on demersal egg masses. Gravel and coarse sand have been identified as preferred substrate for Atlantic herring eggs on Georges Bank and in coastal waters of the Gulf of Maine (Stevenson and Scott 2005).

Conversion of substrate/habitat and changes in community structure

Disposal of residues ("tailings") of the mining process can alter the type, as well as the functions and values, of habitats which can then alter the survival and growth of marine organisms. The tailings are often fine-grained and highly hydrated, making them very dissimilar to the natural seafloor, particularly in depths where wave energy and currents are capable of winnowing or sorting sediments and relocating them to depositional areas. It has been found that wave forces are affecting habitats in the New York Bight at depths in excess of 22 m (USACE 2005a). In laboratory experiments, benthic dwelling flatfishes (Johnson et al. 1998a) and crabs (Johnson et al. 1998b) persistently avoided sediments comprised of mine tailings.

Additionally, there can be adverse impacts from aggregate and/or mineral mining on nearby habitats associated with the removal and disturbance of substrate (Scarrat 1987). Seabed alteration can fragment habitat, reduce habitat availability, and disrupt predator/prey interactions, resulting in negative impacts to fish and shellfish populations. Not all offshore aggregate mining results in adverse impacts on seabed resources. Hitchcock and Bell (2004) conducted a detailed study of the effects from a small-scale, aggregate mining operation off the south coast of the United Kingdom and found physical impacts on the seabed to be limited to a downtide zone approximately 300 m from the dredge area. Related studies at this mining operation reported no detectable impact on the surrounding benthic communities, despite a small change in seabed particle size distribution (Hitchcock and Bell 2004).

Long-term mining can alter the habitat to such a degree that recovery may be extremely protracted and create habitat of limited value to benthic communities during the entire recovery period (van Dalfsen et al. 2000). For example, construction grade aggregate removal in Long Island Sound, Raritan Bay (lower New York Harbor) and the New Jersey portion of the intercoastal waterway have left borrow pits that are more than twice the depth of the surrounding area. The pits have remained chemically, physically, and biologically unstable with limited diversity communities for more than five decades. These pits were used to provide fill material for interstate transportation projects and have been investigated to assess their environmental impact (Pacheco 1984). Borrow pits in Raritan Bay were found to possess depressed benthic communities and elevated levels of highly hydrated and organically enriched sediments (Pacheco 1984). In one example, aggregate mining operations from the 1950s through the 1970s created a 20 m deep borrow pit in an area of Raritan Bay that, although the mining company was required to refill the pit, remains today as a rapid deposition area filling with fine-grained sediment and organic material emanating from the Hudson River and adjacent continental shelf (Pacheco 1984). The highly hydrated sediments filling the depressions are of limited utility to colonizing benthic organisms.

In offshore mining operation sites, the character of the sediment which is exposed or subsequently accumulates at the extraction site is important in predicting the composition of the colonizing benthic community (ICES 1992). If the composition and topography of the extraction site resembles that which originally existed, then colonization of it by the same benthic fauna is likely (ICES 1992).

Changes in sediment composition

A review of studies conducted in Europe and Great Britain found that infilling and subsequent benthic recovery of borrow areas may take from 1-15 years, depending upon the tide and current strength, sediment characteristics, the stock of colonizing species and their immigration distance (ICES 1992). Typically the reestablishment of the community appears to follow a successional process similar to those on abandoned farmlands. Germano et al. (1994) described this process, reporting that pioneering species (i.e., Stage I colonizers) usually do not select any particular habitat but attempt to survive regardless of where they settle. These species are typically filter feeders relying on the availability of food in the overlying water rather than the seafloor on which they reside. Thus, their relationship to the substrate is somewhat tenuous, and their presence is often ephemeral. However, their presence tends to provide some stability to the seafloor, facilitating subsequent immigrations by other species that bioturbate the sediment seeking food and shelter. Their arrival induces further substrate consolidation and compaction. These colonizers are usually deemed to be Stage II community species. The habitat modification activities of Stage I and II species advance substrate stability and consolidation enough for it to support, both physically and nutritionally, the largest community members (i.e., Stage III). The benthic community instability caused by dredging gives rise to one of the principal justifications for retaining benthic disturbances: the disrupted site may become heavily populated by opportunistic (i.e., Stage I) colonizer species that flourish briefly and provide motile species with an abundance of food during late summer and fall periods (Kenny and Rees 1996). However, if environmental stresses are chronic, the expected climax community may never be attained (Germano et al. 1994).

If the borrow area fails to refill with sediment similar to that which was present prior to mining, the disturbed area may not possess the original physical and chemical conditions and recovery of the community structure may be restricted or fail to become reestablished. Dredge pits that have been excavated to depths much greater than the surrounding bottom often have very slow infill rates and can be a sink for sediments finer than those of the surrounding substrate (ICES 1992).

Changes in bottom topography and hydrology

The combination of rapid deposition, anomalous sediment character, and an uneven topography, as compared to the surrounding seafloor, limit recolonization opportunities for harvesting purposes (Wilk and Barr 1994). By altering bottom topography, aggregate mining can reduce localized current strength, resulting in lowered dissolved oxygen concentrations and increased accumulation of fine sediments inside borrow pits (ICES 1992). One potential benefit of some borrow pits is that they appear to provide refugia for pelagic species such as alewife (*Alosa pseudoharengus*) and scup (*Stenotomus chrysops*), as well as demersal species such as tautog (*Tautoga onitis*) and black sea bass (*Centropristis striata*) during seasonally fluctuating water temperatures (Pacheco 1984). However, it is doubtful these benefits outweigh the persistent adverse affects associated with borrow pits (Palermo et al. 1998; Burlas et al. 2001). Other consequences of aggregate mining may include alteration of wave and tidal current patterns which could affect coastal erosion (ICES 1992).

Siltation, sedimentation, and turbidity

Offshore mining can increase the suspended sediment load in the water column, increasing turbidity that can then adversely affect marine organisms, particularly less motile organisms such as shellfish, tunicates, and sponges. The duration of the turbidity plume in the water column depends upon the water temperature, salinity, current speed, and the size range of the suspended particles (ICES 1992). The distance the dredged material is transported from the excavation site will be dependent upon the current strength, storm resuspension, water salinity and temperature, and the grain size of the suspended material (ICES 1992).

The life stages of the affected taxa are an important factor affecting the type and extent of the adverse impacts (Wilber and Clarke 2001). As a general rule, the severity of sedimentation and turbidity effects tends to be greatest for early life stages and for adults of some highly sensitive species (Newcombe and Jensen 1996; Wilber and Clarke 2001). In particular, the eggs and larvae of nonsalmonid estuarine fishes exhibit some of the most sensitive responses to suspended sediment exposures of all the taxa and life history stages for which data are available (Wilber and Clarke 2001). Stewart and Arnold (1994) list the impacts on Atlantic herring from offshore mining to include the effects of the turbidity plume on demersal egg masses.

Impacts to water quality

The release of material into the water column during offshore mining operations can degrade water quality if the excavated material is high in organic content or clay. The effects of mixing on the water column are likely to include increased consumption of oxygen by decomposing organic matter and the release of nutrients (ICES 1992). However, mined aggregate material is typically low in organic content and clay, and any increase in the biological oxygen demand is thought to be minor and of limited spatial extent (ICES 1992).

Deep borrow pits can become anaerobic during certain times of the year. The dissolved oxygen concentration within these pits can be depressed to a level that adversely affects the ability of fish and invertebrates to utilize the area for spawning, feeding, and development (Pacheco 1984).

