OCS Study
BOEM 2013-0119

Review of Biological and Biophysical Impacts from Dredging and Handling of Offshore Sand

U.S. Department of the Interior
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Headquarters

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Review of Biological and Biophysical Impacts from Dredging and Handling of Offshore Sand

Prepared by

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LIST OF ABBREVIATIONS AND ACRONYMS

BACI	before-after, control-impact
BD	base depth
BMP	Best Management Practice
BOEM	Bureau of Ocean Energy Management
C	carbon
cm	centimeter
CPUE	catch per unit effort
CSA	Continental Shelf Associates
CSD	Cutter suction dredge
CV	coefficient of variation
dB	decibel
DEFRA	Department for Environment Food and Rural Affairs
DMM	discarded military munitions
E	Endangered
EA	Environmental Assessment
EIS	Environmental Impact Statement
EFH	Essential Fish Habitat
ESA	Endangered Species Act
FWC	Florida Fish and Wildlife Conservation Commision
g	gram
HAPC	Habitat Areas of Particular Concern
hp	horsepower
ICES	International Council for the Exploration of the Sea
Hz	hertz
Kg	kilogram
km	kilometer
LLD	lower listening depth
LME	Large Marine Ecosystem
m	meter
mg/L	milligrams per liter
µg/L	micrograms per liter
µPa	microPascal
mm	millimeter
MEC	munitions and explosives of concern
MMP	Marine Minerals Program
MMPA	Marine Mammal Protection Act
MMS	Minerals Management Service
NAEMO	Navy Acoustic Effects Model
NASA	National Aeronautics and Space Administration
nm	nautical miles
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NTU	nephelometric turbidity units
OCS	outer continental shelf

OM	organic matter
OSPAR	Oslo/Paris Convention for the Protection of the Marine Environment of the North-East Atlantic
PI	Principal Investigator
PTS	permanent threshold shift
RL	response level
rms	root-mean-squared
SAFMC	South Atlantic Fishery Management Council
SPL	sound pressure level
T	threatened
TSHD	Trailing suction hopper dredge
TTS	temporary threshold shift
ULD	Upper listening depth
USACE	U.S. Army Corps of Engineers
USFWS	U.S. Fish and Wildlife Service
USWTR	undersea warfare training range
UXO	unexploded ordnance
yd^3	cubic yard

1.0 INTRODUCTION

1.1 Background of the Study

The Bureau of Ocean Energy Management (BOEM) is charged with environmentally responsible management of Outer Continental Shelf (OCS) resources (e.g., oil and gas, sand and gravel, renewable energy). Federal jurisdiction starts at 3 nautical miles (nm) offshore of most states, with the exception of Texas and the Gulf coast of Florida, where it starts at 9 nm. Public Law 103-426 (43 U.S.C. 1337(k)(2)), enacted 31 October 1994, gave the Minerals Management Service (MMS) (now BOEM) the authority to negotiate, on a noncompetitive basis, the rights to OCS sand, gravel, and shell resources for shore protection, beach or wetlands restoration projects, or for use in construction projects funded in whole or part by, or authorized by, the federal government. As of January 2013, BOEM has conveyed rights to about 73 million cubic yards of OCS sand for 38 coastal restoration projects in six states.

As the demand for OCS sand increases, the BOEM Marine Minerals Program (MMP) is facing increasingly complex issues, such as resource allocation, cumulative impacts from repeated use, fisheries conflicts, protection of archaeological sites, oil and gas infrastructure, renewable energy infrastructure, and essential fish habitat issues, among others. It is critical that BOEM uses the best available science in their environmental assessments of proposed leases, so that all necessary and effective precautions are taken to reduce potential impacts during sand dredging and conveyance to the placement site.

This report provides a summary of the current *state-of-the-knowledge* of the likely impacts of OCS sand dredging and conveyance operations to biological resources and their habitats and rates of habitat recovery post-dredging. Furthermore, we synthesize dredging guidelines and recommended practices to minimize impacts and speed habitat recovery, and mitigation measures to reduce or eliminate adverse impacts to specific valued resources, such as marine mammals, sea turtles, and fishes. Every lease issued by BOEM includes specifications in the form of mitigation measures to reduce or eliminate adverse environmental impacts that were identified during the environmental review and consultations under the Endangered Species Act (ESA), the Magnuson-Stevens Fishery Conservation and Management Act, and the Coastal Zone Management Act. Although mitigation strategies are implemented, there is little information, based on rigorous collection of quantitative data, on the effectiveness of their intended purpose. It is important to have the scientific basis to show that these requirements are effective.

1.2 Study Objectives

The objectives of this study were to:

1. Review and synthesize relevant environmental research that analyzes the biological effects of and effect-reducing mitigation used in dredging and conveyance operations in the marine environment. This includes reviewing environmental studies sponsored by the BOEM MMP, as well as major and recent domestic and international research. Resource categories included:
    - Benthic communities and habitats within and adjacent to borrow areas and their trophic connections to nektonic communities

- Fishes and essential fish habitat within and adjacent to borrow area
- Foraging seabirds
- Threatened and endangered species at risk (and designated critical habitats):
  - Cetaceans (baleen whales and toothed odontocetes)
  - Sirenians (West Indian manatee)
  - Sea turtles (all species that occur in the vicinity of borrow areas)
  - Staghorn and elkhorn corals (discussed under benthic communities and habitats)

Impact-driving mechanisms included:

- Alteration of benthic habitat at the borrow areas
- Increased turbidity in the water column
- Increased sedimentation/deposition on the seafloor
- Pumping/entrainment near the seafloor
- Sound
- Vessel operations
- Water quality (including accidental spills)
- UXO, shipwrecks, other hard structures temporarily exposed during dredging

2. Identify specific knowledge gaps that may exist and recommend new studies to address the major gaps, for both potential impacts and the efficacy of mitigation measures

1.3 STUDY METHODS

1.3.1 Literature Search

Research staff at RPI and the Principal Investigators (PIs) performed literature searches for each individual discipline included in the project. PIs also made information requests to colleagues as well as points of contacts provided by BOEM. The disciplines included were: Types of OCS Sand Borrow Areas; OCS Dredging and Conveyance Methods and Potential Impacts; Benthic Resources, Fishes and Fish Habitats, Sea Turtles, Marine Mammals; and Foraging Seabirds. Staff from the MMP provided copies of applicable studies conducted for BOEM, Environmental Impact Statements (EISs), Environmental Assessments (EAs), Biological Assessments and Opinions, Essential Fish Habitat (EFH) Assessments and Conservation Recommendations, and a vast amount of other peer-reviewed and grey literature to the research team.

Initial “first cut” searches were conducted, compiled, and submitted to the PIs responsible for writing the section associated with that particular discipline. The PIs, as experts in their respective fields, exercised their professional judgment to determine the appropriateness of each document. Efforts were made to specifically narrow the scope of the synthesis review to literature that would provide a high value on each of the specified disciplines and most relevant topics. The PIs also helped identify additional information sources, such as selected peer-reviewed articles, grey literature, reports, unpublished theses and dissertations, and spatial information. The literature was downloaded from online sources, requested from peers, state agencies, and federal agencies, acquired through academic library resources, or provided by each of the PIs. The primary databases used in literature searches included the following:

- U.S.D.A. National Agricultural Library (NAL, or Agricola)
- CAB Abstracts http://www.ovid.com/site/catalog/DataBase/31.jsp
- CSA Environmental Pollution and Management Database
- GEOBASE
- Google Scholar
- U.S. Census Bureau
- U.S. Geological Survey Publications Warehouse
- Social SciSearch via Web of Science
- Web of Science

1.3.2 Annotated Bibliography

Of the extensive literature reviewed, only those documents that were considered to be of value and cited in the report were compiled in an electronic annotated bibliography using EndNote® software. Each discipline was included as a separate database as a subset of the master database. Each record in the database contains the complete citation. In addition, PDF files of non-copyrighted articles were attached to the appropriate records. Links to online PDFs were also included as appropriate. EndNote® can be queried by searching on: name, title, authors, date, publisher, journal/periodical, keywords, or any combination thereof. The database contains 453 records.

1.4 STRUCTURE OF THE REPORT

This report is divided into five chapters as summarized below:

- Chapter 2 is a brief summary of the types of OCS sand borrow areas and current guidelines and recommended practices for dredging OCS sand. It includes a series of figures showing the bathymetry of five different types of borrow areas before and after a dredging event.
- Chapter 3 describes the types of dredging and conveyance activities and the potential environmental effects by each impacting mechanism.
- Chapter 4 includes summaries of the literature for each resource and impacting mechanism and mitigation methods and effectiveness, and resource-specific data gaps and recommendations for studies to address these data gaps.
- Chapter 5 is a summary of the major data gaps and recommendations for studies to address these data gaps.
- Chapter 6 includes all of the references cited.

2.0 TYPES OF OCS SAND BORROW AREAS

2.1 SURFICIAL SAND DEPOSITS

BOEM currently leases OCS sand along the Atlantic and Gulf of Mexico coasts. Surficial sand deposits on the OCS that are being accessed in these regions occur as: 1) broadly spaced ridges separated by low swales on the open shelf that are isolated from the shoreface, referred to in this report as ridge and swale complexes, such as off Maryland, Virginia, and the east and west coast of Florida; 2) large ebb-tidal deltas associated with major tidal inlets, such as the inlet to Mobile Bay; 3) inner-shelf sand shoals (e.g., reworked barrier island), such as Ship Shoal off Louisiana; 4) low-relief sand ridges and sand sheets between hard-bottom habitats, such as off Myrtle Beach, South Carolina and Broward County, Florida.

The long axis of ridge and swale complexes is oriented directly into the prevailing or storm wave direction. For example, the ridges in the mid-Atlantic OCS (Figure 2.1) orient to the northeast due to the large waves during “nor’easters,” whereas in Alabama, they orient to the southeast due to the prevailing and storm waves from the southeast. The ridges have relief of 3-12 meters (m), are in 5-20 m water depths, and can be tens of kilometers (km) long. The sediments range from fine to coarse sand, and the surfaces are covered with ripples to larger sand waves, indicating that they are being reworked by wave action. These complexes have a wide range of gradients in relief, sediment texture, roughness elements, flow dynamics, and benthic composition and diversity of the ridge versus the swale habitats. Continental Shelf Associates Inc. (CSA) et al. 2010) and Dibajnia and Nairn (2011) provide detailed information on the origin and dominant processes for ridge evolution and maintenance.

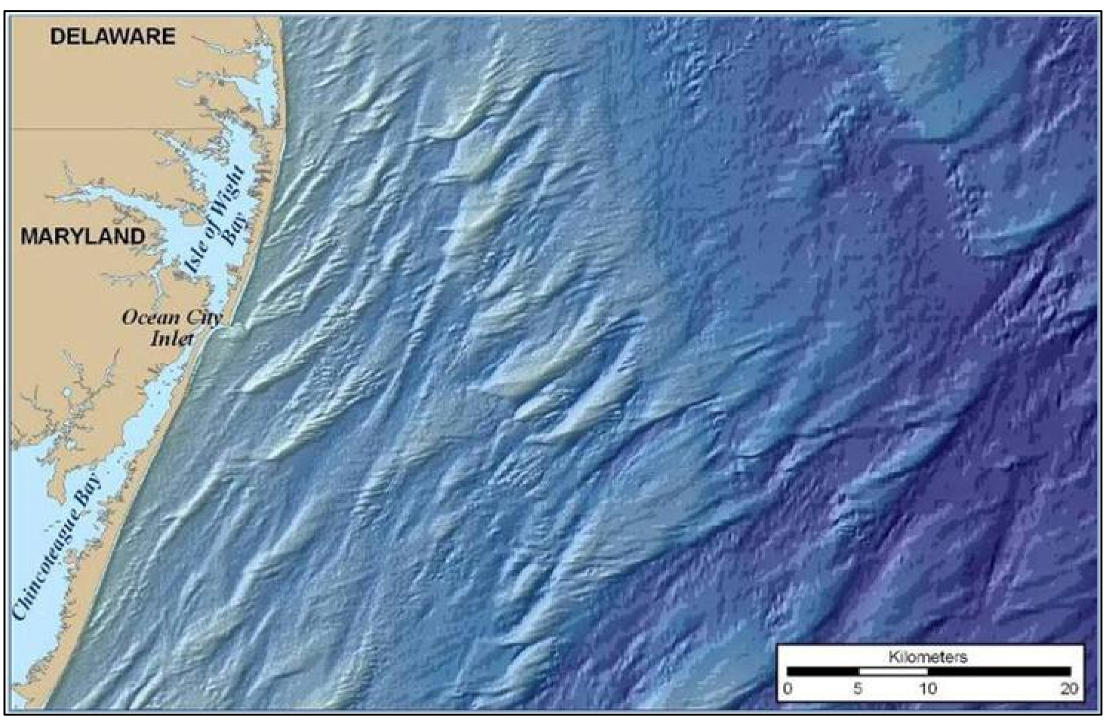


Figure 2.1 Regional bathymetric map showing the classic ridge and swale topography on the mid-Atlantic continental shelf, Maryland and Delaware. From CSA et al. (2010).

Ridge and swale complexes comprise very large potential sources of sand for shoreline protection, from New Jersey to Alabama. Geo-Marine, Inc. (2010) identified at least 35 such features, including some very large complexes offshore New Jersey. Dibajnia and Nairn (2011) mapped 7 off Delaware, 50 off Maryland, and 124 off northern Virginia (to the north entrance to the Chesapeake Bay). They continue on the shelf to central Florida, then again from west Florida to Alabama. Figure 2.2 shows their density off the Alabama coast.

Large ebb-tidal deltas extend into federal waters in some areas, such as the mouth of Mobile Bay (Figure 2-2). Ebb-tidal deltas are active components of the littoral transport system; sand moving alongshore circulates between the ebb-tidal delta and the adjacent beaches in complex patterns for a period of time before bypassing the inlet and continuing its longshore transport (Fitzgerald et al. 1976). Thus, ebb-tidal deltas are not often considered appropriate sources for dredging of OCS sand.

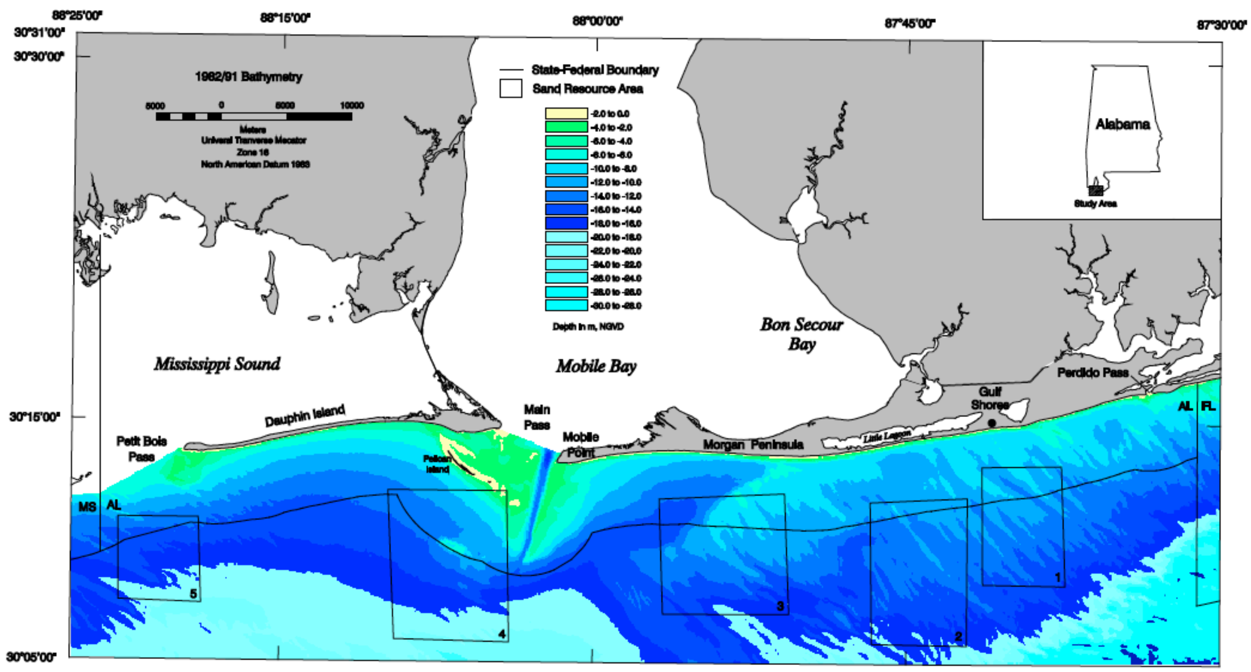


Figure 2.2 Potential borrow areas (boxes) offshore Alabama, showing the large ebb-tidal delta associated with the mouth of Mobile Bay. The blue line delineates the boundary between state and federal waters. From Byrnes et al. (2004).

In the Gulf of Mexico, ridge and swale features are relatively sparse west of Alabama; instead, geomorphic sand features transition to isolated, large-scale sand bodies such as St. Bernard Shoals, Ship Shoal, and Tiger/Trinity Shoal off Louisiana, and Sabine Bank and Heald Bank off Texas. For example, Ship Shoal is about 50 km long and between 4-19 km wide (Figure 2.3). It is 5-7 m above the surrounding seafloor and water depths on the shoal range from 3-8 m. The upper 4 m of the shoal have been reworked into well-sorted, fine-grained sand, making it an important sand source for coastal restoration projects in Louisiana. It is different from many of the other OCS sand resources in the Gulf of Mexico, because of several factors. First, there is extensive oil and gas infrastructure crisscrossing the shoal; thus many are currently not available for use as sand borrow areas. In winter and spring, due to influx of fine-grained fluvial sediment from the Atchafalaya River and a change in sediment transport patterns, a transient fluid mud layer 10-15 cm thick covers the eastern flank (Kobashi et al. 2007). Also, during summer, it is within the area of extensive hypoxia (Rabalais et al. 2001b).

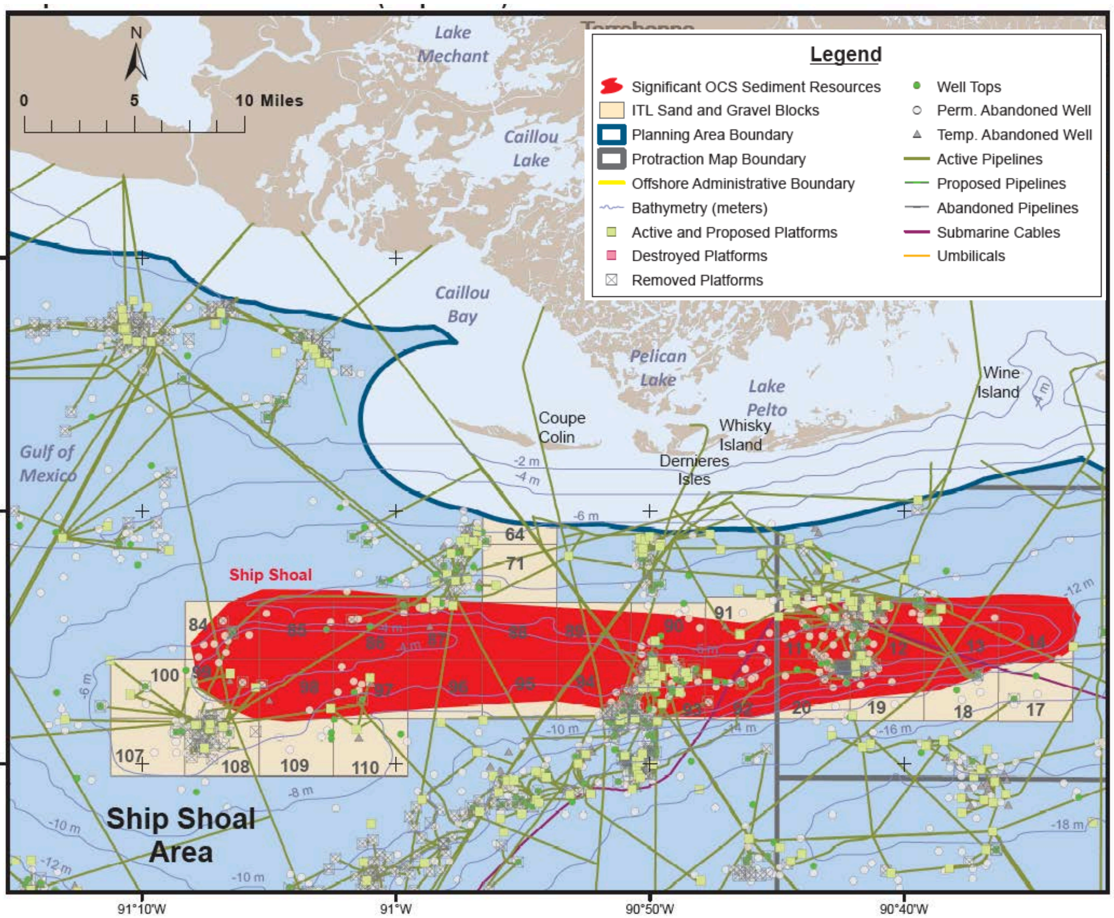


Figure 2.3 The extensive oil and gas infrastructure on Ship Shoal. From the BOEM 2012 Notice to Lessees and Operators concerning Significant OCS Sediment Resources on Ship Shoal, recommending avoidance of these sediment resources to the maximum extent practicable.

The sand resources offshore of Myrtle Beach, South Carolina (which has used OCS sand three times since 1996; see Table 4.29) consist of relatively thin (<3 m) relict deposits of fine to coarse sand in complex patterns based on their origin as relict inlet shoals or shallow, filled channels (Denny et al. 2005; Figure 2.4). The thicker deposits are associated with modern inlets, such as Murrell's Inlet and North Inlet. There are extensive areas of relatively flat hard ground, with a thin coarse sediment veneer, indicating that the inner shelf has undergone long-term erosion.

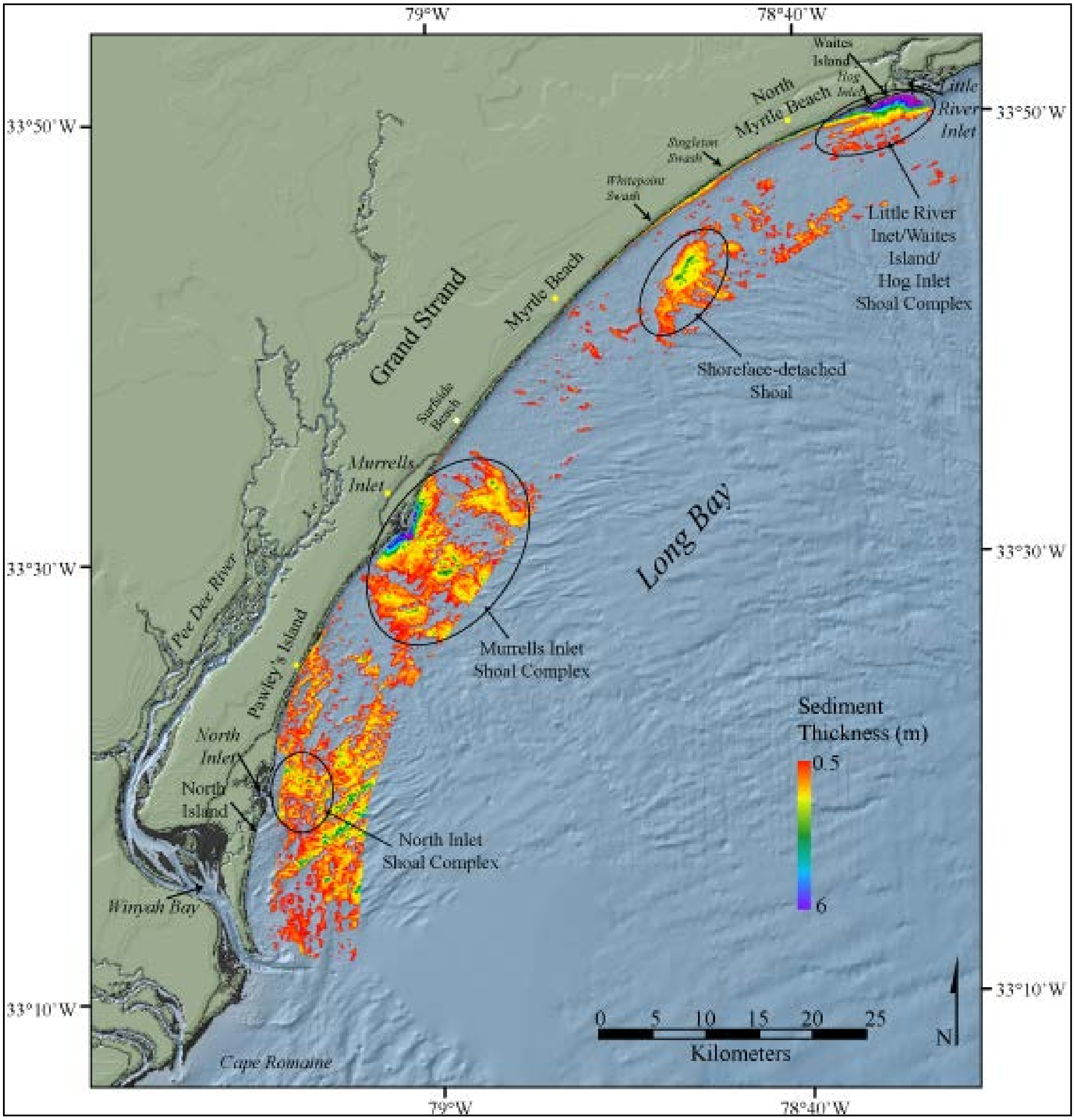


Figure 2.4 Map of the thickness of Holocene sediment in Long Bay, South Carolina. Inlet shoal complexes and shore-detached shoals are outlined. Areas where the seafloor bathymetry is visible indicate areas of surficial sediment < 0.5 m thick. From Denny et al. (2005).

2.2 BURIED SAND DEPOSITS

Most buried sand deposits in the OCS occur as paleofluvial channels (low-stand valley fills) and sand sheets that were deposited on the shelf during periods of lower sea level and are now capped mostly by mud. Nairn et al. (2007) provided a comprehensive summary of the origin, sediment characteristics, likely thickness of fine-grained overburden, and a state-by-state summary of potential sand sources in low-stand valley fills. The thickness of the fine-grained overburden is a key factor in determining the economic feasibility of extracting buried deposits.

Figure 2.5 shows the major low-stand valleys on the inner shelf of eastern Texas and western Louisiana.

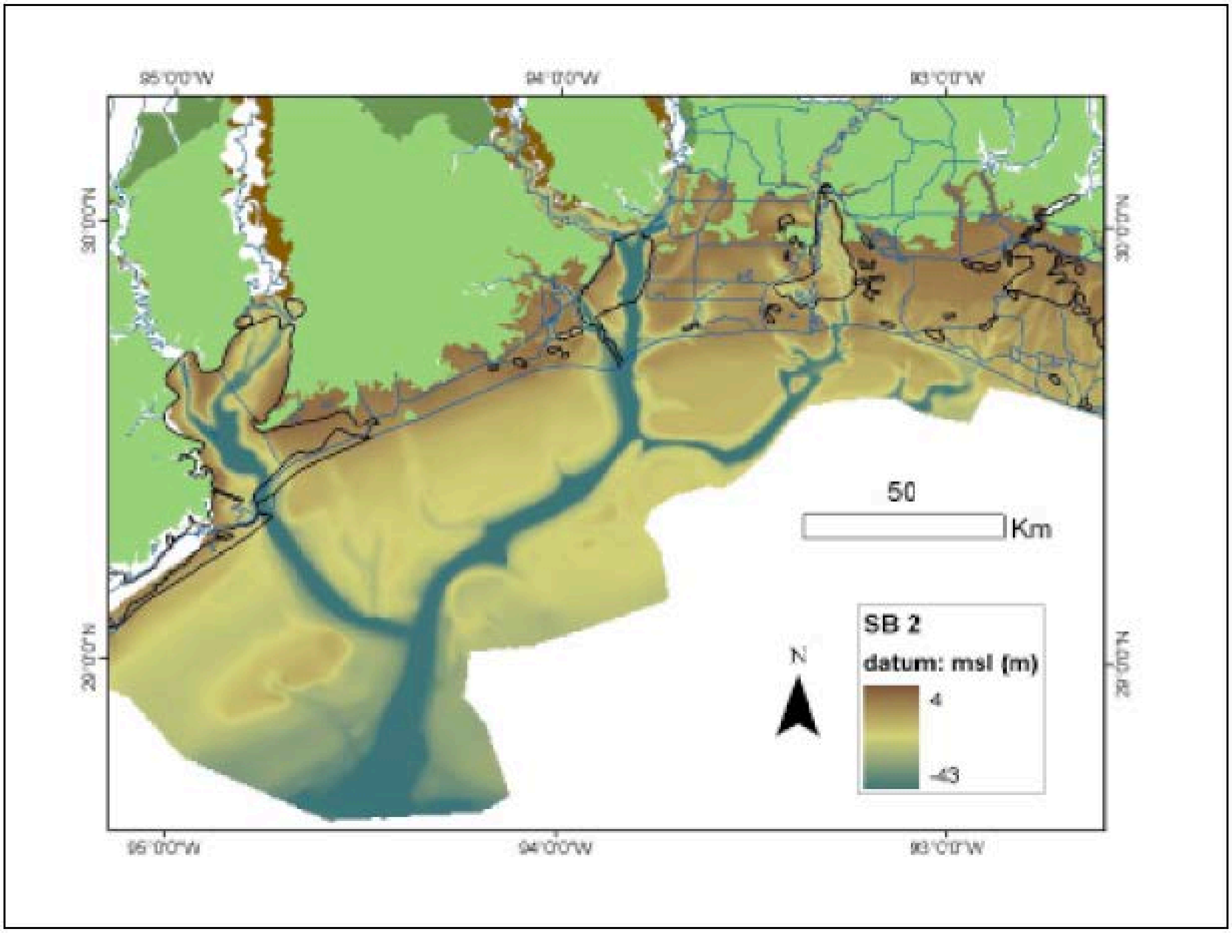


Figure 2.5 The incised river valleys of the eastern Texas and western Louisiana continental shelf. Studies have found that the fluvial sands lie beneath 10-20 m of marine and bay mud. From Anderson et al. in Nairn et al. (2007).