Release of contaminants

A number of factors (i.e., environmental, geochemical, and biological) influence the potential release and bioavailability of sediment contaminants. The toxicity of such releases, in general, is primarily dependent upon the contaminant involved, its concentration in the sediments and its chemical/geochemical state. Persistent organic pollutants (POPs), such as polyaromatic hydrocarbons (PAHs), pesticides, and polychlorinated biphenyl (PCBs), are sequestered in the total organic carbon (TOC) fraction of sediments (USEPA 2003a; USEPA 2003b; USEPA 2003c). Similarly, heavy metals are sequestered by acid volatile sulfides (AVS) and the TOC fraction of marine sediments (USEPA 2005a). For POPs like PAHs, the ratio of the concentrations of these contaminants relative to those of the fractions govern bioavailability and hence toxicity (USEPA 2003a). In the case of metals, bioavailability is governed by an excess of AVS concentrations relative to the metal concentrations as normalized by TOC (USEPA 2005a). Sand and gravel sediments typically contain low TOC and AVS concentrations, and where there is a prominent source of POPs and metals, such as in highly industrialized riverways, these coarser sediments could in fact release such contaminants when disturbed or oxidized. However, the coarse-grained sediments typically targeted for aggregate mining tend to be found in high-energy environments which are not depositional areas that can be sinks for fine-grained material containing POPs and metals. Since most offshore sand and gravel deposits do not have prominent nearby sources of POPs and metals, these deposits are generally low in contaminants (ICES 1992; Pearce 1994). Thus, the mining of offshore sand and gravel material typically do not release high levels of contaminants. In addition, because of their relatively large particle size, low surface area relative to total bulk, and low surface activity (i.e., few clay or organic materials to interact chemically), there is usually little chemical interaction in the water column (Pearce 1994). However, extraction of material in estuaries or deep channels, where fine material accumulates and is subject to anthropogenic pollution deposition, may be more likely to release harmful chemicals during dredging and excavation (Pearce 1994). Refer to the chapters on Coastal Development, Marine Transportation, and Chemical Effects: Water Discharge Facilities for additional information on the release of contaminants during dredging and excavation.

Sediment transport from site

Excavation at an offshore mining site that contains fine material can release suspended sediments into the water column during the excavation, as well as in the sorting or screening process. The distance the dredged material is transported from the excavation site will be dependent upon the current strength, storm resuspension, water salinity and temperature, and the grain size of the suspended material (ICES 1992). Some of the potential effects of redeposition of fines include smothering of demersal fish eggs on spawning grounds and the suffocation of filter-feeding benthos, such as shellfish and anemones (ICES 1992; Pearce 1994). Small-scale aggregate mining operations that are conducted in relatively shallow water and involving sandy, coarse-grained sediments often have relatively minimal physical and biological impacts on the surrounding seabed (Hitchcock and Bell 2004).

Noise impacts

Anthropogenic sources of ocean noise appear to have increased over the past decades, and have been primarily attributed to commercial shipping, offshore gas and oil exploration and drilling, and naval and other uses of sonar (Hildebrand 2004). Offshore mineral mining likely contributes to the overall range of anthropogenic ocean noise, but little information exists regarding specific effects on marine organisms and their habitats or the importance of offshore mining relative to other sources of anthropogenic noise. The dredging equipment noise generated in offshore mining may be similar to navigation channel dredging in nearshore habitats; however, because of the greater water depths involved in offshore mining, the noise may be propagated for greater distances than in confined nearshore areas (Hildebrand 2004). Reductions in Atlantic herring catches on the Finnish coast were hypothesized to be due to disturbance to the herring movement patterns by noise and activity associated with sand and gravel mining activities (Stewart and Arnold 1994). Refer to the chapters on Global Affects and Other Impacts and Marine Transportation for additional information on noise impacts.

Conservation measures and best management practices for offshore mineral mining

1. Avoid mining in areas containing sensitive or unique marine benthic habitats (e.g., spawning and feeding sites, surface deposits of cobble/gravel substrate).
2. Complete a comprehensive characterization of the borrow site and its resources prior to permit completion. Some of the components of a thorough assessment include:
    a. Determine the optimum dimensions of the borrow pit (i.e., small and deep areas or wide and shallow areas) in terms of minimizing the effects on resources.
    b. Prioritize the optimal locations of sand mining in terms of effects on resources.
    c. Assess the sand infill rates of borrow pits after completion.
    d. Assess the sediment migration patterns and rates as well as the side slope and adjacent natural seabed stability of the borrow pits after completion.
    e. Model and estimate the effect of massive and/or long-term sand mining on the surrounding seabed, shoreface (i.e., inner continental shelf), sand budgets, and resources.
    f. Assess the effect of removal (by dredging) of offshore sand banks/shoals on the surrounding natural seabed, adjacent shoreline, and the resources that use those habitats.
    g. Assess the effect of massive and/or long-term sand mining on the ecological structure of the seabed.
    h. Assess the effect of noise from mining operations on the feeding, reproduction, and migratory behavior of marine mammals and finfish.
3. Use site characterization and appropriate modeling to determine the areal extent and depth of extraction that affords expedited and/or complete recovery and recolonization times.
4. Employ sediment dispersion models to characterize sediment resuspension and dispersion during mining operations. Use model outputs to design mining operations, including "at sea" processing, to limit impacts of suspended sediment and turbidity on fishery resources and minimize the area affected.
5. Address the cumulative impacts of past, present, and foreseeable future development activities on aquatic habitats by considering them in offshore mining review processes.
6. Use seasonal restrictions when appropriate to avoid temporary impacts to habitat during species critical life history stages (e.g., spawning, and egg, embryo, and juvenile development). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements. Resource managers should incorporate adequate time for habitat recovery of affected functions and values to levels required by managed species.

Petroleum Extraction

Introduction

After some intense but unsuccessful petroleum exploration on the northeastern US continental shelf, the attention for commercial quantities of oil and gas have been directed elsewhere. Georges Bank and the continental shelf off New Jersey were thought to contain significant reserves of natural gas and several exploratory wells were drilled to locate and characterize those reserves in the late 1980s and early 1990s. At that time, few commercially viable reserves were found and the focus of petroleum exploration shifted to other regions. However, this could change in the future considering the escalating market prices and dwindling supplies of petroleum. Should renewed interest in offshore petroleum exploration and extraction in the northeast region occur, existing regulatory guidance on petroleum exploration and extraction, as well any recent research and development efforts, should be employed to ensure that marine resource impacts can be avoided, minimized, and compensated for these types of activity.

Petroleum extraction has impacts similar to mineral mining but usually with significantly less of an impact footprint (excluding spills). However, there is more risk and occurrence of adverse impacts associated with equipment operation, process related wastes and handling of byproducts (e.g., drill cuttings and spent drilling mud) which can disrupt and destroy pelagic and benthic habitats (Malins 1977; Wilk and Barr 1994). Potential releases of oil and petroleum byproducts into the marine environment may also occur as a result of production well blow-outs and spills.

Drilling muds are used to provide pressure and lubrication for the drill bit and to carry drill cuttings (crushed rock produced by the drill bit) back to the surface. Drilling muds and their additives are complex and variable mixtures of fluids, fine-grained solids, and chemicals (MMS 2005b). Some of the possible impacts associated with petroleum extraction include the dispersion of soluble and colloidal pollutants, as well as the alteration of turbidity levels and benthic substrates. Many of these impacts can be mitigated by on-site reprocessing and by transferring substances deemed inappropriate for unrestricted openwater disposal to landside disposal.

For more information on petroleum-related impacts and conservation recommendations for petroleum exploration, production, and transportation refer to the Energy-related Activities chapter of this report.

Offshore Dredged Material Disposal

Introduction

The disposal of dredged material in offshore waters involves environmental effects beyond those associated with the actual dredging operations. The US Army Corps of Engineers (USACE) disposes approximately 65% of its dredged material in open water, as opposed to “upland,” or land disposal (Kurland et al. 1994). Although some adverse environmental effects can be avoided with land disposal, there are a number of drawbacks including securing large tracts of land, material handling problems, overflow and runoff of polluted water, saltwater intrusion into groundwater, and costs of transporting material to land disposal sites (Kurland et al. 1994).

Disposal of dredged material is regulated under the Clean Water Act (CWA) and the Marine Protection, Research, and Sanctuaries Act (MPRSA), also known as the Ocean Dumping Ban Act (33 U.S.C. § 1251 and 1401 et seq.). The differences in the two Acts are found in the necessity and type(s) of sediment testing required by each. Generally, ocean dumping only requires biological testing if it is determined that the sediments do not meet the testing exclusion criteria as specified under the MPRSA (i.e., are contaminated). While the CWA provides for biological testing, it does not require such tests to determine whether the sediment meets the 404b testing guidelines unless specified by the USACE or the US Environmental Protection Agency (US EPA). The US EPA and the USACE are currently involved in discussions intended to combine the testing and evaluation protocols described in regulations, and in the "Greenbook" (Ocean Dumping Ban Act) and "Inland" (CWA) testing manuals. Currently, the US EPA and USACE use a tiered approach under both Acts, based upon empirical data gathered from each evaluated dredging project for determining the appropriate management options for dredge spoils (i.e., unconfined open water disposal, open water disposal with capping [CWA only], no open water disposal, or confined area disposal in harbors). Under the CWA, sediment quality guidelines or benchmarks can be used in the lower tiers to determine compliance with 404b guidelines or the need for futher testing. Although not required under the MPRSA, regulators in practice often use sediment chemistry to help determine the contaminant and sampling requirements for biological tests.