BOEM has issued leases for OCS sand from three buried OCS borrow areas as of January 2013, all off Louisiana: Peveto Channel (Holly Beach); Sandy Point borrow area (Pelican Island; and Raccoon Island borrow area (Raccoon Island). There have been extensive field surveys to evaluate the potential volumes of beach-quality sand from buried channels on the Texas inner shelf by researchers at Rice University (Anderson et al., Appendix A2 in Nairn et al. 2007).

To access the sand, the overlying fine-grained sediments have to be stripped away, thus generating relatively deep pits: the Holly Beach borrow pit was 8 m deep; the two Sandy Point borrow areas involved removal of up to 5 m of overburden then removal of up to 10 m of sand, creating pits up to 12 m deep. Cutterhead suction dredges (CSDs) are usually used to remove the fine-grained overburden, which is placed at a nearby in-water disposal area; they are also better able to remove sand from a confined pit area.

2.3 GUIDELINES AND RECOMMENDED PRACTICES FOR DREDGING OFFSHORE SAND SHOALS

In some regions, offshore sand shoals are being considered as sources to meet long-term sand needs. For example, the U.S. Army Corps of Engineers (USACE) (2008b) evaluated the impacts of dredging several offshore shoals to provide sand for the Atlantic Coast of Maryland Shoreline Protection Project for the years 2010-2044. Between 5,200,000 and 11,500,000 m^3 of sand would be needed through 2044, depending on the frequency of future storms. As part of the regulatory consultations with federal and state agencies, the USACE (2008a) agreed that no more than about 5% of the total volume of any shoal in this project should be dredged as a precautionary principle. It is noted that this limitation will not be feasible in other areas. The National Aeronautics and Space Administration (NASA) plan to protect the Wallops Island facility requires an initial placement of 2,400,000 m^3 (completed in 2012) and approximately 616,000 m^3 every five years (NASA 2010c).

As discussed under the different resources in Section 4, little is known about the ecological importance of offshore ridge and swale complexes (individually and regionally). Until more is known about their habitat functions and importance, the approach has been to identify and implement dredging guidelines to maintain the geomorphic integrity of shoals, to the extent possible. BOEM funded two modeling studies to assess the responses to different dredging scenarios for shoals in the Mid-Atlantic region off Maryland. The CSA et al. (2010) study looked at changes after two different three-day storm events for three shoals using a half-plane wave model. Dibajnia and Nairn (2011) ran a full-plane wave model using measured wave and current data for 2007-2008, but only for the 258 hours with bed shear stress above the threshold for which sediment transport was likely. They then multiplied sediment transport rates by a factor of ten, which translated into predicted changes in morphology over a thirteen-year period. Both studies modeled the response of shoals for a baseline condition (no dredging) as compared to removal of different volumes of sediment in different configurations from different locations on the shoals. Both reports include specific dredging guidelines to maintain the shoal's integrity, as summarized below. However, it is important to note that, as of 2013, these recommendations remain untested and have not been evaluated for technical or cost feasibility. Also, the recommendations are similar in some areas, but different in other areas.

CSA et al. (2010) recommended the following procedures to dredge shoal and ridge features that will minimize ecological impacts and/or speed recovery:

- Extracting sand from a depocenter, leading edge, or downdrift margin of a shoal, to avoid interrupting natural shoal migration and potentially reduce the time required for borrow area refilling or equilibrium;
- Avoid dredging in erosional areas that supply downdrift depocenters, which also may be slow to refill after dredging;
- Shallow dredging over large areas rather than excavating small but deep pits may be preferred depending on the infaunal characteristics;
- Dredging in strips to leave sediment sources adjacent to and interspersed throughout target areas, leading to a more uniformly distributed infilling process; and
- Excavation should occur on shoal crests and higher areas of the leading edge rather than

lower areas of the shoals because of greater exposure to wave-generated turbulence and greater sediment mobility, which potentially would result in more rapid sediment reworking and area infilling, and harbor a benthic community capable of recovering more rapidly.

Dibajnia and Nairn (2011) identified the following key processes controlling shoal morphology and modeled how shoals in the Mid-Atlantic region could morphodynamically respond after dredging:

- Waves are the primary factor in shoal growth and maintenance, whereas currents are more responsible for shoal migration, increasingly so as water depth increases;
- The shoals migrate at a rate of a few meters per year, which is very small compared to overall shoal dimensions. (It is noted that migration occurs more rapidly during storms; most of the time, wave and current forcing maintains a dynamic equilibrium);
- Upon removal of material from a shoal, the shoal gradually reforms into a shoal of similar geometry and character but with a smaller volume due to removal of the sediment. The volume taken by dredging is not completely replaced by transport of material from outside of the shoal (there may be some along-shelf transport);
- Despite the reduction in volume, the model results for some of the dredging scenarios indicated that the reformed shoal would have the same height as that of the pre-dredge shoal conditions; and
- Although shoals get volumetrically smaller as a result of dredging, there was no indication of possible shoal diminishing/deflation after dredging.

Dibajnia and Nairn (2011) recommended the following guidelines for design of borrow areas and dredging practices for offshore sand shoals in the area offshore Delaware, Maryland, and Virginia between Delaware Bay and Chesapeake Bay:

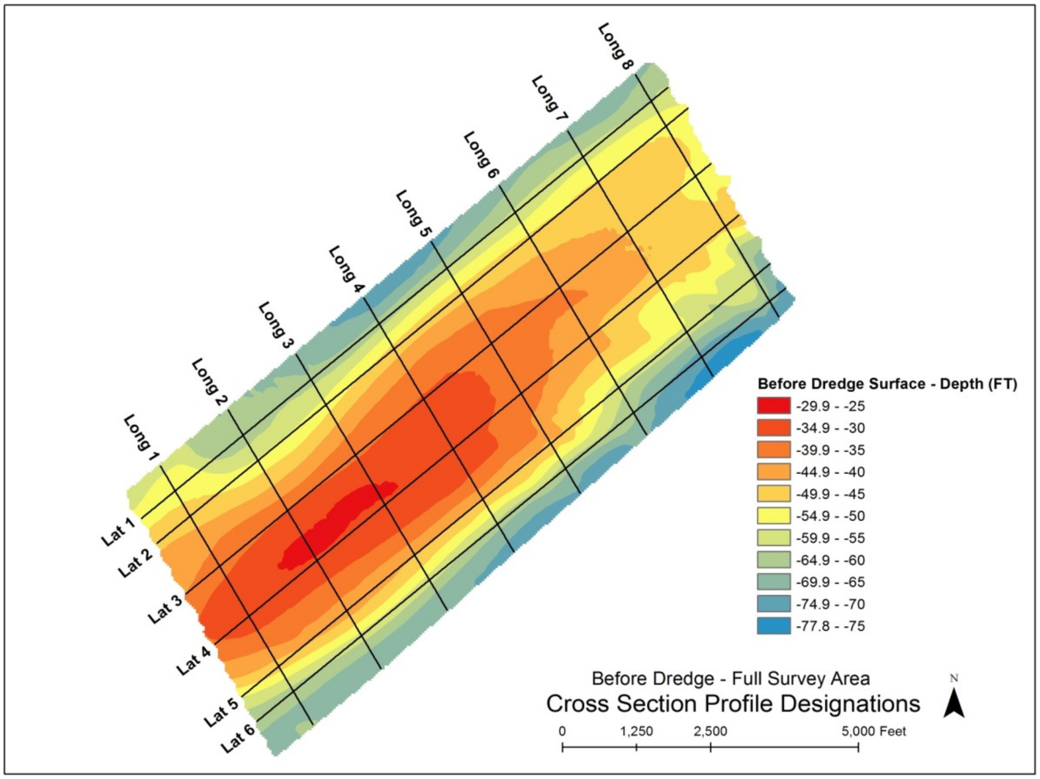
- The final dredging approach should be determined based on suitability of the dredged sand for nourishment, as well as ecosystem services associated with the reformed shoal shape. A determination is required regarding the importance of maintaining the pre-dredge shoal height from an ecological perspective;
- Only those shoals located in less than 30 m depth are predicted to have greater potential to re-grow after dredging and, therefore, shoals with a Base Depth (BD; the depth to the seafloor at the base of the shoal) of greater than 30 m should not be dredged if maintaining the pre-dredge shoal height is determined to be important from an ecological perspective;
- Shoals with Relative Shoal Height (defined as H/BD) of less than 0.5 are not likely to recover after dredging. Therefore, shoals with H/BD of less than 0.5 should not be dredged if shoal recovery to its pre-dredge height is desired;
- The maximum H/BD_{max} of the shoals in the mid-Atlantic OCS varied from 0.5 at 10 m depth to 0.75 at 20 m depth. A shoal that has reached the maximum Relative Shoal Height corresponding to its BD may be considered as a fully grown shoal at that depth. A fully grown shoal (in height) can potentially re-grow to the same height after being dredged. Therefore, if shoal recovery to its pre-dredge height is desired, shoals that have reached their maximum relative shoal height are recommended for dredging. In the mid-

Atlantic region, maximum Relative Shoal Height at a certain BD may be estimated as: $(H/BD)_{max} = (BD-5)/BD$;

- Sand should not be removed from the entire length of the shoal, to the extent possible. Longitudinal dredging (i.e., dredging all along the longer axis) is not preferred because it affects wave-focusing processes and, as a result, the shoal is much less likely to recover to the same pre-dredge height;
- For mid-Atlantic shoals, they recommended dredging sand from the leading edge (SW side) of a shoal. This is because 1) wave focusing is concentrated on the trailing edge (NE side) of a shoal, and 2) overall shoal migration is towards the southwest. Therefore, after removal of material from the SW side of a shoal, a new shoal crest can be formed over the excavated area by transport of material from the NE trailing edge;
- Dredging from shoal flanks below the -10 m contour over the SW half of the shoal is expected to have little effect on shoal integrity and little change is anticipated to happen to the dredged area. This dredging option is thus recommended if it can provide sand suitable for nourishment;
- The proposed guidelines are not universal and are dictated by the local storm wave height, storm wave direction, and storm-related subtidal currents; and
- Similar guidelines are expected to apply to shoals in areas other than the mid-Atlantic region. Details, however, would be dictated by local wave and current conditions. It is recommended that a similar study be completed for other regions when the ecological role of the shoal height/shape is very important, to justify the associated study cost.

It is important to acknowledge the last point made by Dibajnia and Nairn (2011), who voiced caution in the application of these guidelines to other areas. During the EFH consultation with the National Marine Fisheries Service (NMFS) for the Wallops Island, Virginia project, NASA (2010a) noted that the targeted Shoal A had notably different properties and patterns of change, compared to the Isle of Wright shoal on which the more specific Dibajnia and Nairn (2011) guidelines were developed. The dredging guidelines proposed by NASA (2010a) applied many of the guiding principles to Shoal A, such as dredging depositional areas along the leading edge, maximum depth of cut at 3 m, and dredging from shoal flanks. Figure 2.6 shows the before and after dredging bathymetry of Shoal A, indicating that these guidelines were followed.

Zarillo and Zarillo (2011) conducted a modeling study of seven ridge and swale complexes off central Florida with crest elevations of -11 to -18 m in water depths of -15 to -22 m. Over a two-year period, the shoals were reworked only during severe storms, with 0.2-1 m of topographic change over the crest areas of the shoals. Clearly, each group of ridge and swale complexes would be affected differently by waves, currents, and dredging patterns. However, researchers are building on their understanding of the processes that affect offshore ridge and swale complexes and how they might respond to different dredging methods. In addition, pre- and post-dredge bathymetry, particularly over time, will provide real data for model validation and refinement of dredging practices in the longer term.



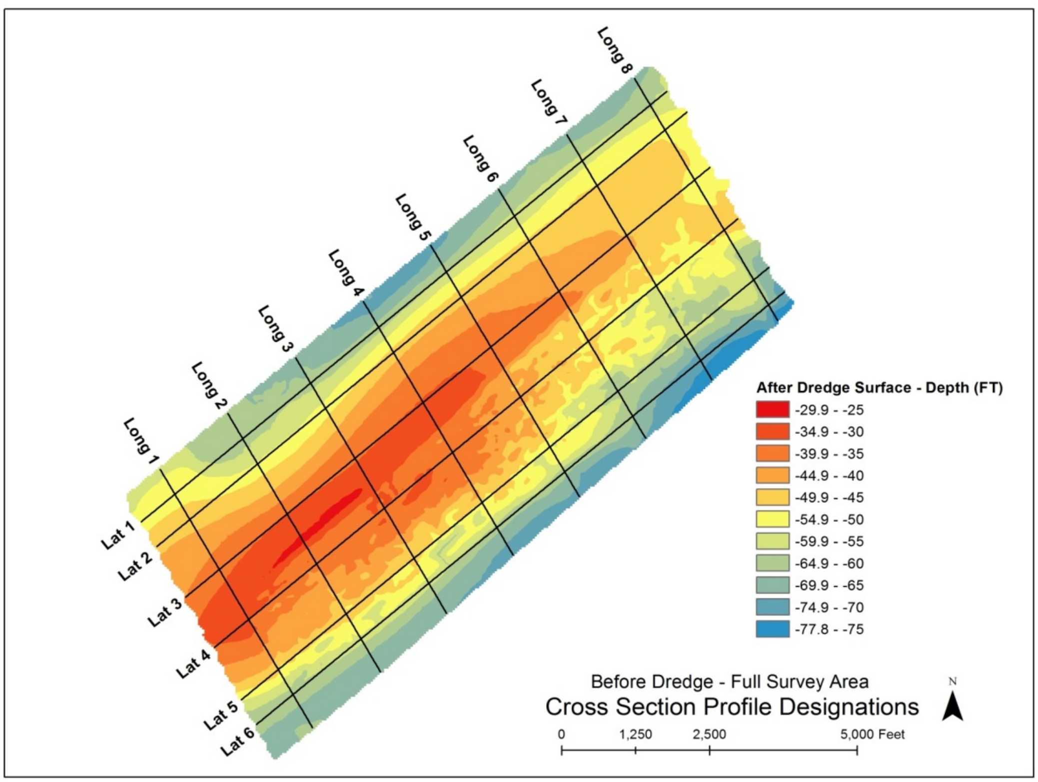


Figure 2.6 Wallops Island dredging project, Shoal A off Virginia. Top: Before dredging bathymetry (31 March 2012). Bottom: After removal of about 3.2 million yd^3 (17 August 2012). Most of the sand was removed from the southeastern flank. The maximum cut was about 3 m. From Brown (2013).

2.4 POST-DREDGING BATHYMETRIC MAPS OF REPRESENTATIVE BORROW AREAS

Post-dredging bathymetric surveys are required to confirm the location and volumes of sand removed for each project. BOEM developed elevation models from the bathymetric data for five representative OCS dredging projects. Figure 2.7 shows the post-dredging bathymetry for these five projects plotted at the same horizontal scale (1:2,500) and vertical exaggeration (10:1) to allow comparison of their spatial extent and depth of sediment removal. Table 2.1 provides the details for each project, showing the wide range in sand removal volumes, with Wallops Island being the largest and Dade County being the smallest. Hopper dredges were used for all but the Pelican Island project, which used a cutterhead suction dredge.

Table 2.1
Project data for the five dredging projects shown in Figure 2.7.

Project/Date	Borrow Area	OCS Sand Resource Type	Sand Volume (yd^3)
Wallops Island, VA/2012	Shoal A	Ridge and swale complex	3,200,000
Myrtle Beach, SC/2007	Surfside	Low-relief sand ridges	778.600
Duval County, FL/2011	Duval Borrow Area A	Low-relief sand ridges	1,200,000
Dade County, FL/2012	Southern Government Cut	Low-relief sand sheet	474,000
Pelican Island, LA/2012	Sandy Point	Buried paleochannel	2,200,000

The detailed elevation models for each project in Figures 2.8 through 2.12 are presented at different horizontal scales and vertical exaggerations to optimize their view. Each figure consists of three presentations: A) shaded-relief map that also shows the location of the topographic cross section; B) oblique view of the sonar imagery draped over the 3D bathymetry color-coded to show depth contours; and C) a cross section along the transect shown in A. During review of each of the specific projects, it is important to refer back to Figure 2.7 for a perspective of the relative scale of each project.

The Wallops Island borrow area is a sand ridge with the crest at about -30 m water depth (Figure 2.8). Sand was removed from the shoal flank with a maximum cut of 3 m for this very large project. The resulting surface is clearly irregular with several meters of relief. The dredging pattern is very consistent, with straight, parallel tracks that allow for very efficient dredging.

The Myrtle Beach borrow area consists of low-relief sand ridges between areas of exposed hard ground (Figure 2.9). The bathymetry was mapped using single-beam sonar so the resolution is coarser. The cross section in C includes data collected shortly after dredging and about one year later, showing the rate and areas of infilling in this relatively sand-poor offshore area. Infilling is highly variable depending on location, sediment supply, hydrodynamics, and sediment transport patterns.

The Duval County borrow area (offshore of Jacksonville, Florida) consists of a sand ridge field (Figure 2.10). The 2011 project removed sand from the central area with the east-west

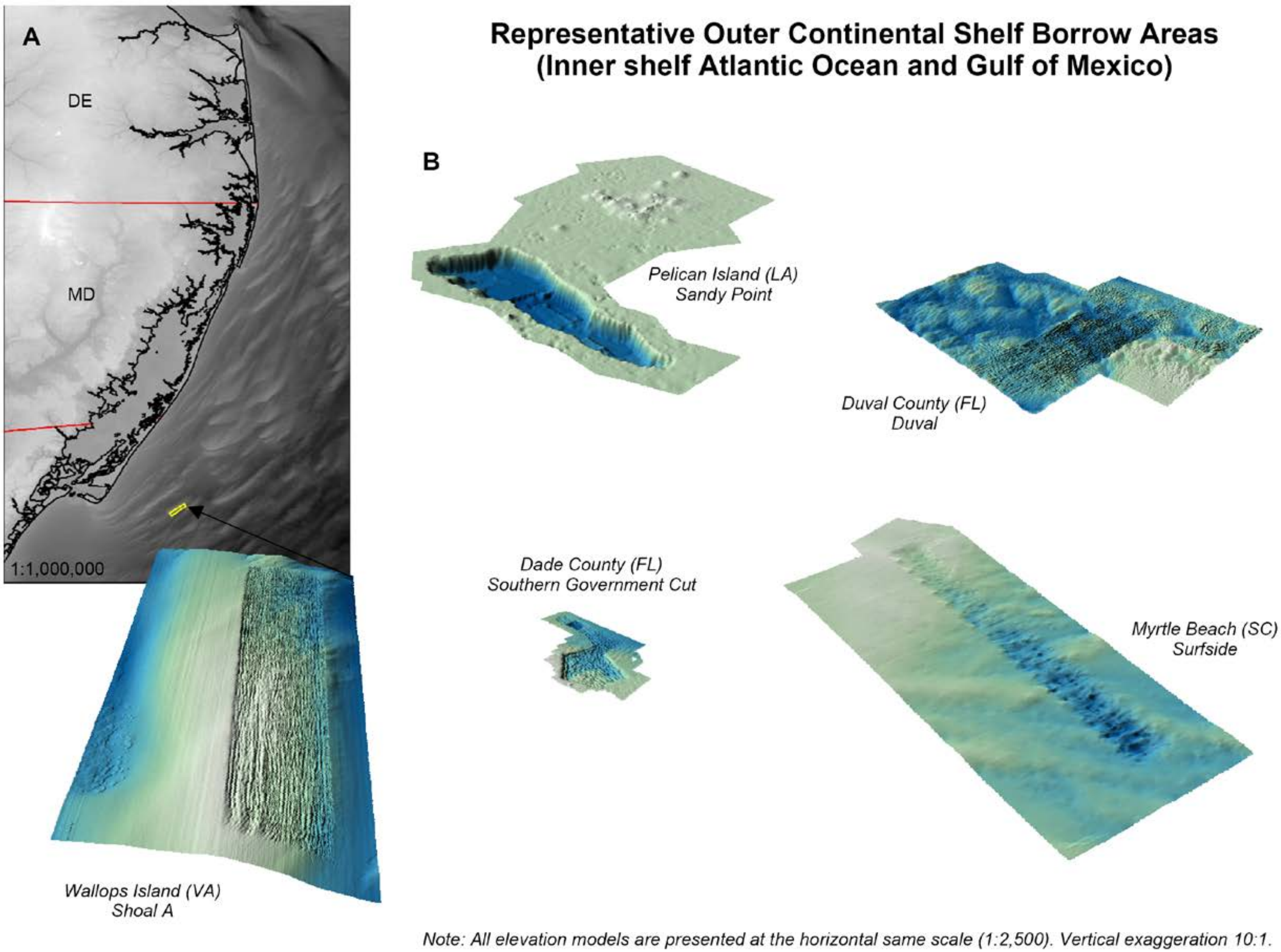


Figure 2.7 Digital elevation models of the post-dredging bathymetry for five OCS borrow areas representing the range of borrow area types.

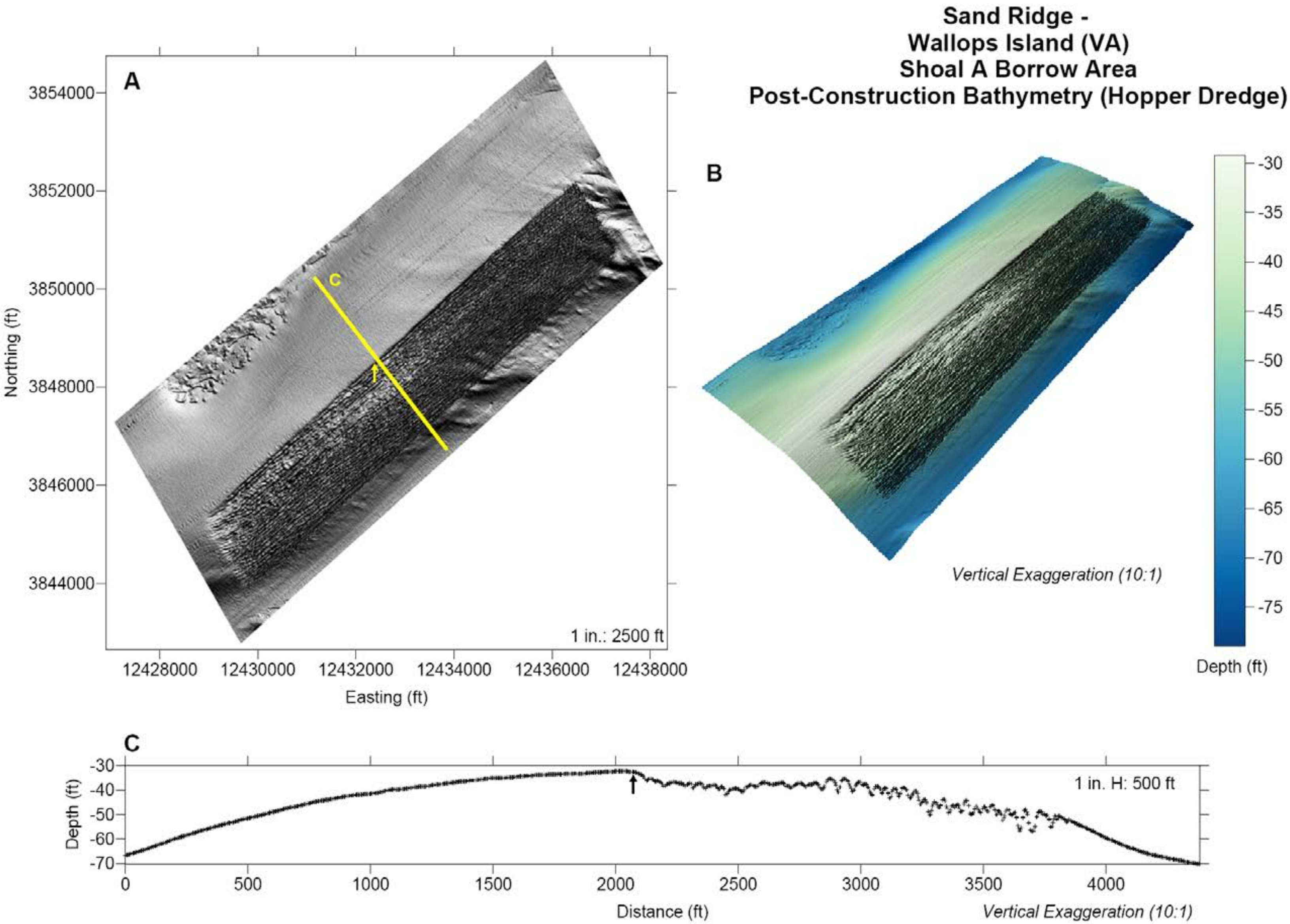


Figure 2.8 Digital elevation models of Shoal A used for the Wallops Island, Virginia project in 2012.

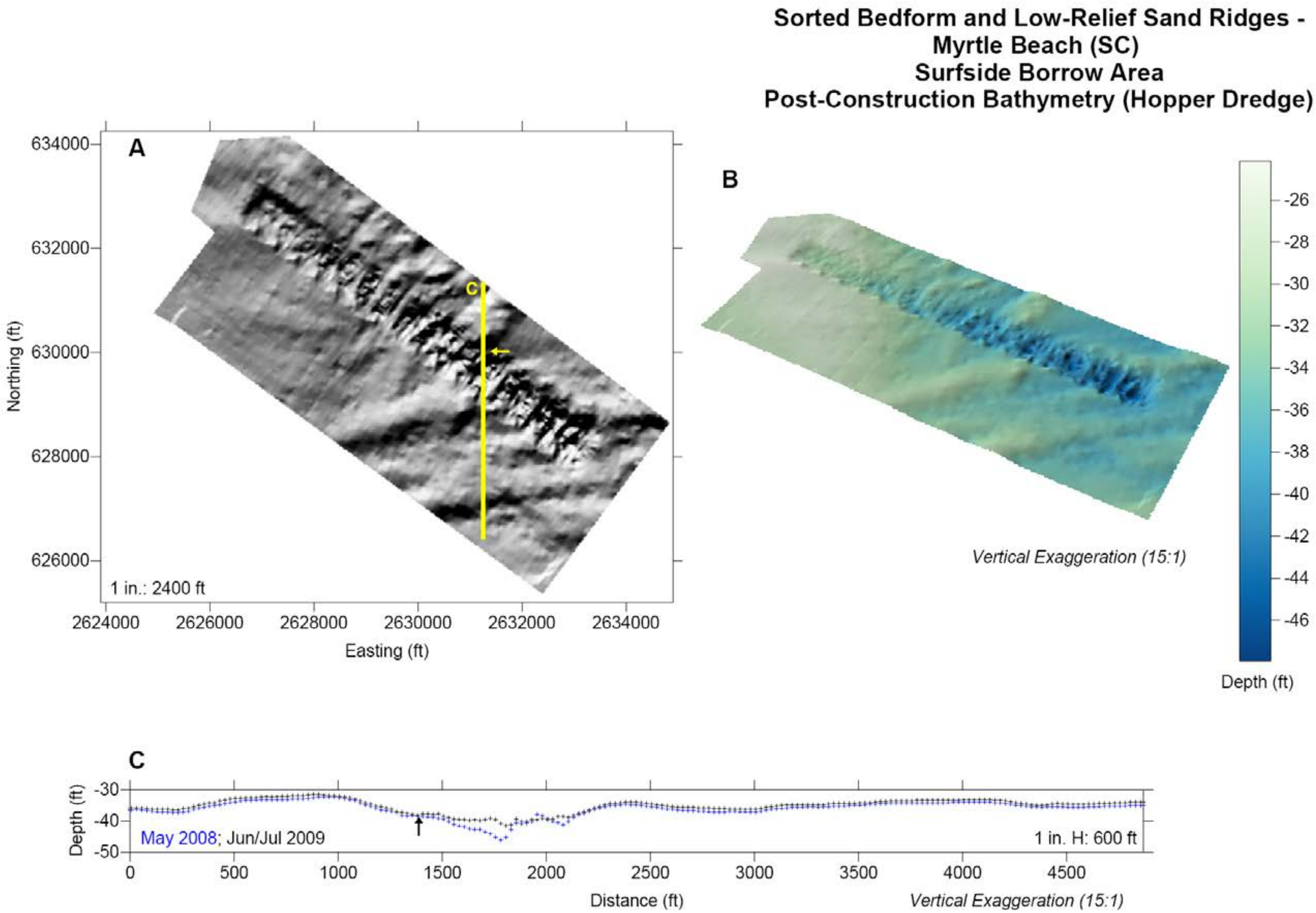


Figure 2.9 Digital elevation models of the Surfside borrow area (single-beam sonar) used for the Myrtle Beach, South Carolina, project in 2007. Note that the cross section in C shows data for both immediately after dredging and about one year post-dredging.