Offshore disposal sites are identified and designated by the US EPA using a combination of the MPRSA and National Environmental Policy Act (NEPA) criteria. However, the permitted use of designated disposal sites under these laws is not usually associated with the designation of the sites. To be eligible to use an offshore (i.e., federal waters) disposal site for dredged materials, project proponents must demonstrate: (1) that there are no reasonable and practical alternative disposal options available and; (2) that the sediments are compatible with natural sediments at the disposal site and are not likely to disrupt or degrade natural habitats and/or biotic communities (USEPA 2005b). Dredge material disposed at sites managed under the MPRSA must meet Ocean Dumping Ban Act criteria, which do not permit disposal of contaminated dredged material (USEPA 2005b).

Burial/disturbance of benthic habitat

Studies using sidescan sonar and bottom video have been used to distinguish natural sediment character and evidence of past dumping of mud and boulders on sand bottom (Buchholtz ten Brink et al. 1996). These studies have indicated that not only have dumped materials disturbed and altered benthic habitats, but that in some cases (such as on Stellwagen Basin) the material dumped in the past was scattered far from the intended target areas (Buchholtz ten Brink et al. 1996). The discharge of dredged material disturbs benthic and pelagic communities during and after disposal. The duration and persistence of those impacts to the water column and seafloor are related to the grain size and specific gravity of the dredge spoil. Impacts to benthic communities are identified and assessed in the site designation documents (Battelle 2004; URI 2003), which may include benthic communities being buried and smothered and the physicochemical environment in which they reside being altered.

However, Rhoads and Germano (1982, 1986) and Germano et al. (1994) note that recolonization of benthic infauna at a disposal site following dumping often leads to increased occurrences of opportunistic species (Stage I), which are then heavily preyed upon by Stage II and III (e.g., target fisheries) species. According to these studies, this plethora of prey, resulting from the disturbance of the community structure, can at least temporarily increase the productivity at the disposal site. However, chronic disturbance from repeated disposal may prevent Stage III communities from establishing (Germano et al. 1994).

Conversion of substrate/habitat and changes in sediment composition

Dumping dredged materials results in varying degrees of change in the physical, chemical, and biological characteristics of the substrate. The discharges can adversely affect infauna, including benthic and epibenthic organisms at and adjacent to the disposal site by burying immobile organisms or forcing motile organisms to migrate from the area. Benthic infauna species that have greater burrowing capabilities may be better able to extricate themselves from the overburden of sediment. Seasonal constraints on dredging and disposal not withstanding, it is assumed that there is a cyclical and localized reduction in the populations of benthic organisms at a disposal site. Plants and benthic infauna present prior to a discharge are unlikely to recolonize if the composition of the deposited material is significantly different (NEFMC 1998). Altered sediment composition at the disposal site may reduce the availability of infaunal prey species, leading to reduced habitat quality (Wilber et al. 2005).

Siltation, sedimentation, and turbidity

Increased suspended sediment released during the discharge process and the associated increase in turbidity may hinder or disrupt activities in the pelagic zone (i.e., predator–prey relationships and photosynthesis rates). It has been estimated that less than 5% of the material in each disposal vessel is unaccounted for during and after the disposal activity (Bohlen et al. 1996), but the specific volume is influenced by both mechanical and sediment characteristics.

The discharge of dredged material usually results in elevated levels of fine-grained mineral particles, usually smaller than sand (i.e., silt/clay), and organic particles being introduced into the water column (i.e., suspended sediment plumes). The suspended particulates reduce light penetration, which affects the rate of photosynthesis and the primary productivity of an aquatic area. Typically, the suspended materials are dispersed and diluted to levels approaching ambient within 1-4 hours of the release (Bohlen et al. 1996). However, the turbidity plume resulting from a discharge can last much longer, particularly near the bottom, if the dredge material is composed of fine-grain material. In the plume field, living marine resources may experience either reduced or enhanced feeding ability as a result of the disruption of water clarity, depending upon the predator-prey relationships and the type(s) of avoidance/feeding methodologies used by the species. For instance, summer flounder (*Paralichthys dentatus*) and bluefish (*Pomatomus saltatrix*) are sight feeders and avoid areas with reduced water clarity resulting from suspended sediment such as might be found at a dredging or disposal site (Packer et al. 1999). Conversely, recent deposits of sediment at dumpsites have been reported to act as an attractant for other species of fish and crustaceans such as winter flounder (*Pseudopleuronectes americanus*) and American lobster (*Homarus americanus*) even though winnowing of fine-grained material from the excavation site or deposit mound was ongoing at the site (USACE 2001).

Generally, the severity of the effects of suspended sediments on aquatic organisms increases as a function of the sediment concentration and the duration of exposure (Newcombe and Jensen 1996). Some of the effects of suspended sediments on marine organisms can include altered foraging patterns and success (Breitburg 1988), gill abrasion and reduced respiratory functions, and death (Wilber and Clark 2001). The sensitivity of species to suspended sediments is highly variable and dependent upon the nature of the sediment and the life history stage of the species. Mortality caused by suspended sediments for estuarine species have been reported from less than 1000 mg/L for 24 hours in highly sensitive species (e.g., Atlantic silversides [*Menidia menidia*], juvenile bluefish [*Pomatomus saltatrix*]) to greater than 10,000 mg/L for 24 hours in tolerant species (e.g., mummichog [*Fundulus heteroclitus*], striped killifish [*Fundulus majalis*], spot [*Leiostomus xanthurus*], oyster toadfish [*Opsanus tau*], hogchoker [*Trinectes maculates*]) (Wilber and Clark 2001). The egg and larval stages of marine and estuarine fish exhibit some of the most sensitive responses to suspended sediment exposures of all the taxa and life history stages studied (Wilber and Clark 2001). Impacts that have been identified for demersal eggs of fish from sedimentation and suspended sediments include delayed hatching and decreased hatching success (Wilber and Clark 2001; Berry et al. 2004). The development of larvae may be delayed or altered after exposure of elevated suspended sediments, and increased mortality rates in the larvae of some species, such as striped bass (*Morone saxatilis*) and American shad (*Alosa sapidissima*), have been reported with exposure of suspended sediment concentrations less than or equal to 500 mg/L for 3 to 4 days (Wilber and Clark 2001).

The effects of sedimentation on benthic organisms can include smothering and decreased gas exchange, toxicity from exposure to anaerobic sediments, reduced light intensity, and physical abrasion (Wilber et al. 2005). Mobile benthic species that require coarse substrates, such as gravel or cobble (e.g., American lobster) may be forced to seek alternate habitat that is less optimal or compete with other species or individuals for suitable habitat (Wilber et al. 2005). Messieh et al. (1981) investigated sedimentation impacts on Atlantic herring in laboratory experiments and found increased mortality in herring eggs, early hatching and shorter hatching lengths, and reduced feeding success in herring larvae leading to stunted growth and increased mortality.

Although there is generally a consensus among scientists and resource managers that elevated suspended sediments and sedimentation on benthic habitat caused by dredging and disposal of dredge spoils result in adverse impacts to marine organisms, the specific effects on biological communities need to be better quantified. Additional research is needed to investigate dose-response models at scales appropriate for dredging and disposal and for appropriate species and life history stages (Wilber et al. 2005).

Release of contaminants

Dredged material suspended in the water column can react with the dissolved oxygen in the water and result in localized depression of the oxygen level. However, research has indicated that reductions in dissolved oxygen levels during offshore sediment disposal is not appreciable or persistent in the general sediment classes found in the northeast region (USACE 1982; Fredette and French 2004; USEPA 2004).