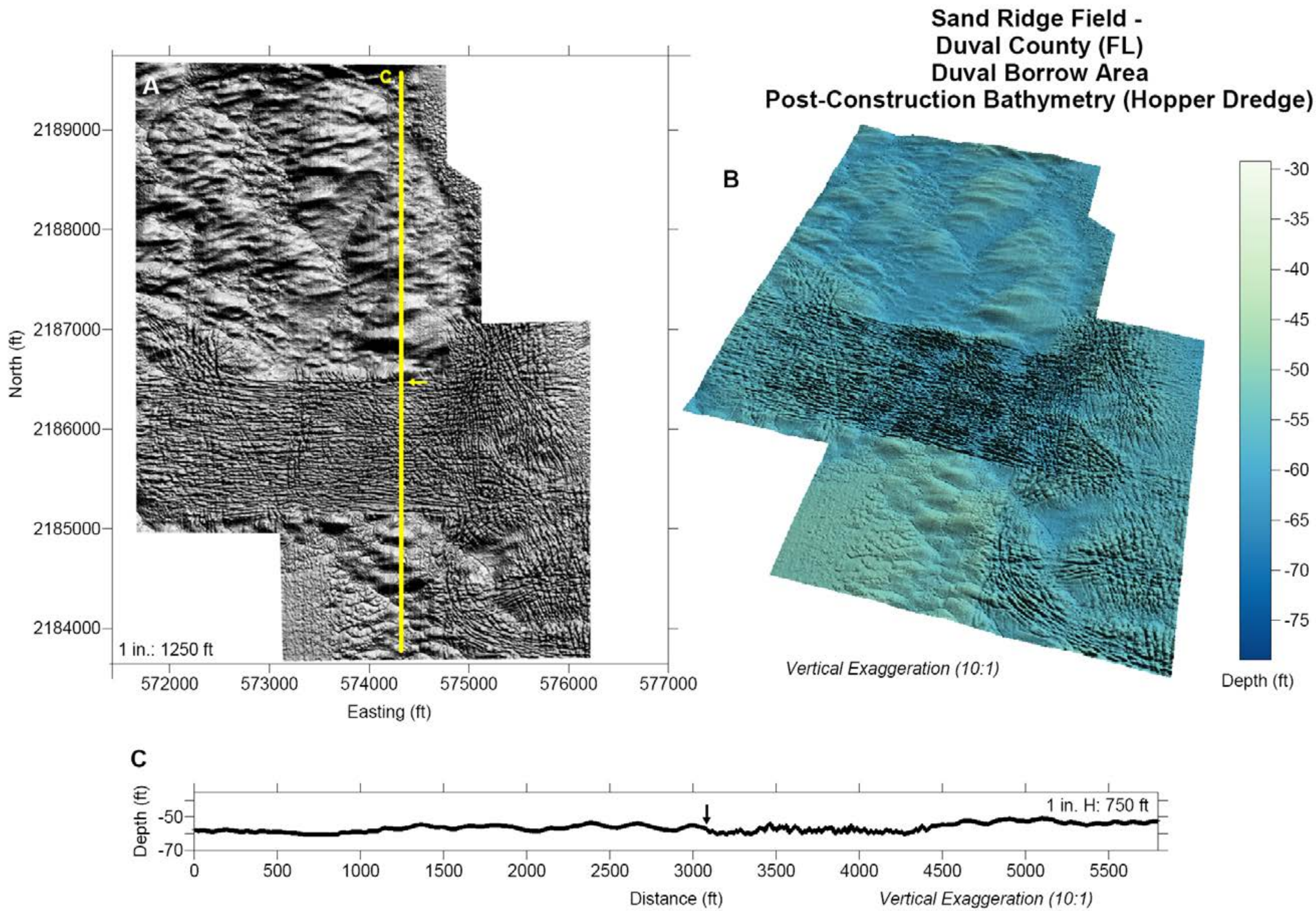


Figure 2.10 Digital elevation models of the Duval borrow area used for the Duval County, Florida project in 2011.

dredging scars; the eastern area was dredged in 2005 and removed 1,500,000 yd^3. With the high-resolution multi-beam sonar data, it is possible to see the irregular bathymetry left after each dredging event. The water depths are around 18 m (-60 ft), thus sediment reworking would occur only during large storms.

The Dade County borrow area consists of a sand sheet located off Miami, Florida where 474,000 yd^3 were removed during a three-week period in 2012. The image in Figure 2.11A shows the adjacent hard bottom in red and an artificial reef in purple. Note the unusual shape of the borrow area to conform to buffers away from hard-bottom habitats, and the irregular bathymetry in the footprint of the borrow area.

The Pelican Island borrow area was a buried paleochannel called Sandy Point located about 13 km offshore Louisiana. Therefore, up to 5 m of muddy overburden were removed and disposed of to the east of the borrow area (readily seen as bathymetric highs in Figure 2.12), followed by removal of up to 10 m of sand, creating an elongated pit with very steep walls and depths up to 12 m deep.

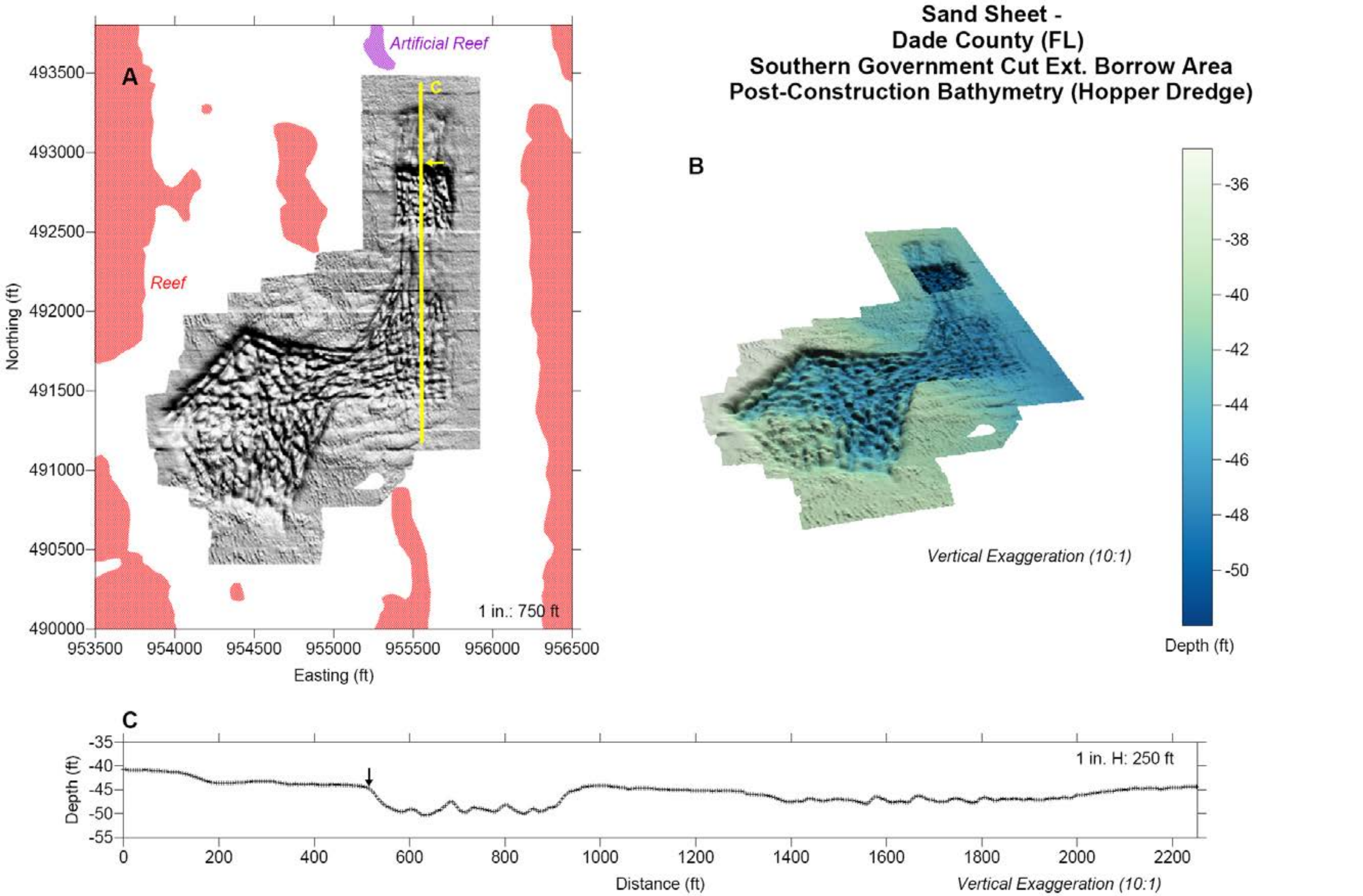


Figure 2.11 Digital elevation models of the Southern Government Cut borrow area used for the Dade County, Florida project in 2012. Hard-bottom habitats are shown in pink in A.

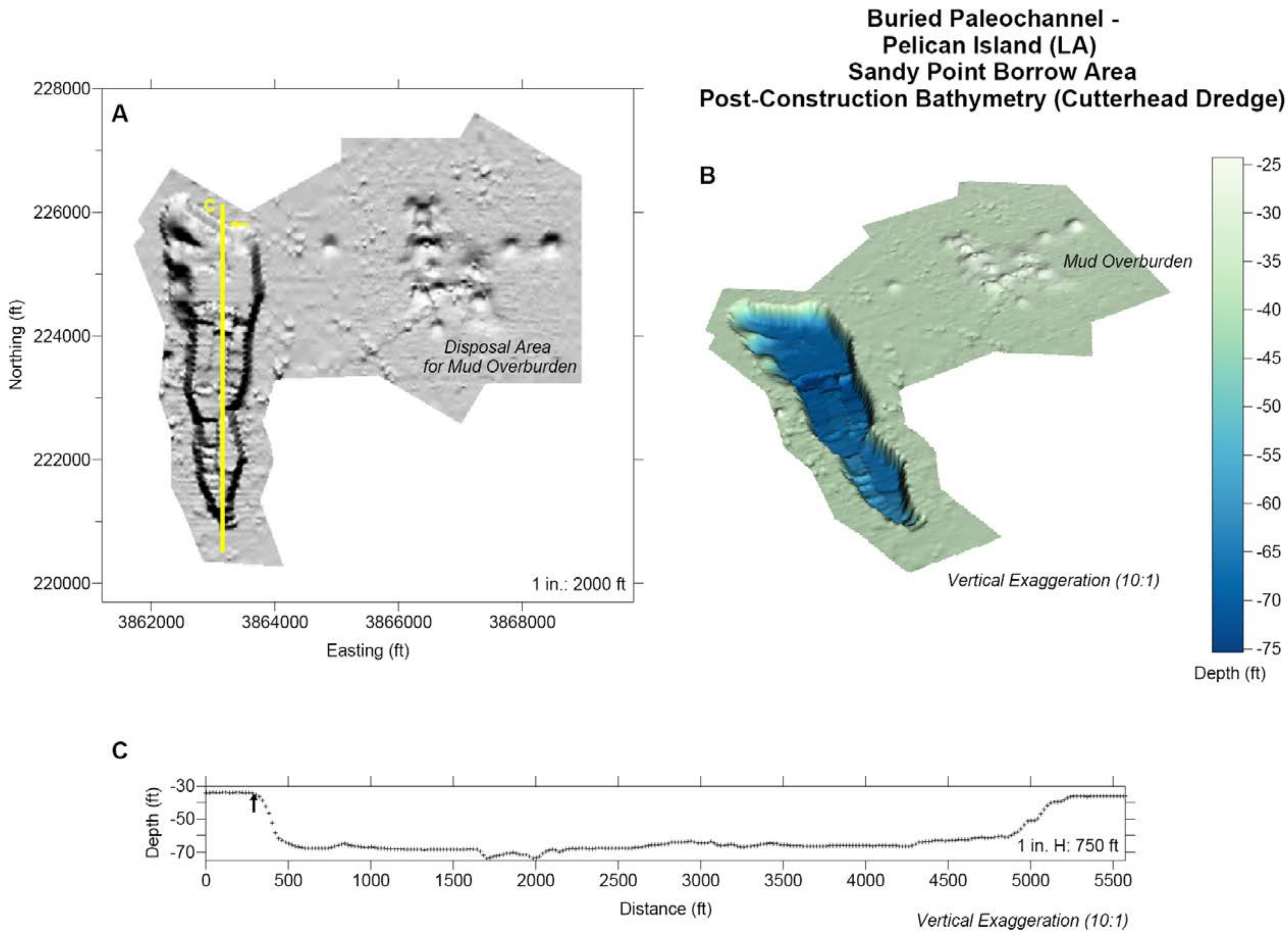


Figure 2.12 Digital elevation models of the Sandy Point borrow area used for the Pelican Island, Louisiana project in 2012.

3.0 OCS DREDGING METHODS AND POTENTIAL EFFECTS

3.1 DREDGING AT THE BORROW AREA

3.1.1 Hopper Dredges and Associated Vessels

Trailing suction hopper dredges (TSHD), as shown in Figure 3.1, are the most common type of equipment for beach restoration and coastal protection projects that use OCS sand because of the water depth, project size, oceanographic conditions, etc. of typical borrow areas. This type of equipment is self-propelled, deploys the suction dredge, and stores the dredged material in hoppers located in the hull of the ship. Typical components of a TSHD include the drag arms, suction pipe, the dragheads (located at the end of the suction pipe), and the dredge pump and the hopper located onboard the ship. TSHDs discharge the material at the placement site by opening doors in the hull to dump the sediment, pumping the sediment via pipeline laid on the seafloor between the dredge and the placement site, or by "rainbowing" where the material is dispersed through the air via an inclined pipe (rainbowing is not used often in the U.S.). The sand can also be dumped in a temporary rehandling area (subject to Ocean Dumping Act permitting) in the nearshore for secondary dredging and transport by pipeline to the placement site.

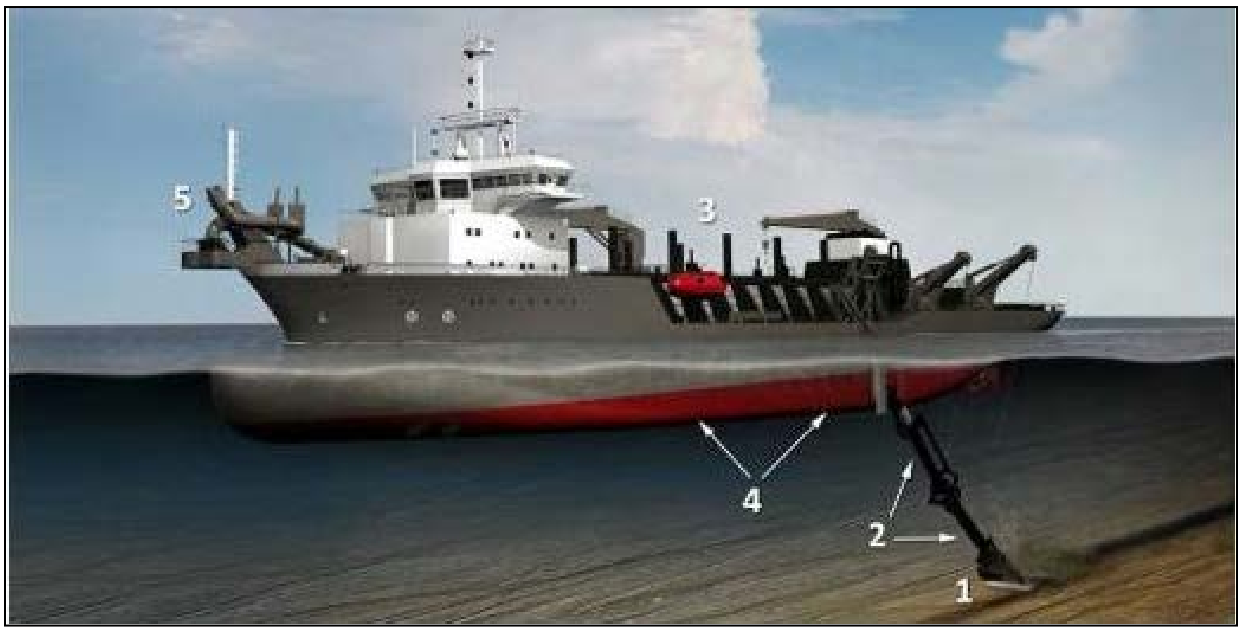


Figure 3.1 Trailing suction hopper dredge components: 1) draghead, on the end of a large 2) suction pipe, through which large centrifugal pumps transport the dredged material as a slurry to the 3) hopper from where it is later discharged either through 4) bottom doors or 5) pumped through a pipeline from the bow (from http://www.marinelog.com/DOCS/NEWSMMIX/2010feb00100.html)

TSHDs used for beach nourishment projects in the U.S. have hopper capacities up to 10,000 m^3, though most are in the range of 3,000-7,000 m^3. During dredging operations, hopper dredges travel at a ground speed of 3 to 5 km per hour (1.5-3 knots) and can dredge in depths of 3-24 m. The draghead, which varies in width from 1.5-4 m, can remove 9-46 centimeters (cm) of material in each pass (CSA et al. 2010). Dredgers are required to maintain the Dredging Quality Management System (formerly call the "Silent Inspector") that automatically records the following information:

- Dredging position
- Dredging depth
- Vessel displacement
- Cargo tonnage
- Tons of dry solids
- Vessel speed
- Vessel heading
- Dredge cycle time
- Slurry flow-rate
- Slurry density
- Vessel status (loading, sailing, dumping, and idle).

Buffer zones are required to be entered into positioning systems to avoid sensitive areas.

Vessels associated with TSHDs include crew boats for ferrying people, small supplies, and groceries to the dredge and geophysical survey vessels.

3.1.2 Cutterhead Suction Dredges (CSD) and Associated Vessels

These dredges use a cutterhead to excavate the material for removal and create a slurry that is pumped into a 76 cm (30 inch) pipeline for transport to the placement or disposal site (Figure 3.2). Spider and hopper barges may be used for long distances or where pipelines would hinder navigation. CSDs operate by moving around a stud pole or stud, using multiple side anchors to allow movement. The cutterhead is swung laterally (back and forth) in an arc. The spuds can be used to walk the dredge in the desired direction. They can also be operated using anchors if the waves make use of a stud pole or spud dangerous. Moving the dredge may require frequent repositioning of anchors. Turbidity is generated at the seabed by the cutter head, or temporarily at loose/leaky connections (that are immediately fixed or production decreases significantly). CSDs can be used for removal of an overlying mud layer and sand from the borrow area, and they often are used to place sand stockpiled at a temporary rehandling area onto the beach.

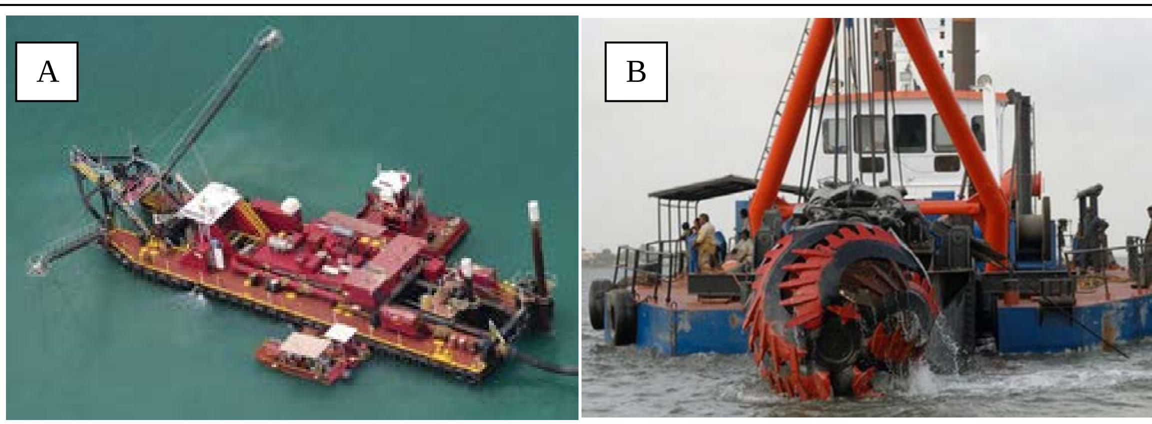


Figure 3.2 (A) Cutterhead suction dredge and (B) cutterhead. From www.dredgepoint.org.

CSD operations include use of support boats, crew boats, and survey boats, while barges are often used as support platforms. During dredging, there may be 15-60% spillage, which is the material that is excavated from the seafloor but not removed by the suction mouth (CSA et al. 2010). This spillage material can form rows or piles that leave the dredged surface very uneven.

3.2 Conveyance to and Handling at the Placement Site

3.2.1 Vessel Transport

TSHDs are often used for OCS dredging because of the long distances between the borrow area and placement site. They can travel at speeds of up to 14 knots when unloaded, and 1-2 knots slower when loaded. There may be speed restrictions in certain areas in certain times of the year; for example, vessels greater than 19.8 m (65 feet) are restricted to 10 knots in North Atlantic right whale special management areas. Barges are seldom used for transport of sand from OCS borrow areas, but could be used in conjunction with CSDs.

3.2.2 Pipeline Transport

The following descriptions were summarized from CSA et al. (2010). When pipelines are used for transport of sand from OCS borrow areas or rehandling sites, they are placed on the seafloor. Floating or flex lines and riser pipelines are used to connect the submerged pipeline (or subline) to the dredge, booster pumps, and at the pump-out connection. The subline is assembled in sections (rafted) using a derrick barge with a crane, connecting each section using collars and ball joints. Multiple tugs (2-3) and tending vessels then mobilize the rafted subline offshore, using float buoys to help float the rafted subline. Final positioning of the subline is accomplished offshore using a derrick barge and crane and, on the beach, using pipeline loaders. Once the subline is in position, any float buoys are disconnected and the subline is flooded into place. The pipeline is marked with surface buoys along its length. Multiple boosters may be installed in-line, depending on the horsepower of the dredge and the distance to shore, using anchors and anchor wires or clump weights. During maintenance and repair operations, a derrick barge mechanically retrieves sections of the subline. During demobilization, a plate is welded onto the discharge end, and the subline is purged. Air is pumped into the subline and sections are floated. The subline is mechanically retrieved in rafted sections using tugs and derrick barges. Rafted sections are towed back to the staging area and disassembled. Figure 3.3 shows various components of a submerged pipeline system being used to transport materials from the Sandy Point borrow area to the placement site on Pelican Island, Louisiana during 2012 operations.

3.2.3 Direct Pump Out and Rehandling in State Waters

There are two options for placing the dredged material: 1) direct pump out at the placement site; and 2) temporary seabed storage and rehandling. A direct pump-out operation by TSHD may incorporate some or all of the following equipment (CSA et al. 2010):

- Anchorage by anchors or other suitable equipment
- Floating pipeline (hose) extending from the bow of the ship
- Booster(s) to assist with the discharge of the material to the shoreline
- Riser pipeline
- Submerged pipeline

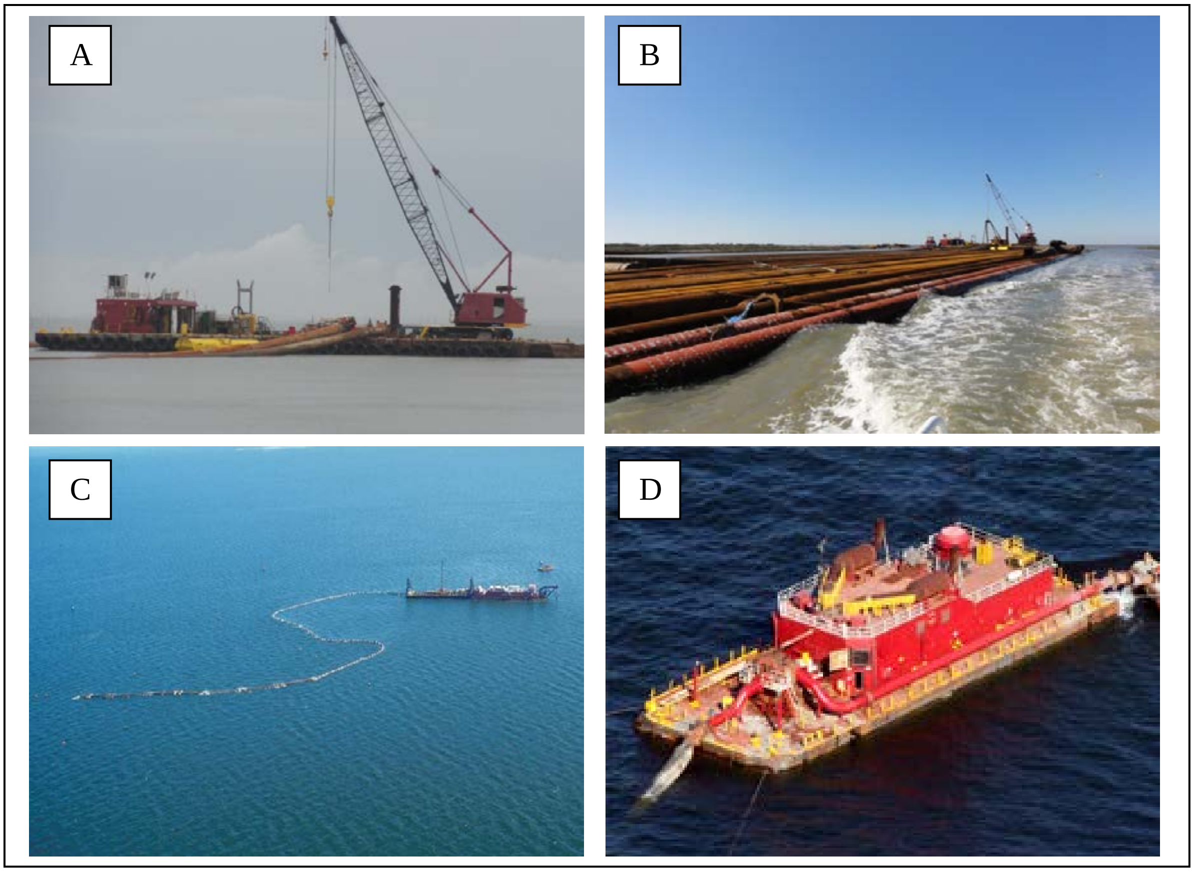


Figure 3.3 Operation of a submerged pipeline (subline) for the Pelican Island restoration project. A) Barge and crane being used to connect sections of pipe. B) Pipes rafted together ready to be connected. C) Floating flex line between the CSD and the subline at the borrow area. D). In-line booster pump connected to the subline by flex lines and secured by anchor lines. Photographs courtesy of Great Lakes Dredge & Dock.

- Shore pipeline, dozers, tractors, lights, and maintenance equipment located on shore
- Tug to assist the connection of the dredge to the pipeline
- Mono-buoy to assist with the connection of the pipeline
- Crane barge for working with floating equipment in the vicinity of the project
- Crew boat(s) for moving people and equipment around the site
- Survey boat(s) to monitor progress and productivity of the dredge

There has been more frequent use of the option for dredgers to use rehandling areas for subsequent placement on the shoreline, particularly when the borrow area is located a long distance from the placement site, which can often be the case for OCS sand borrow areas. Bodge (2002) provides this explanation:

> "The rehandling concept endeavors to make maximum efficiency of the dredging equipment and fleet. Specifically, hopper dredges are efficient at picking up sand and dumping it. Cutterhead dredges are efficient at continuous transfer and placement of sand. Their combined use can potentially improve (speed) production. Further, the demand for hopper versus cutterhead dredges changes throughout the year; and the

> opportunity to employ various pieces of otherwise idle dredge plant allows the contractor to optimize the fleet's schedules."

Because rehandling requires temporary seabed storage of the sand, Ocean Dumping Act permits require that the storage area be located to avoid sensitive areas and could require a buffer of clean sand over the original seabed so that there is no disturbance of the sediments. Rehandling areas are included in the plans for several shoreline restoration projects in Louisiana, including: 1) the Caminada Headlands project, where sand will be dredged from Ship Shoal 39 km from the placement site, includes three potential pump-out areas that range in size from 40 m x 200 m along Belle Pass to an offshore area that is 460 m by 460 m; and 2) the Cameron Parish Restoration Project, where sand will be dredged from Sabine Bank 30 km distant from the placement site, includes options for either a rehandling area ~10 km offshore or a pump-out site within Calcasieu Pass. Two rehandling sites are proposed for the Longboat Key, Florida project.

3.3 POTENTIAL ENVIRONMENTAL EFFECTS BY IMPACTING MECHANISM

In this section, we describe the potential environmental effects from dredging OCS sand, organized by the eight impacting mechanisms listed in Section 1.2. These effects vary in scale by time (resilience, recovery, and permanence) and space (near-field, far-field, and regional), depending on the resource receptor being considered. These varying scales and levels of impacts are discussed by resource in Section 4. Furthermore, there are indirect effects, synergistic feedbacks, and biophysical and bioenergetic couplings that are very important elements to assessing the cumulative effects from OCS dredging; we address the ecological interactions among biological resources in Section 4.7.

3.3.1 Alteration of Benthic Habitat at the Borrow Area

Dredging of any kind results in the direct removal of benthic habitat along with infaunal and epifaunal organisms with limited mobility, resulting in reductions in the number of individuals, number of species, and biomass. Based on studies of benthic community recovery following dredging (Van Dolah et al. 1994a; Blake et al. 1996; Newell et al. 1998; Brooks et al. 2006), communities of comparable total abundance and diversity can be expected to re-colonize dredge areas within several years. More long-term study of macrobenthic recovery has been completed in the North Sea off the U.K. than in the U.S., allowing more confident conclusions on recovery rates for environments in shelf sediments targeted for mining there. For example, Newell and Seiderer (2003) summarized recovery rates of benthic communities post-dredging for different substrate types, showing that sandy substrates typically recover within 2 to 4 years. In contrast, dredging in coarse sandy plains with moderate tidal stress in the U.K. revealed biological recovery times of 8.7 years (Foden et al. 2009). In the U.S. Gulf and Atlantic east coasts, Brooks et al. (2006) review results of all studies of recovery rates after dredging for sands in the OCS and state waters, reporting "general faunal recovery in 3 months to 2.5 years." However, monitoring after dredging in U.S. studies typically ends before recovery of community composition and diversity has been achieved (Brooks et al. 2004, 2006): species composition after sand dredging differed still after 43 weeks (Boesch 1979), 1 year (Johnson and Nelson 1985), 30 months (Jutte et al. 2002), 1 year (Posey and Alphin 2002), 3 years (Saloman 1974), 1 year (Saloman et al. 1982), 5 years (Turbeville and Marsh 1982), and 38 months (Palmer et al. 2008). So while the macrobenthic communities on the OCS generally recover total abundance and biomass within 3 months to 2.5 years, their taxonomic composition and species diversity, reflecting dominant species and species abundances, can remain different from pre-dredging to post-dredging for more than 3-5 years. How long the differences persist cannot be inferred from the paucity of long-term U.S. studies.

Benthic communities can also be affected indirectly by post-dredging changes in sediment characteristics, mostly from infilling of the dredged area by finer-grained sediments provided there are some specific site conditions (proximity to a fluvial/estuarine source of fine-grained material, fine-grained sediment supply in substrate exposed or alongshore/alongshelf sediment supply, a depression created that provides the accommodation space, and the bottom boundary shear stress is not sufficient to suspend/mobilize fine-grained/flocculated sediment.) In a well-cited case, Van Dolah et al. (1998) studied six dredged areas in South Carolina and found that, at three of the sites, the borrow area had filled with muddy sediments that formed a cap over clean sand. This study led to a “no pits” rule for similar environments to minimize the potential for infilling with muddy sediments, which would lead to a change in benthic communities. More recent studies of these and other borrow areas in South Carolina have confirmed these results (Bergquist and Crowe 2009; Bergquist et al. 2011a, b), showing an increase in deposition of finer-grained sediments in the form of silt, clay, and fine sand, along with dominance of the benthic community by polychaetes and disturbance-tolerant or early successional taxa and reductions in taxa that prefer coarse and sandy substrates that persisted for 1-2 years post-dredging. The degree to which the recovery of these borrow areas, in the nearshore zone of South Carolina, represent recovery conditions on the broader OCS is uncertain. However, this collection of studies and their meta-analysis (Bergquist and Crowe 2009) offer insight on the impact of dredged depth on the recovery of borrow areas, with consistent results found at the South Government Cut borrow area near Duval County, FL. Because long-term data on the recovery of borrow sites are limited, we recommend additional studies to examine the impact of borrow area depth on benthic recovery in section 5.1 of this report.