In certain situations, trace levels of toxic metals and organics, pathogens, and viruses adsorbed or adhered to fine-grained particulates in the dredged material may become biologically available to organisms either in the water column or through food chain processes. Some of these pollutants and their concentrations are evaluated during project-specific sediment testing required under the MPRSA and CWA. Adverse chemical effects at the disposal site can be minimized through the sediment testing requirements under the MPRSA and CWA, since the discharge of potentially toxic materials are generally prohibited. Risk assessment approaches are used to further evaluate potential impacts using results from the MPRSA and CWA bioaccumulation and toxicity testing. In addition, monitoring is conducted to ensure that the biological and ecological functions and values are maintained within the site, notwithstanding the physical impacts associated with continued use of the site. However, some discharges of contaminated material may be permitted under CWA disposal regulations, if the sediments meet minimum testing criteria or the toxic affects can be managed by capping with clean material.

Fredette and French (2004) concluded that, after thirty-five years of monitoring and research, dredged material evaluated through preproject testing and deposited in properly located ocean disposal sites will remain where it is placed and have no unacceptable adverse effects on nearby marine resources. Furthermore, they concluded that the only discernible adverse impacts were near-field and short-term. These determinations were based on the magnitude of disposal activity relative to natural (e.g., storms) and other anthropogenic (e.g., outfalls) impacts (Rhoads

1994; Rhoads et al. 1995) and the low level of disposal-related impacts that have been documented (Fredette et al. 1993).

Changes in bottom topography, altered hydrological regimes, and altered current patterns

A concern often raised is the stability of dredge spoil sediments placed on the seafloor. Because ocean disposal sites are typically located in low current areas with water depths in excess of the active erosion zone, the material is generally contained within the disposal site. However, before 1985, dredged material sites were occasionally located in water depths insufficient to retain materials placed there (USEPA 1986). For example, the Mud Dump Site, located in the New York Bight Apex slope area off New York Harbor, contains water depths as shallow as 15 m and the site experienced extensive erosion by a nor'easter storm in October 1992 (USEPA 1997). Reclassified as a remediation site in 1997, the site is now known as the Historic Area Remediation Site (HARS). Erosion was reported at depths of 26 m, and the winnowed sediment included grain sizes up to small cobble. Fortunately, much of the sediment was relocated into deeper portions of the site westward of the erosion field (USEPA 1997). More comprehensive evaluation protocols have been put into place since 1985 to prevent dredged or fill material discharged at authorized sites from modifying current patterns and water circulation by obstructing the flow, changing the direction or velocity of water flow and circulation, or otherwise significantly altering the dimensions of a water body.

The USACE utilizes more than twenty selected or designated offshore dredged material disposal sites in the northeast region of the United States. Several of these sites have been used because they are dispersive in nature. These sites are used, normally, to put littoral material back into the nearshore drift pattern. The containment sites have an average size of 1.15 square nautical miles in size (USACE 2005b). By law and regulation, the significant adverse effects of dredged material disposal activities must be contained within the designated or selected disposal site and even those impacts must not degrade the area's overall ecological health. There is some dispersion of fine-grained sediments and contaminants outside the sites. Each site is required to have and be managed under a dredged material monitoring and management plan that assesses the health and well-being of the site and surrounding environment. Monitoring of disposal sites is a part of these plans, which is designed to ensure that any degradation of resources or alteration in seafloor characteristics are identified and would illicit actions by permitting agencies (USEPA 2004).

Release of nutrients/eutrophication

Nutrient overenrichment, or eutrophication, is one of the major causes of aquatic habitat decline associated with human activities (Deegan and Buchsbaum 2005). There are point sources of nutrients, such as sewage treatment outfalls, and nonpoint sources, such as urban storm water runoff, agricultural runoff, and atmospheric deposition, which have been discussed in other chapters of this report. Elevated levels of nutrients have undesirable effects, including: (1) increased incidence, extent, and persistence of blooms of noxious or toxic species of phytoplankton; (2) increased frequency, severity, spatial extent, and persistence of hypoxia; (3) alterations in the dominant phytoplankton species, which can reduce the nutritional and biochemical nature of primary productivity; and (4) increased turbidity levels of surface waters, leading to reductions in submerged aquatic vegetation (O'Reilly 1994).

Sediment particles can bind to some nutrients, and resuspension of sediments following dredge material disposal can cause a rapid release of nutrients to the water column (Lohrer and Wetz 2003). Ocean disposal of dredge material with high organic content can result in oxygen reduction (hypoxia) or even anaerobic conditions (anoxic) on the bottom and overlaying waters, particularly during periods when strong thermoclines are present (Kurland et al. 1994). Hypoxic and anoxic conditions can kill benthic organisms or even entire communities and lead to a proliferation of stress-tolerant species of reduced value to the ecosystem (Kurland et al. 1994). Generally, offshore waters are less sensitive to disposal of dredge material containing nutrients than inshore, enclosed water bodies.

Both the MPRSA and CWA regulations prohibit the discharge of dredge material containing high organic content and nutrient levels if the discharge results in adverse effects to the marine environment. However, prior to the stricter regulations instituted in the 1980s, the discharge of sewage sludge was permitted for decades in nearshore and offshore waters of many urbanized centers of the northeastern US coast (Barr and Wilk 1994).

Conservation measures and best management practices for dredge material disposal

1. Ensure that all options for disposal of dredged materials at sea are comprehensively assessed. The consideration of upland alternatives for dredged material disposal sites must be evaluated before offshore sites are considered.
2. Ensure that adequate sediment characterizations are completed and available for making informed decisions.
3. Ensure that adequate resource assessments are completed and available during project evaluation.
4. Employ sediment dispersion models to characterize sediment resuspension and dispersion during operations. Use model outputs to design disposal operations, including measures to avoid and minimize impacts from suspended sediment and turbidity on living marine resources. Sediment dispersion models should be field-verified to various sediment and hydraulic conditions to ensure they have been calibrated appropriately to predict sediment transport and dispersion.
5. Consider “beneficial uses” of dredged material, as appropriate.
6. Ensure that the site evaluation criteria developed for selection or designation of dredged material disposal sites have been invoked and evaluated, as appropriate.
7. Avoid dredged material disposal activities in areas containing sensitive or unique marine benthic habitats (e.g., spawning and feeding sites, surface deposits of cobble/gravel substrate).
8. Employ all practicable methods for limiting the loss of sediment from the activity. Consider closed or “environmental” buckets, when appropriate.
9. Ensure that disposal sites are being properly managed (e.g., disposal site marking buoys, inspectors, the use of sediment capping and dredge sequencing) and monitored (e.g., chemical and toxicity testing, benthic recovery) to minimize impacts associated with dredge material.
10. Use sequential dredging to avoid dredging activity during specific time periods in particularly environmentally sensitive areas of large navigation channel dredging projects. This can avoid turbidity and sedimentation, bottom disruption, and noise in sensitive areas used by fishery resources during spawning, migration, and egg development.
11. Require appropriate monitoring to avoid and minimize individual and cumulative impacts of the disposal operations.
12. Use seasonal restrictions when appropriate to avoid temporary impacts to habitat during critical life history stages (e.g., spawning, egg and embryo development, and juvenile growth). Recommended seasonal work windows are generally specific to regional or watershed-level environmental conditions and species requirements. Resource managers should incorporate

adequate time for habitat recovery of affected functions and values to levels required by managed species.

Fish Waste Disposal

Introduction

Fish waste or material resulting from industrial fish processing operations from either wild stocks or aquaculture consists of particles of flesh, skin, bones, entrails, shells, or process water (i.e., liquid "stickwater" or "gurry"). The organic components of fish waste have a high biological oxygen demand and, if not managed properly, can pose environmental and health problems. Generally, the solid wastes make up 30-40% of total production, depending on the species processed (IMO 2005a). Most fish wastes degrade rapidly in warm weather and can cause aesthetic problems and strong odors as a result of bacterial decomposition if not stored properly or disposed of quickly. Because these waste streams are generally required to be pretreated and fully processed on-site, disposed at a suitable upland site, or sent through municipal sewage treatment, at sea disposal is no longer widely employed in the northeastern United States. However, these materials are sometimes discharged at sea, when appropriate.