3.3.2 Increased Turbidity and Suspended Sediments in the Water Column

Temporary sediment plumes and increased turbidity in the water column can arise from the mechanical disturbance of the seabed by the draghead or cutterhead during dredging operations (bottom plumes) and overspill of surplus sediment/water mixture from the vessel hopper and rejection of unwanted sediment fractions during the sorting and screening process (surface plumes) (Sutton and Boyd 2009). Although infrequent, rehandling and a leaking subline can result in increased turbidity. The spatial and temporal extent of these plumes depends on several factors including site-specific dredging processes (e.g., dredging equipment, amount of dredging, thickness of the dredged layer) (CSA et al. 2010), sediment characteristics and particle-size distribution, and site-specific hydrodynamic and sediment transport regimes (Hitchcock et al. 1999; Newell et al. 2004a; Sutton and Boyd 2009). Based on studies of dredging of coarse aggregate in the U.K., coarse particles (i.e., sand) typically settle out of the water column within 300-600 m of the source, while fine material (i.e., silt) can be detected as far as 3.5 km, possibly due to the elevated concentration of low-density organic matter from fragmented benthos discharged during sorting (Newell et al. 2004a). Although surface and bottom plumes in the water column are typically short-lived (10-15 minutes following release) and highly localized (Hitchcock et al. 1999), currents and waves can resuspend and redisperse sediment particles, propagating dredge-related turbidity for longer periods post-dredging (CSA et al. 2010) and beyond the footprint of extraction (Newell et al. 2004a).

At OCS borrow areas, the sediments are usually composed of clean sand, so water-column turbidity from overflow discharges has seldom been of concern at the borrow area. There are general statements in the literature that plumes are short-lived and measured typically in thousands of meters (Hammer et al. 1993). Increased turbidity is mostly of concern when dredging adjacent to sensitive benthic habitats such as coral reefs and other hard-bottom habitats. During the 2009-2010 Brevard County South Reach, Florida dredging project, water quality was monitored every six hours during daylight hours at the borrow area and beach fill site at 500 m up-current and no more than 150 m down-current from the activity in the densest portion of the visible plume. Overall, the turbidity averaged about +4.5 nephelometric turbidity units (NTU) above background measurements, and none of the 228 measurements exceeded the maximum allowed of +29 NTU above background (Olsen Associates Inc. 2010). Monitoring of the 2005/2006 Longboat Key, Florida nourishment project showed that turbidity levels never exceeded 29 NTU above background at the dredge nor the fill site (Coastal Planning & Engineering Inc (CPE) 2011b).

Bodge (2002) reported construction-related turbidity levels above background for different phases of dredging operations on a homogeneous coarse sand in Brevard County, Florida as:

Borrow Area–Dredging: 2.7 NTU above background
Rehandling Area–Disposal: 2.2 NTU above background
Rehandling Area–Dredging: 1.9 NTU above background
Beach Fill–Rehandled: 2.9 NTU above background
Beach Fill–Hopper Pump-out: 3.9 NTU above background

There could be water-column turbidity concerns from the use of CSDs to remove and sidecast a fine-grained sediment overburden to access a sand resource, such as buried channels in the Gulf of Mexico. For example, about 1 m of fine sediment was stripped from the OCS dredge site off Holly Beach, LA, and up to 5 m of mud were removed from the Sandy Point borrow area and placed in an adjacent disposal area (Figure 2.12). CSDs generate relatively low amounts of suspended sediments, which are mostly confined to the immediate vicinity of the cutterhead where it is dredging (typically within 3 m above the cutterhead and extending several hundred meters laterally) and dissipate rapidly (CSA et al. 2010).

3.3.3 Increased Sediment Deposition on the Seafloor

Increased sedimentation from suspended sediments would be of concern when this process causes a change in the characteristics (grain size, organic content, redox potential, etc.) of the benthic sediments, smothers sensitive habitats such as coral reefs or hard-bottom communities, or affects spawning habitats. There are general statements in the literature that open, offshore areas are dynamic and so have reduced risk of settlement of fines, and that organisms in these settings are acclimated to natural sedimentation and scour during storms (CSA et al. 2010).

Goldberg (1988) summarized measurements of increased sedimentation in adjacent hard-bottom habitat during dredging of borrow areas in state waters off South Florida, with two sites showing increased siltation and biological effects: 1) Hallandale Beach in 1971, where siltation caused large-scale stony coral mortality at distances of 130-220 m from the borrow area; and 2) Miami Beach from 1977-1982 where layers of fine sediments 1.3-3.3 cm thick covered patch reefs in the vicinity of the borrow areas (the borrow material contained 4-46% silt, averaging 15.2%, which was a key factor). More recently, there are state regulations limiting the mud content of the sand placed on the beach.

Jordan et al. (2010) report on monitoring of sedimentation in reef habitat at ten sites adjacent to five borrow areas between shore-parallel reef tracts in state waters off Broward County, Florida where dredging occurred over an eight-month period in 2005-2006 (which included the passage of several hurricanes that affected the area). They found that sampling stations in close proximity to dredging in the borrow area closest to shore (in state waters) exhibited higher sedimentation rates and lower percentages of fines during construction when compared to control stations. The middle and outer borrow areas (all still in state waters) did not show any significant differences. They suggested that the fines in the turbidity plumes from the hopper overflow were advected away from the dredging areas. Interpretation of their results was complicated by the widespread resuspension of bottom sediments during the passage of several hurricanes during the dredging and study periods.

During the 13 March-4 April 2012 South Government Cut dredging project off Miami, Florida, sedimentation was measured at stations 120-670 m from the borrow area (PERA CS-14 2012). Most stations showed no or minor increases in sediment thickness on the seafloor and coral communities; however the two closest sites (140 m south and 180 m north) showed higher sedimentation (up to increase an ~30 mm above pre-dredging amounts). Most stations returned to pre-dredging conditions within two weeks after dredging, with the exception of the one station to the north where the borrow area wrapped around the site on two sides (shown in Figure 2.11A as the isolated hard bottom just north of the image boundary). When the monitoring indicated high sedimentation and increases in coral stress indicators, the buffer distances were increased.

3.3.4 Entrainment Near the Seafloor

Dredging with TSHDs will entrain all slow-moving fauna on or near the sediment surface, such as crabs, shrimps, and some types of demersal fishes. Reine and Clarke (1998) summarized the literature on entrainment for USACE dredging projects in estuaries and navigational channels, reporting entrainment rates as numbers of individuals per yd^3 for fish and invertebrates (discussed in Section 4.3.2.5) and number of individuals per 100,000 yd^3 for sea turtles. The primary concern for dredging in OCS borrow areas is entrainment of species listed under ESA, with sea turtles of greatest concern (see Section 4.6). Entrainment of sea turtles is more of a concern with USACE dredging of navigational waterways, with 698 sea turtle “takes” over the period 1995-2011 from Texas to Virginia (Dickerson 2011). There have been 19 sea turtle takes (all loggerheads) for 21 projects and eighteen years of dredging operations (1995-2012) during dredging of OCS sand borrow areas in the South Atlantic region (discussed in more detail in Section 4.6 on sea turtles). There have been no sea turtle takes associated with dredging OCS borrow areas in the Gulf of Mexico or mid-Atlantic region. In 2006, a spotted dolphin was captured during trawling at the T1 borrow area for the Collier County, Florida project.

3.3.5 Sound

Processes associated with marine sand extraction contribute to increased sound levels above background and have different sound characteristics that vary depending on operational (i.e., dredge type, ship/pump size) and environmental factors (i.e., seabed type, depth) (Department for Environment Food and Rural Affairs (DEFRA) 2003; Thomsen et al. 2009; Saunders and Roberts 2010; Tillin et al. 2011). Key sources of sound include pump driving, transport, deposition, and draghead movement over the seabed, sound generated by engine noise, propeller cavitation, and wave-hull interaction, and impulsive sound sources such as fathometers (DEFRA 2003).

One of the technical challenges with the literature on sounds generated during dredging operations is the variations in units used to report sound. The following discussion of sound units is extracted from the Discovery of Sound in the Sea website (http://www.dosits.org/). The sound pressure level (SPL) in decibels (dB) is defined as 10 times the logarithm of the ratio of the intensity of a sound wave to a reference intensity as shown in the equation below:

$$dB = 10 \log_{10} (dB_{sound}/dB_{reference})$$

Thus, sound intensity in dB is a relative, not an absolute unit and, to compare sound levels, a standard reference pressure must always be used. The reference sound intensity for underwater sound is the intensity of a sound wave with a pressure of 1 microPascal (µPa). In contrast, the reference intensity for sound in air is 20 µPa, the threshold of human hearing at 1,000 Hz.

Sound travels as a wave, and the wave amplitude is related to the amount of acoustic energy it carries. There are different ways to describe the amplitude of a sound wave, as shown in Figure 3.4: peak pressure, peak-to-peak pressure, and root-mean-squared (rms) pressure. It is often not clear how the sound intensity reported in the literature was calculated. Studies report sound as *source level* (in dB at 1 m from the sound source), *received level* (in dB at a specified distance from the sound source), or *sound exposure level* (the dB level of the cumulative sum-of-square pressures over the duration of a sound for sustained non-pulse sounds where the exposure is of a constant nature, reported as dB re: 1 µPa^2-s; Southall et al. 2007).

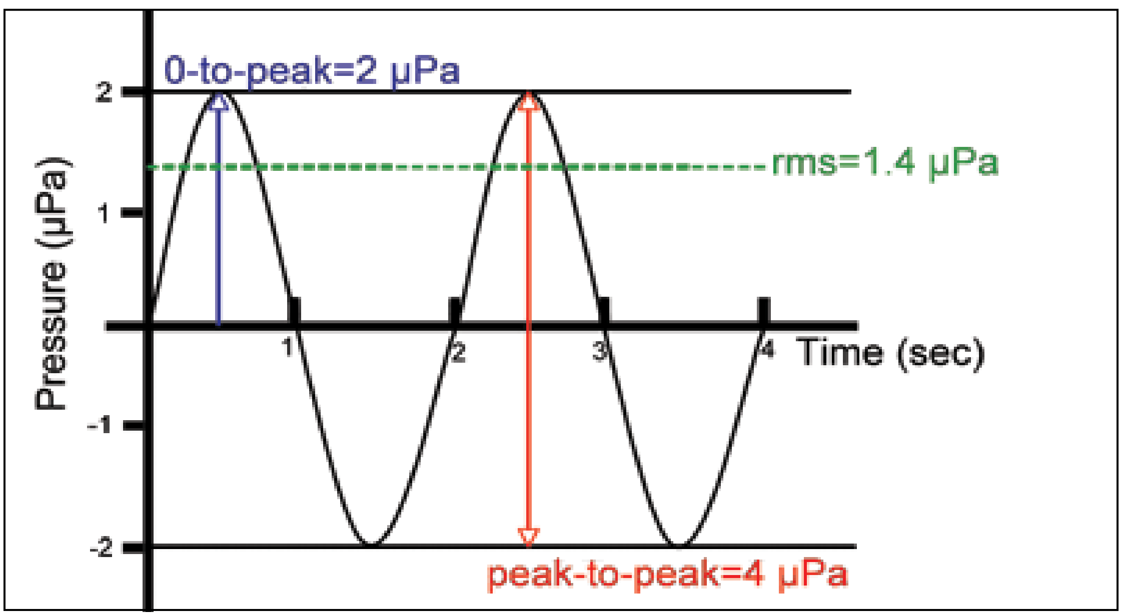


Figure 3.4 A simple sound wave, showing the different ways to describe the amplitude of a sound wave and calculate the wave amplitude. From **http://www.dosits.org/**.

For broadband sound that includes many frequencies, such as the sounds generated during dredging operations, the sound intensity can be reported in octave bands that quantify the effective frequencies without looking at each frequency one at a time; it is characterized by the center frequency of the band edges. The band width is a constant fraction of the band's center frequency (about 23% for third-octave bands); thus, at the center frequency of 1,000 Hz, the bandwidth is 230 Hz (Dahl et al. 2007).

Greene (1987) measured hopper dredge sound during gravel mining operations in the Beaufort Sea and reported sound levels of 142 dB re 1 µPa at 0.93 km for loading operations at a depth of 20 m, 127 dB re 1 µPa at 0.93 km while underway, and 117 dB re 1 µPa at 13.4 km while pumping at a depth of 13 m. Richardson et al. (1990) made recordings of dredge operations in the Canadian Beaufort Sea and played them underwater to observe reactions of bowhead whales, as well as monitored whale behavior at various distances from dredging operations. Clarke et al. (2002) monitored sounds from maintenance dredging in estuaries including a bucket dredge (Cook Inlet, Alaska), a hydraulic CSD (Mississippi Sound), and a TSHD (upper Mobile Bay, Alabama). They measured sound during the material removal operations, observing that TSHDs produced sounds that were comparable to those made by vessels of similar size, but more intense than CSDs. Hopper dredge sounds peaked in the 120-140 dB range, while CSD sounds peaked in the 100-110 dB range with sounds almost inaudible at ~500 m from the source.

de Jong et al. (2010) reported on measurements of underwater sound of seven TSHDs during a range of sand dredging operations in the Netherlands, with dredges ranging in size from 4,500-20,000 m^3 (5,900-26,200 yd^3). Figure 3.5 shows plots of the source levels for the seven dredges and the different types of operations. Their summary stated:

> "Sand dredging generally produced source levels at a few decibels lower than for transiting dredgers. Pumping and rainbowing resulted in source levels similar to dredging in the frequency range between 500 Hz and l0 kHz and significantly lower levels outside this range. The broadband noise characteristics above 100 Hz are very similar for all dredger activities except sand dumping. It is likely that the noise is dominated by cavitation noise from propellers and bow thrusters."

Robinson et al. (2011) measured the sounds generated by five TSHDs during marine aggregate extraction. They collected 140 hours of measurements during six dredging operations (five vessels) using hydrophones set at 50, 120, and 400 m from the dredging operations. All dredgers were monitoring during dredging and screening, except for the *City of Chichester*.