Permitting of at sea disposal should be coordinated with appropriate federal and state agencies. Processors should contact the US EPA to determine whether federal permits are necessary for the activity. In order to determine if a federal permit applies, the US EPA must determine if the material constitutes an environmental risk or is a traditional and acceptable "fish waste" disposal defined under Section 102(d) of the Ocean Dumping Ban Act, 33 U.S.C. Part 1412(d) and the regulations promulgated at 40 C.F.R. Part 220. Generally, permits are not required for the transportation or the ocean disposal of fish waste unless: 1) disposal is proposed in harbors or other protected and enclosed waters, and the location is deemed by the EPA as potentially endangering human health, the marine environment or ecological systems; or 2) the waste contains additives or disinfectants from the processing or treatment. In these cases, National Pollutant Discharge Elimination System (NPDES) permits may be required if chlorine or other similar chemicals are used. If an environmental or human health risk is determined, the applicant may be required to submit an assessment of the disposal area and potential impacts to marine resources and follow disposal guidelines consistent with the provisions of the London Convention 1972 (IMO 2005a). Permits required for ocean disposal of fish wastes define the discharge rate of the fluids, residual tissue, and hard part pieces by using a dispersion model. Inputs to the model include discharge flow rate, tissue dimensions, mixing rates, local current patterns, and the specific gravity of the solids (USEPA 2005c). The US EPA may also consult with applicable federal and state regulatory and resource agencies and regional fisheries councils, to identify any areas of concern with respect to the disposal area and activity. Persons wishing to dispose of fish wastes in the ocean may be required to submit specific dilution modeling in support of the proposed disposal and participate in monitoring to verify the results of the modeling (USEPA 2005c).

Bivalve shells, when brought ashore and processed, are not allowed to be returned to the ocean for the purpose of waste disposal. Reuse of the shells as "cultch" in oyster farming operations is a standard, traditional fishing practice in the northeastern United States and does not require permitting, but prior to disposal the shells may be required to meet water quality criteria, principally regarding residual tissue volume.

The guidelines established by the London Convention 1972 place emphasis on progressively reducing the need to use the sea for dumping of wastes. Implementation of these guidelines and the regulations promulgated by US EPA for the disposal of fish wastes includes consideration of potential waste management options that reduce or avoid fish waste to the disposal stream. For example, applications for disposal should consider reprocessing to fishmeal, composting, production of silage (i.e., food for domestic animals/aquaculture), use in biochemical industry products, use as fertilizer in land farming, and reduction of liquid wastes by evaporation (IMO 2005a).

Introduction of pathogens

Ocean disposal of fish wastes has the potential to introduce pathogens to the marine ecosystem that could infect fish and shellfish. In particular, aquaculture operations that raise nonnative species or those that provide food to animals derived from nonindigenous sources could introduce disease vectors to native species (IMO 2005a). However, the disposal guideline provisions implemented as part of the Ocean Dumping Ban Act is designed to ensure wide dispersion of the gurry and limited accumulation of soft parts waste on the sea floor. Models developed to predict the effects of authorized discharges of fish wastes were designed to avoid the accumulation of biodegradable materials on the seafloor and introduction of pathogens.

Release of nutrients/eutrophication

The organic components of fish wastes have a high biological oxygen demand (BOD) and if not managed properly could result in nutrient over-enrichment and reductions in the dissolved oxygen. In ocean disposal, these affects may be seen with mounding of wastes, subsequent increases in BOD and contamination with bacteria associated with partly degraded organic wastes (IMO 2005a). However, disposal guidelines require that dumpsite selection criteria maximizes waste dispersion and consumption of the wastes by marine organisms.

Release of biosolids

Generally, the solid wastes generated by fish waste disposal comprises approximately 30-40% of total production, depending upon the species processed (IMO 2005a). Biosolid waste at fish disposal sites could result in nutrient over-enrichment and reduced dissolved oxygen concentration. However, the disposal guideline provisions implemented as part of the Ocean Dumping Ban Act require wide dispersion of the gurry and limited accumulation of soft parts waste on the sea floor.

Alteration of benthic habitat

Ocean disposal of fish wastes that fail to meet permit conditions and guidelines have the potential to degrade fishery habitat by adversely affecting the productivity and ecological functions of the benthic community. Concentration and mounding of wastes can increase the BOD and reduce dissolved oxygen concentration of an area resulting in reductions in the ability to support small consumer organisms such zooplankton and amphipods. This can then affect species at higher trophic levels that depend upon these consumers for food. However, disposal guidelines require dump-site selection criteria that maximize waste dispersion and consumption of the wastes by marine organisms and disposal monitoring that ensures permit conditions are met (USEPA 2005c). In addition, guidelines and permit review must consider chemical contamination of the marine environment from the waste disposal. For example, the potential presence of chemicals used in aquaculture and fish wastes subjected to chemical treatment must be assessed prior to disposal (IMO 2005a).

Behavioral effects

The presence of biodegradable tissue in the water column has the potential to alter the behavior of organisms in various ways, such as causing an attractant source for scavengers. This could alter the diet of individuals and interfere with trophic-level energy dynamics and community structure. The discharge of process water and biosolid wastes should be monitored carefully to ensure conditions within state and federal permits are met.

Conservation measures and best management practices for disposal of fish wastes

1. Consider the practical availability of alternative methods of disposal to reuse, recycle, or treat the waste as a comparative risk assessment involving both ocean dumping and alternatives.
2. Perform site assessments of the proposed ocean disposal location prior to dumping, including the water depths, average velocities of tidal and nontidal currents, prevailing winds throughout the year, sediment and benthic habitat types, and nature of the sea floor (depositional versus dispersive). Information collected in the site assessment will be used in predictive models developed for the waste disposal activities. Existing uses of the site should be assessed, such as commercial and recreational fishing and whale watching vessels.
3. Use predictive models for plume dispersion and waste settlement based upon physical dynamics of the disposal area, nature of the fish waste, and the method of disposal. The models should be used to assess the probability of the waste plume reaching nearshore coastal waters or other protected areas, such as marine sanctuary waters. The models should also estimate the mass flux of nitrogen and organic carbon associated with the proposed discharges on a daily and annual basis, and how this input may affect phytoplankton production and benthic communities.
4. Dispose material at a steady rate while the vessel maintains headway speed (e.g., 3 nautical miles per hour) as opposed to dumping the entire load at once in a fixed location in order to provide better dilution of fish waste.
5. Grind organic materials to appropriate sizes (e.g., 0.5 inch) prior to discharge where they will be consumed or degraded in the water column dispersion field during and subsequent to their discharge. The intent should be to avoid water quality degradation and tissue deposition and accumulation on the seafloor.
6. Ensure that the waste will be rendered biologically inert during its residence time in the water column and avoid adverse effects on water quality, including reductions in dissolved oxygen concentrations and nutrient over-enrichment.
7. Require monitoring of the waste plume during and after discharge to verify model outputs and advance the knowledge regarding the practice of at-sea disposal of fish processing wastes.

Vessel Disposal

Introduction

When vessels are no longer needed, there are several options for their disposition, including reuse of the vessel or parts of the vessel, recycling or scrapping, creating artificial reefs, and disposal on land or sea (USEPA 2006). This section discusses the potential habitat and marine fisheries impacts associated with disposal at sea.

The disposal of vessels in the open ocean is regulated by the US EPA under section 102(a) of the MPRSA (Ocean Dumping Ban Act) and under 40 CFR § 229.3 of the US EPA regulations. In part, these regulations require that (1) vessels sink to the bottom rapidly and permanently and that marine navigation is not otherwise impaired by the sunk vessel; (2) all vessels shall be disposed of in depths of at least 1,000 fathoms (6,000 feet) and at least 50 nautical miles from land; and (3) before sinking, appropriate measures shall be taken to remove to the maximum extent practicable all materials which may degrade the marine environment, including emptying of all fuel tanks and lines so that they are essentially free of petroleum and removing from the hulls other pollutants and all readily detachable material capable of creating debris or contributing to chemical pollution.

The US EPA and US Department of Transportation Maritime Administration have developed national guidance, including criteria and best management practices for the disposal of ships at sea when the vessels are intended for creation or addition to artificial reefs (USEPA 2006). Vessels disposed of to create artificial reefs have historically been designed and intended to enhance fishery resources for recreational fishermen. However, in recent years artificial reefs have been constructed for a number of nonextractive purposes such as: (1) recreational SCUBA diving opportunities; (2) socioeconomic benefits to local coastal communities; (3) increase habitat to reduce user pressure on nearby natural reefs; (4) reduce user conflicts (e.g., diving in heavily fished areas), and; (5) provide mitigation or restoration to habitat loss for commercial activities (e.g., beach nourishment, dredging, pipeline routes) (NOAA 2007). Some vessels may be sunk to provide a combination of these purposes. Vessels prepared for use as artificial reefs should: (1) be "environmentally sound" and free from hazardous and potentially polluting materials; (2) have had resource assessments for the disposal locations conducted to avoid adverse impacts to existing benthic habitats; and (3) have had stability analyses for the sinking and the ship's ultimate location conducted to ensure there is minimal expectation of adverse impacts on adjacent benthic habitats. Several guidance documents have been developed for the planning and preparation of vessels as artificial reef material, including the National Artificial Reef Plan (NOAA 2007), Coastal Artificial Reef Planning Guide (ASMFC and GSMFC 1998), the Guidelines for Marine Artificial Reef Materials (ASMFC and GSMFC 2004), and the National Guidance: Best Management Practices for Preparing Vessels Intended to Create Artificial Reefs (USEPA 2006). These documents should be consulted to ensure that conflicts with existing uses of the potential disposal site/artificial reef site are addressed and that materials onboard the vessel do not adversely impact the marine environment. Section 203 of the National Fishing Enhancement Act of 1984 (Title II of P.L. 98-623, Appendix C) established that artificial reefs in waters covered under the Act shall "be sited and constructed, and subsequently monitored and managed in a manner which will: (1) enhance fishery resources to the maximum extent practicable; (2) facilitate access and utilization by US recreational and commercial fishermen; (3) minimize conflicts among competing uses of waters covered under this title and the resources in such waters; (4) minimize environmental risks and risks to personal health and property; and (5) be consistent with generally accepted principles of international law and shall not create any unreasonable obstruction to navigation."

The appropriate siting is vital to the overall success of an artificial reef. Considerations and options for site placement and function in the environmental setting should be carefully weighed to ensure program success. Since placement of a reef involves displacement and disturbance of the existing habitat, and building the reef presumably accrues some benefits that could not exist in the absence of the reef, documentation of these effects should be brought out in the initial steps to justify artificial reef site selection. Placement of a vessel to create an artificial reef should: (1) enhance and conserve targeted fishery resources to the maximum extent practicable; (2) minimize conflicts among competing uses of water and water resources; (3) minimize the potential for environmental risks related to site location; (4) be consistent with international law and national fishing law and not create an obstruction to navigation; (5) be based on scientific information; and (6) conform to any federal, state, or local requirements or policies for artificial reefs (USEPA 2006).

The Coastal Artificial Reef Planning Guide (ASMFC and GSMFC 1998) state that when an artificial reef has been constructed, another important phase of reef management begins: monitoring and maintenance. Monitoring provides an assessment of the predicted performance of reefs and assures that reefs meet the general standards established in the Section 203 of the National Fishing Enhancement Act as listed above. It also ensures compliance with the conditions of any authorizing permits. Artificial reef monitoring should be linked with performance objectives, which ensures that NOAA National Marine Fisheries Service responsibilities to protect, restore, and manage living marine resources, and to avoid and minimize any adverse effects on these resources are fulfilled.

Release of contaminants

Ships disposed of at sea, including those intended to create artificial reefs, are often military and commercial vessels which typically contain various materials that, if released into the marine environment, could have adverse effects on the marine environment. Some of the materials of concern include fuels and oil, asbestos, polychlorinated biphenyl (PCB), paint, debris (e.g., vessel debris, floatables, introduced material), and other materials of environmental concern (e.g., mercury, refrigerants) (USEPA 2006). Depending upon the nature of the contaminant and the concentration and duration of the release of contaminant(s) adverse effects to marine organisms may be acute or chronic and either lethal or sublethal. Some contaminants, such as PCB and mercury, can be persistent and bioaccumulate in the tissues of organisms resulting in more serious impacts in higher trophic level organisms. The Ocean Dumping Ban Act and the various guidance documents available for offshore disposal of vessels prohibit materials containing contaminants which may impact the marine environment. The guidance documents provide detailed best management practices regarding recommended measures to remove and abate contaminants contained within and as part of a vessel.

Release of debris

Debris, including solids and floatables, are materials that could break free from a vessel during transportation to the disposal site, and during and after sinking. The release of debris can adversely affect the ecological and aesthetic value of the marine environment. Debris released from vessels is generally categorized into vessel debris (material that was once part of the vessel) and clean-up debris (material that was not part of the vessel but was brought on board the vessel during preparation for disposal).

Some debris released from vessels is not highly degradable and can be persistent in the marine environment for long periods of time, increasing the threat it poses to the environment. Some of the impacts associated with debris include: (1) entanglement and/or ingestion, leading to injury, infection, or death of marine animals that may be attracted to or fail perceive the debris in the water; (2) alteration of the benthic floral and faunal habitat structure, leading to injury or mortality or indirect impact to other species linked in the benthic food web and; (3) elevation of the risk of spills and other environmental impacts caused by impacts with other vessels (e.g., hull damage, damage to cooling or propulsion systems) (USEPA 2006). The Ocean Dumping Ban Act and the various guidance documents available for offshore disposal of vessels require all debris to be removed from vessels prior to sinking. The guidance documents provide detailed best management practices regarding recommended measures to remove vessel and clean-up debris.

Conversion of substrate/habitat and changes in community structure

Vessels that are sunk for the purpose of discarding obsolete or decommissioned ships, as well as those sunk to create an artificial reef, can convert bottom habitat type and alter the ecological balance of marine communities inhabiting the area. For example, placement of vessels over sand bottom can change niche space and predator/prey interactions for species or life history stages utilizing that habitat type. Large structures such as ships tend to attract adult fish and larger predators, which may increase predation rates on smaller and juvenile fish or displace smaller fish and juveniles to other areas (USEPA 2006). Large, anthropogenic structures, such as oil and gas platforms in the Gulf of Mexico, have been shown to affect the distribution of larval and juvenile fish (Lindquist et al. 2005). In addition, large structures tend to provide proportionally less shelter for demersal fishes and invertebrates than smaller, lower profile structures, while the surfaces of steel hull vessels are less ideal for colonization by epibenthos than are natural surfaces like rock (ASMFC and GSMFC 2004). Certain types of habitat and areas may be more susceptible to physical and chemical impacts from the placement of vessels, particularly those vessels sunk as artificial reefs. Generally, vessels sunk for disposal only are located in deeper water (> 6,000 feet) and very far offshore (> 50 nautical miles from land) and may have less impacts on sensitive benthic habitats. However, vessels sunk as artificial reefs are usually located in nearshore coastal waters that also support or are frequented by marine resources that may be adversely impacted by the placement of the structure. Artificial reefs should not be sited in sensitive areas that contain coral reefs or other reef communities, submerged aquatic vegetation, or habitats known to be utilized by endangered or threatened species (USEPA 2006). The Ocean Dumping Ban Act prohibits vessel disposal in areas that may adversely effect the marine environment.

Changes in bathymetry and hydrodynamics

The location of a vessel on the ocean bottom will change the bathymetry and can potentially alter the current flow of the disposal area. A proposed disposal site should be assessed as to the effects the vessel disposal and subsequent bathymetry change may have on the hydrodynamics and geomorphology of the immediate and adjacent habitats. For example, even small vessels placed on the bottom can alter currents and create turbulence around the vessel that may scour existing soft substrates and adversely affect adjacent habitats and communities. In addition, the high vertical profile may cause some vessels to be prone to movement and structural damage from ocean currents and wave surge during storm events. For example, during Hurricane Andrew, a category 5 storm, in south Florida during 1992, nearly all steel-hulled vessels sunk as artificial reefs in the area of the storm's path sustained structural damage, and a number moved 100-700 m because of the storm surge (ASMFC and GSMFC 2004). The movement of vessels after disposal can impact adjacent habitats and relocate the vessels to areas that could alter the ecological balance of marine communities in the area. In addition, reductions in navigational clearance, either as a result of the vessel being sunk in the wrong location and in an area too shallow or because later movement of the vessel from storm surge or currents may increase the potential danger to vessel navigation (e.g., hull damage, damage to cooling or propulsion systems) which may cause further damage from oil/fuel spills or groundings (ASMFC and GSMFC 2004). To minimize the risk of alterations to the bathymetry and hydrodynamics of the disposal area and vessel movement, the Ocean Dumping Ban Act and the various guidance documents available for offshore disposal of vessels require a number of evaluations prior to dumping activities, including: (1) stability analyses; (2) assessments of the seabed, including topography and geological characteristics and; (3) assessment of mean direction and velocity of currents and storm-wave induced bottom currents (ASMFC and GSMFC 2004; IMO 2005b).

Deployment impacts

Some risks to the marine environment exist during the deployment (i.e., sinking) of vessels for disposal or as an artificial reef. Some potential impacts that may occur during deployment include the release of contaminants accidentally left onboard the vessel, damage to adjacent benthic habitats from anchors and cables used to maintain the vessel position as it sinks, impacts to benthic habitats from a vessel accidentally sinking in an unintended location while being towed or from movement of the ship after deployment (ASMFC and GSMFC 2004). However, careful planning during the assessment stages and adherence to operational protocols can avoid impacts during deployment.

Conservation measures and best management practices for disposal of vessels

1. Require that a vessel disposal site assessment adequately characterize the physical and biological environment of the site. In addition to identifying the habitat types and species utilizing the area and targeted for enhancement, ecological investigations should include community settlement and recruitment and predator/prey dynamics and anticipated changes in competition and niche space as a result of the vessel disposal (USEPA 2006).
2. Identify the locations of any sensitive marine habitats in the area. Potential vessel disposal sites should generally not be located near any of the following marine resources: coral reefs; significant beds of aquatic vegetation or macroalgae; oyster reefs; scallop, mussel, or clam beds; existing live bottom (i.e., marine areas supporting sponges, sea fans, corals, or other sessile invertebrates generally associated with rock outcrops); and habitats of endangered or threatened species (federal and state listed) (USEPA 2006).
3. Conduct vessel stability analysis to ensure the vessel is retained in the intended location, including characterization of anticipated weather conditions, tidal dynamics, mean direction and velocity of surface and bottom drifts and storm-wave induced currents, and general wind and wave characteristics (IMO 2005b).
4. Ensure that a thorough inventory and assessment of all potential contaminants on the vessel are completed and that all preplacement cleaning and inspections are completed thoroughly and effectively.
5. Avoid the use of explosives to the extent possible in sinking vessels under 150 feet in length where alternate methods (e.g., opening seacocks, flooding with pumps, etc.) are feasible (ASMFC and GSMFC 2004).
6. Monitor the disposal operation and the placement site for adherence to permit compliance and performance objectives.
7. Ensure that physical and biological monitoring plans for vessels disposed of as artificial reefs are developed as appropriate and that monitoring and reporting requirements are met throughout the designed timeframe.

References for Offshore Dredging and Disposal

[ASMFC], [GSMFC] Atlantic and Gulf States Marine Fisheries Commissions. 1998. Coastal artificial reef planning guide. Washington (DC): Joint Artificial Reef Technical Committee of the Atlantic and Gulf States Marine Fisheries Commissions. 45 p.

[ASMFC], [GSMFC] Atlantic and Gulf States Marine Fisheries Commissions. 2004. Guidelines for marine artificial reef materials, 2nd ed. Washington (DC): Subcommittees of the Atlantic and Gulf States Marine Fisheries Commissions. Number 121. 198 p.

Barr BW, Wilk SJ. 1994. Sewage sludge and industrial waste dumping. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 41-4.

Battelle. 2004. Designating an ocean disposal site for dredged material. [Internet]. Columbus (OH): Battelle Memorial Institute; [cited 2007 Dec 21]. Available from: http://www.battelle.org/Environment/publications/EnvUpdates/summer2004/article1.stm.

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

Bohlen WF, Howard-Strobel MM, Cohen DR, Morton ET. 1996. An investigation of the dispersion of sediments resuspended by dredging operations in New Haven Harbor. Waltham (MA): New England Division, US Army Corps of Engineers. DAMOS Contribution #112. 12 p + appendices.

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

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

Burlas M, Ray GL, Clarke D. 2001. The New York District's biological monitoring program for the Atlantic Coast of New Jersey, Asbury Park to Manasquan Section beach erosion control project. [Internet]. Final Report. Vicksburg (MS): US Army Corps of Engineers, New York and US Army Engineer Research and Development Center, Waterways Experiment Station. [cited 2007 Dec 21]. Available from: http://www.nan.usace.army.mil/business/prjlinks/coastal/asbury/index.htm.

Deegan LA, Buchsbaum RN. 2005. The effect of habitat loss and degradation on fisheries. In: Buchsbaum R, Pederson J, Robinson WE, editors. The decline on fisheries resources in New

England: evaluating the impact of overfishing, contamination, and habitat degradation. Cambridge (MA): MIT Sea Grant College Program; Publication No. MITSG 05-5. p 67-96.

Drucker BS, Waskes W, Byrnes MR. 2004. The US Minerals Management Service outer continental shelf sand and gravel program: environmental studies to assess the potential effects of offshore dredging operations in federal waters. Journal of Coastal Research 20(1):1-5.

Fredette TJ, French GT. 2004. Understanding the physical and environmental consequences of dredged material disposal: history in New England and current perspectives. Marine Pollution Bulletin 49(1-2):93-102.

Fredette TJ, Kullberg PG, Carey DA, Morton RW, Germano JD. 1993. Twenty-five years of dredged material disposal site monitoring in Long Island Sound: a long-term perspective. In: Van Patten MS, editor. Proceedings of the Long Island Sound Research Conference; 1992 Oct 23-24; New Haven, CT. Groton (CT): Connecticut Sea Grant Program, Publication No. CT-SG-93-03, p 153-61.

Germano JD, Rhoads DC, Lunz JD. 1994. An integrated, tiered approach to monitoring and management of dredged material sites in the New England region. Waltham (MA): US Army Corps of Engineers, New England Division. DAMOS Contribution No. 87, Report No. SAIC-90/7575&234. 55 p + appendices.

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

Hitchcock DR, Bell S. 2004. Physical impacts of marine aggregate dredging on seabed resources in coastal deposits. Journal of Coastal Research 20(1):101-114.

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

[IMO] International Maritime Organization. 2005a. Specific guidelines for assessment of fish wastes, or material resulting from industrial fish processing operations. [Internet]. London (England): IMO, Office for the London Convention 1972. [cited 2007 Dec 21]. 12 p. Available from: http://www.imo.org/includes/blastDataOnly.asp/data_id%3D17022/3-FishWaste.pdf.

[IMO] International Maritime Organization. 2005b. Specific guidelines for assessment of vessels. [Internet]. London (England): IMO, Office for the London Convention 1972. [cited 2007 Dec 21]. 13 p. Available from: http://www.imo.org/includes/blastDataOnly.asp/data_id%3D17023/4-Vessels.pdf.

Johnson SW, Stanley DR, Moles DA. 1998a. Effects of submarine mine tailings disposal on juvenile yellowfin sole (*Pleuronectes asper*): a laboratory study. Marine Pollution Bulletin 36(4):278-87.

Johnson SW, Stone RP, Love DC. 1998b. Avoidance behavior of ovigerous Tanner crabs (*Chionoecetes bairdi*) exposed to mine tailings: a laboratory study. Alaska Fishery Research Bulletin 5:39-45.

Kenny AJ, Rees HL. 1996. The effects of marine gravel extraction on the macrobenthos: results 2 years post-dredging. Marine Pollution Bulletin 32(8-9):615-22.

Kurland JM, Ludwig FM, Gorski SW, Mantazaris C. 1994. Dredging and dredged-material disposal. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 44-7.

Lindquist DC, Shaw RF, Hernandez FJ Jr. 2005. Distribution patterns of larval and juvenile fishes at offshore petroleum platforms in the north-central Gulf of Mexico. Estuarine, Coastal and Shelf Science 62(4):655-65.

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

Malins DC, editor. 1977. Effects of petroleum on arctic and subarctic marine environments and organisms. New York (NY): Academic Press.

Messieh SN, Wildish DJ, Peterson RH. 1981. Possible impact from dredging and spoil disposal on the Miramichi Bay herring fishery. St. Andrews (New Brunswick): Canadian Technical Report of Fisheries and Aquatic Sciences. No. 1008. iv + 33 p.

[MMS] Minerals Management Service. 2005a. Marine Minerals Program, offshore energy and minerals management. [Internet]. Herndon (VA): MMS, [updated 2007 Jun 28; cited 2007 Dec 21]. Available from: http://www.mms.gov/sandandgravel/.

[MMS] Minerals Management Service. 2005b. Offshore discharges from oil and gas development operations – FAQ. Gulf of Mexico Region, Offshore Information. [Internet]. New Orleans (LA): MMS. [updated 2001 June 6; cited 2007 Dec 21]. Available from: http://www.gomr.mms.gov/homepg/offshore/egom/factshee.html.

[NOAA] National Oceanic and Atmospheric Administration. 2007. National Artificial Reef Plan (as amended): guidelines for siting, construction, development, and assessment of artificial reefs. Washington (DC): US Department of Commerce, National Oceanic and Atmospheric Administration. 51 p.

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

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

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

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

Packer DB, Griesbach SJ, Berrien PL, Zetlin CA, Johnson DL, Morse WW. 1999. Essential fish habitat source document: summer flounder, *Paralichthys dentatus*, life history and habitat characteristics. [Internet]. Woods Hole (MA): NOAA, NMFS NER, NEFSC. NOAA Technical Memorandum NMFS-NE-151. [cited 2008 Jul 22]. 88 p. Available from: http://www.nefsc.noaa.gov/nefsc/publications/tm/tm151/tm151.pdf.

Palermo MR, Clausner JE, Rollings MP, Williams GL, Myers TE, Fredette TJ, Randall RE. 1998. Guidance for subaqueous dredged material capping. Vicksburg (MS): US Army Corps of Engineers, WES, Dredging Operations and Environmental Research (DOER). Technical Report DOER-1. 302 p.

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

Rhoads DC. 1994. Analysis of the contribution of dredged material to sediment and contaminant fluxes in Long Island Sound. Special Technical Report. Waltham (MA): US Army Corps of Engineers, New England Division, Regulatory Division. DAMOS Contribution No. 88. 32 p.

Rhoads DC, Germano JD. 1982. Characterization of organism-sediment relations using sediment profile imaging: an efficient method of remote ecological monitoring of the seafloor (REMOTS ™ System). Marine Ecology Progress Series 8:115-28.

Rhoads DC, Germano JD. 1986. Interpreting long-term changes in benthic community structure: a new protocol. Hydrobiologia 142(1):291-308.

Rhoads DC, Kullberg PG, Fredette TJ. 1995. Dredged material disposal within an estuary: is it a significant source of sediment and contaminant fluxes? The case of Long Island Sound. In:

Knott MA, editor. Proceedings of the Conference for Ports '95, Volume 1; 1995 Mar 13-15; Tampa, FL. Reston (VA): American Society of Civil Engineers. p 428-39.

Scarrat DJ. 1987. Fisheries' interests and ocean mining. Marine Mining 6:141-7.

Stevenson DK, Scott ML. 2005. Essential fish habitat source document: Atlantic herring, *Clupea harengus*, life history and habitat characteristics (2nd ed). [Internet]. Woods Hole (MA): NOAA, NMFS, Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-NE-192. [cited 2008 Jul 22]. 94 p. Available from: http://www.nefsc.noaa.gov/nefsc/habitat/efh.

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

[URI] University of Rhode Island Coastal Institute. 2003. Designation of a long-term offshore disposal site for dredged material. [Internet]. [cited 2007 Dec 21]. Available from: http://www.ci.uri.edu/Projects/DD/.

[USACE] US Army Corps of Engineers. 1982. Final Supplemental Environmental Impact Statement for the designation of a disposal site for dredged material in Western Long Island Sound WLIS III. Waltham (MA): USACE, New England Division.

[USACE] US Army Corps of Engineers. 2001. Monitoring at the New London Disposal Site 1992-1998 Vol. I. [Internet]. Concord (MA): USACE, New England District. DAMOS Contribution No. 128, SAIC No. 515. [cited 2008 Jul 22]. 157 p + appendices. Available from: http://www.nae.usace.army.mil/damos/pdf/128.pdf.

[USACE] US Army Corps of Engineers. 2005a. HARS – Historic Area Remediation Site. [Internet]. [cited 2007 Dec 21]. Available from: http://www.nan.usace.army.mil/business/prjlinks/dmmp/benefic/hars.htm.

[USACE] US Army Corps of Engineers. 2005b. Managed dredged material disposal sites in the North Atlantic Division. [Internet]. [updated 2007 Apr 12; cited 2007 Dec 21]. Available from: http://www.nae.usace.army.mil/damos/maps.asp.

[USEPA] US Environmental Protection Agency. 1986. Ocean dumping site designation delegation handbook for dredged material. Washington (DC): US EPA Office of Marine and Estuarine Protection.

[USEPA] US Environmental Protection Agency. 1997. Supplemental to the Environmental Impact Statement on the New York dredged material disposal site designation for the designation of the Historic Area Remediation Site (HARS) in the New York Bight. New York (NY): US EPA.

[USEPA] US Environmental Protection Agency. 2003a. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: PAH mixtures. [Internet]. Washington (DC): US EPA Office of Research and

Development. Report No. EPA-600-R-02-013. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/PAHESB.pdf.

[USEPA] US Environmental Protection Agency. 2003b. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: dieldrin. [Internet]. Washington (DC): US EPA Office of Research and Development. Report No. EPA-600-R-02-010. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/dieldrin.pdf.

[USEPA] US Environmental Protection Agency. 2003c. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: endrin. [Internet]. Washington (DC): US EPA Office of Research and Development. Report No. EPA-600-R-02-009. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/endrin.pdf.

[USEPA] US Environmental Protection Agency. 2004. Rhode Island region long-term dredged material disposal site evaluation project. Boston (MA): US EPA, New England Region.

[USEPA] US Environmental Protection Agency. 2005a. Procedures for the derivation of equilibrium partitioning sediment benchmarks (ESBs) for the protection of benthic organisms: metal mixtures (cadmium, copper, lead, nickel, silver, and zinc). [Internet]. Washington (DC): US EPA Office of Research and Development. Report No. EPA-600-R-02-011. [cited 2008 Jul 24]. Available from: http://www.epa.gov/nheerl/publications/files/metalsESB_022405.pdf.

[USEPA] US Environmental Protection Agency. 2005b. Ocean dumping in the Southeast. [Internet]. [updated 2007 Nov 21; cited 2007 Dec 21]. Atlanta (GA): US EPA, Region 4. Available from: http://www.epa.gov/region4/water/oceans/.

[USEPA] US Environmental Protection Agency. 2005c. Disposal of fish waste. [Internet]. [updated 2007 Nov 1; cited 2007 Dec 21]. New York (NY): US EPA, Region 2. Available from: http://www.epa.gov/region2/water/oceans/fwaste.htm.

[USEPA] US Environmental Protection Agency. 2006. National guidance: best management practices for preparing vessels intended to create artificial reefs. [Internet]. Washington (DC): US EPA Office of Wetlands, Oceans, and Watersheds. EPA Report 842-B-06-002. [cited 2008 Jul 24]. 77 p. Available from: http://www.epa.gov/owow/oceans/habitat/artificialreefs/documents/0605finalreefguidance.pdf.

van Dalfsen JA, Essink K, Toxvig Madsen H, Birklund J, Romero J, Manzanera M. 2000. Differential responses of macrozoobenthos to marine sand extraction in the North Sea and the Western Mediterranean. Copenhagen (Denmark): ICES Journal of Marine Science 57:1439-45.

Wilber D, Brostoff W, Clarke D, Ray G. 2005. Sedimentation: potential biological effects of dredging operations in estuarine and marine environments. [Internet]. DOER Technical Notes Collection. Vicksburg (MS): US Army Engineer Research and Development Center.

ERDC TN-DOER-E20. [cited 2008 Jul 22]. 14 p. Available from: http://el.erdc.usace.army.mil/dots/doer/pdf/doere20.pdf.

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

Wilber DH, Clarke DG, Ray GL, Burlas M. 2003. Response of surf zone fish to beach nourishment operations on the northern coast of New Jersey, USA. Marine Ecology Progress Series 250:231-46.

Wilk SJ, Barr BW. 1994. Multiple-use issues in estuarine and coastal habitat loss. In: Langton RW, Pearce JB, Gibson JA, editors. Selected living resources, habitat conditions, and human perturbations of the Gulf of Maine: environmental and ecological considerations for fishery management. Woods Hole (MA): NOAA Technical Memorandum NMFS-NE-106. p 51-3.