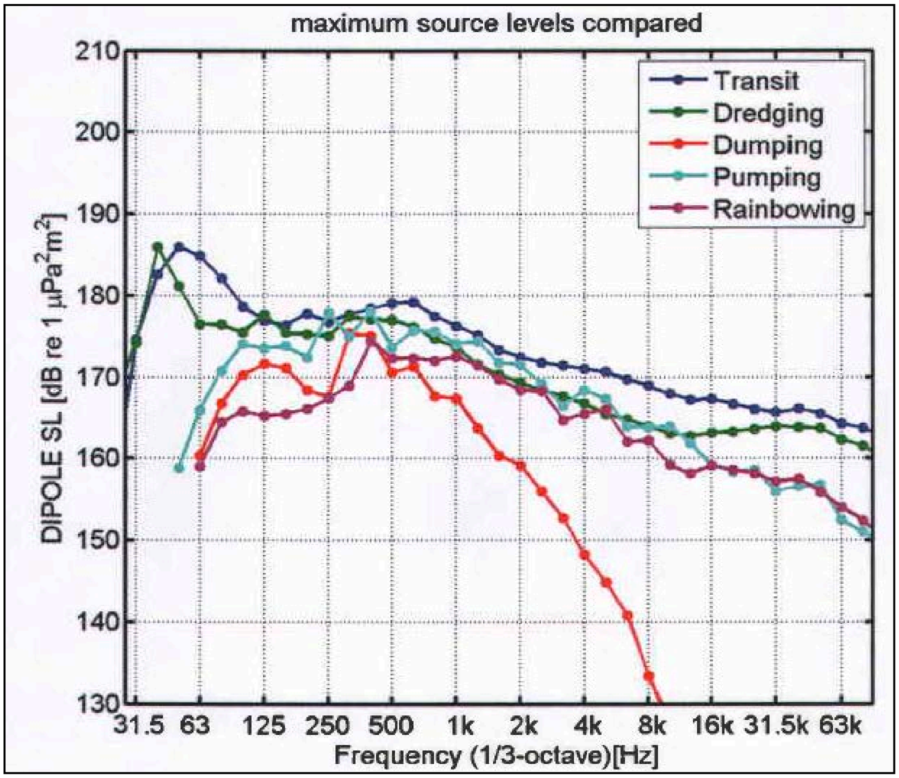


Figure 3.5 The upper envelope of the power-averaged dipole source level spectra of seven TSHDs dredging sand for the various activities monitored in the Netherlands. The dipole source level better represents the sounds from ships (versus a point source) and factors in reflections at the water surface in sound propagation models. From de Jong et al. 2010).

Figure 3.6 shows the data from a dredge working in two different areas, dredging gravel at 40 m in one area and sand at 30 m in a different area, indicating that dredging of gravel is noisier than sand at higher frequencies. Sound levels were in the 160-180 dB range at a frequency of 200 Hz during full dredging operations (draghead down, pump on, extracting aggregate), while the ambient sound at a frequency of 200 Hz was in the 90-100 dB range. Dredge sound levels were also considerably lower when measurements were taken with the draghead raised and the pump turned off, but higher than those reported for merchant vessels travelling at a slow speed. When compared on the basis of cumulative operation duration, small differences (<30 dB) were noted between the sound levels generated by dredging operation (6-8 hours compared) and those of marine pile driving operations (2 hours). This analysis (Robinson et al. 2011) showed that high-frequency, broadband sound correlates strongly with dredging activities, with sound contributions from aggregate material travelling up the pipe and through the pump.

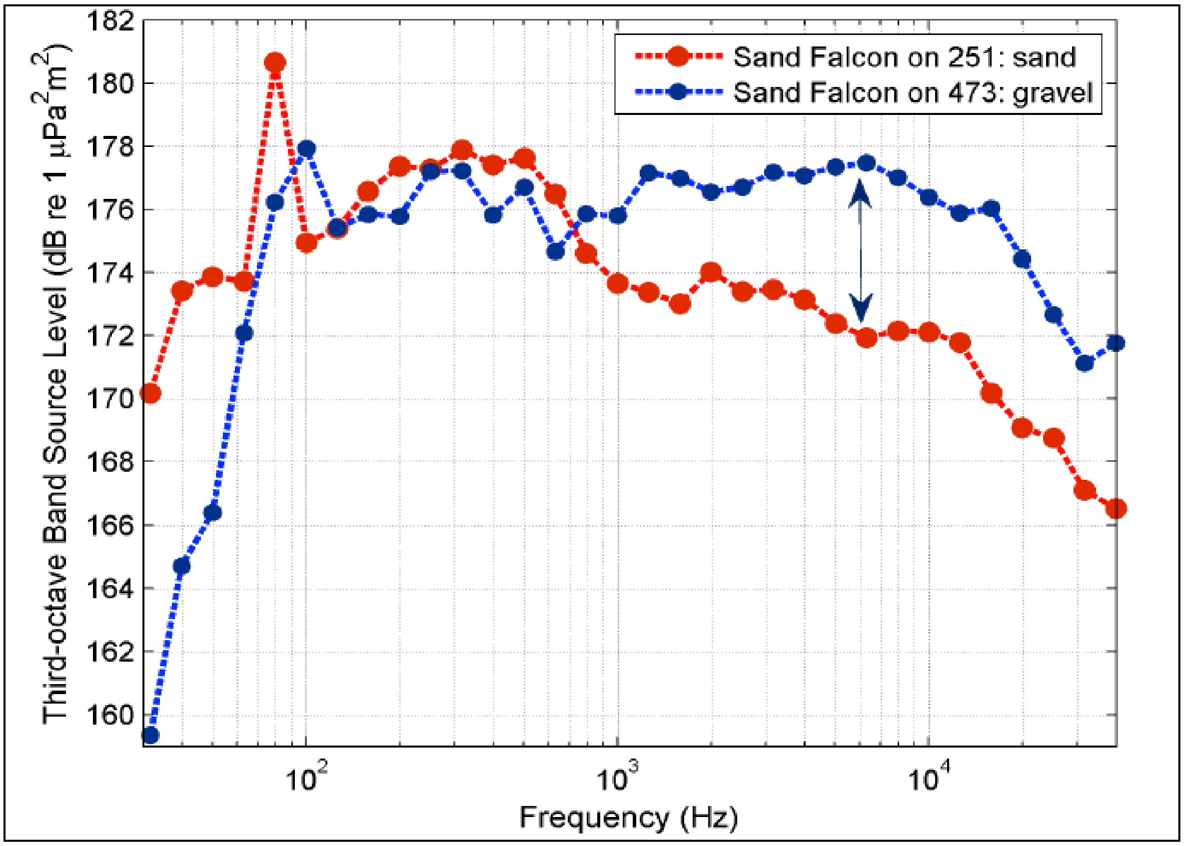


Figure 3.6 The source level for the *Sand Falcon* while loading sand (red) and gravel (blue). Pumping gravel generates more high-frequency sound, compared to sand. From Robinson et al. (2011).

The USACE conducted a study, in part funded by BOEM, of the sounds generated by TSHD operations during the 2012 Wallops Island, Virginia project, which placed 2.75 million m^3 (3.2 million yd^3) on the beach from an offshore shoal about 22 km from the placement beach in water depths of 10-12 m. Table 3.1 is a summary of the results for the different phases of the dredging operations. The dredge *Liberty Island* (total installed power of 16,566 horsepower [hp]) tended to generate higher source levels than the *Dodge Island* (propulsion power of 4,350 hp; total installed power of 9,350 hp) and the *Padre Island* (propulsion power of 3,000 hp; total installed power of 9,395 hp). The highest source level was generated by *Liberty Island* during the transition from the cessation of dredging to the start of the transit from the borrow area, at 178.7 dB re 1 µPa at 1 m; the dredge *Padre Island* generated 170.1 dB re 1 µPa at 1 m for the same activity. All three dredges generated about the same source levels during sediment removal (173.0-174.5 dB re 1 µPa at 1 m). Source levels during transits ranged from a high of 178.2 dB for the *Liberty Island*, to 173.8 dB for the dredge *Dodge*, to 169.5 dB re 1 µPa at 1 m for the *Padre Island*. Attenuation distances ranged from a high of 2.9 km for the stopping and pump out event, to a low of 0.45 km for the period from completion of pump out and heading back out event, both which occurred close to shore. Peak frequencies were in the low range, generally less than 1,000 Hz, though there were some higher peaks up to 3,000 Hz.

These data represent the very first field measurements of dredging operations in the OCS, for three different dredges. Reine et al. (In prep) logarithmically averaged the SPLs and combined the results for all three dredges by dredging phase to assess which dredging activity emitted the loudest underwater sounds.

Table 3.1
Rank order of source levels by dredging activity during the Wallops Island, Virginia project. From Reine et al. (Reine et al. In prep).

Dredge	Dredging Event	SPL Distance[1] (m)	SL[2]	Attenuation Distance[3] (km)	Peak Frequencies (Hz)
Liberty Island	Transition--Digging to Transit	150	178.7	2.5	100-500, 1700, 3000
Liberty Island	Transit to Borrow Area	450	178.2	2.45	150, 1000
Liberty Island	Transit to Pump Out	350	176.2	2.65	100, 400, 1100
Liberty Island	Pump-out Material	150	176.0	No by 1.2	200, 500, 1200
Liberty Island	Pump-out Water	450	175.1	No by 1.2	200, 500
Liberty Island	Transition--Transit to Pump-out	150	175.3	1.1	150, 500, 1500
Dodge Island	Sediment Removal	150	174.5	1.55	100, 700-800
Dodge Island	Transition--Transit to Digging	150	174.3	1.85	100-500, 1700
Liberty Island	Sediment Removal	50	174.2	1.65	200, 500, 1100
Dodge Island	Transit to Borrow Area	150	173.8	2.75	200, 700
Padre Island	Sediment Removal	150	173.0	2.05	150, 250, 1200
Padre Island	Pump-out Material	150	172.0	ULD-1.35; LLD-No	150, 600, 1100
Liberty Island	Transition—Pump-out to Transit	450	171.0	1.1	200-500, 1000, 1600
Dodge Island	Transit to Pump-out	350	170.9	2.9	200, 800
Padre Island	Transition--Digging to Transit	150	170.1	1.1	100, 300, 1000
Padre Island	Transit to Pump-out	350	169.5	1.75	80, 300, 400
Padre Island	Transit to Borrow Area	450	169.5	2.1	100, 300-400
Dodge Island	Pump-out Material	150	166.8	1.35	100-200, 1000
Padre Island	Pump-out Water	450	166.2	0.75	200, 500
Padre Island	Transition-Pump-out to Transit	150	166.0	ULD-0.45; LLD-1	180, 400, 1000
Padre Island	Transition--Transit to Digging	1250	163.2	0.95	100, 500-600
Padre Island	Transition--Transit to Pump-out	150	163.3	No by 1.65	200, 600, 2500
Dodge Island	Pump-out Water	150	162.3	0.85	150, 1000
Dodge	Transition--Pump-out to Transit	450	161.3	ULD-0.95; LLD-2	500

[1]Note that source levels were back calculated using SPL (dB re 1 µPa-1 m) obtained from the listed distance for each event.
[2]SL referenced to dB re 1µPa-1m based on 15.778LogR obtained from fitted regression equation.
[3]Single number indicates that attenuation occurred by this distance at both the upper and lower listening station. ULD = Upper listening depth; LLD = lower listening depth.

Figure 3.7 shows the results (log average of both listening depths). At 50 m from the dredge, sediment removal (digging) produced the highest SPL at 144.9 dB. At 150 m, the transition from digging to transit (turning the fully loaded dredge shoreward toward the pump-out station while increasing to maximum transit speed) produced the highest SPL at 141.8 dB.

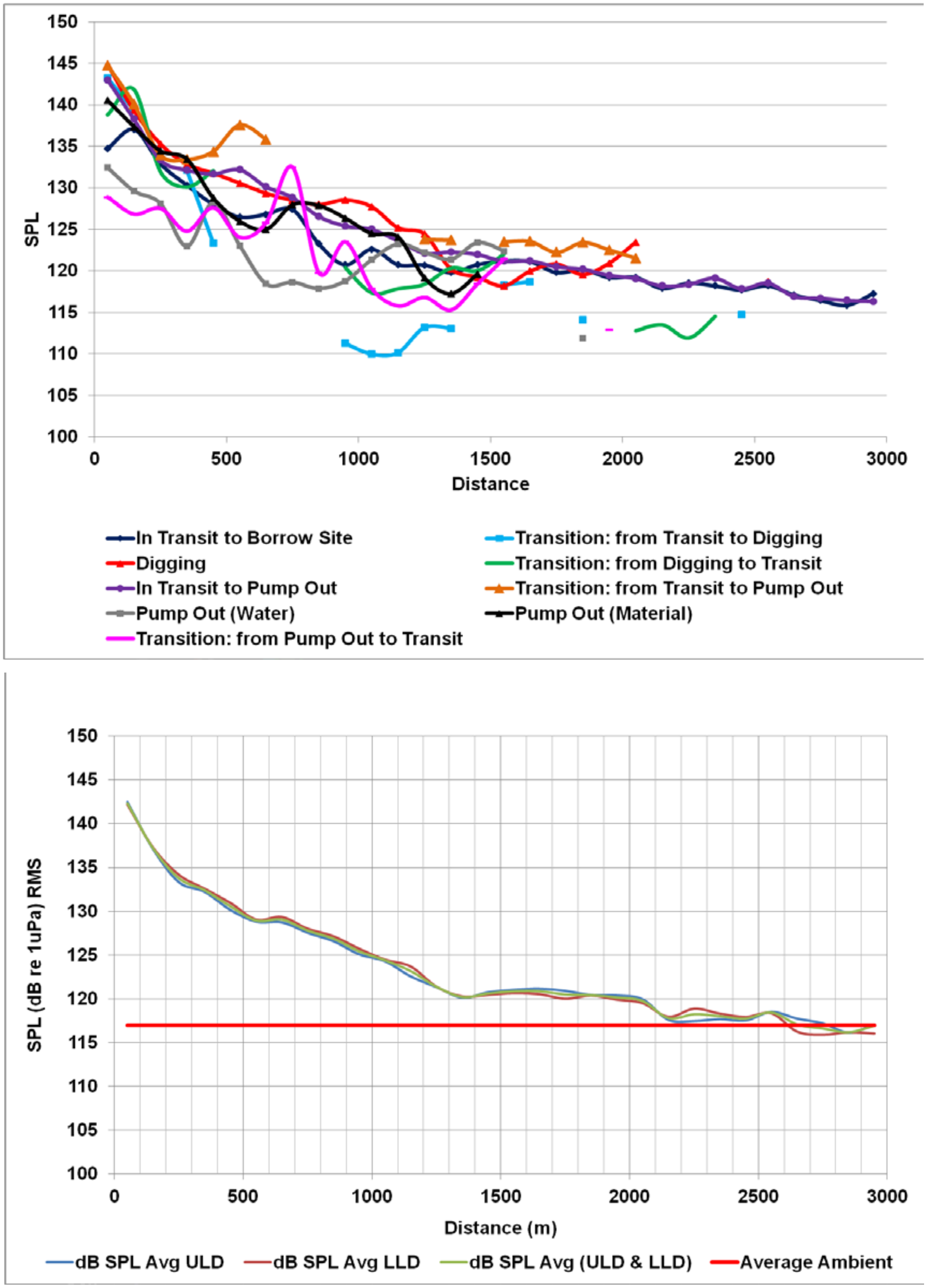


Figure 3.7 **A.** SPL (dB re 1 µPa rms) versus distance by dredging activity. SPLs logarithmically averaged by activity for all three dredges. **B.** SPL (dB re 1 µPa rms) versus distance for all dredges

and dredging events combined. At 2.5 km from the source, underwater sounds generated by all three dredges for all events combined had attenuated to background levels. From Reine et al. (In prep).

At 2.5 km from the source, underwater sounds generated by all three dredges for all events combined had attenuated to background levels. The relatively high background SPLs at this site (116-118 dB for the borrow area, inshore areas, and pump-out sites) is interesting. Nearshore areas, with wind, currents, breaking waves, shallow water, and other sources of "background" sounds such as shipping and industrial activities, appear to be noisier than in the open ocean, where ambient levels are usually less than 90 dB.

Based on these studies, dredging is not as noisy at the source as seismic surveys, pile driving, and sonar, but it is louder than most merchant shipping operating offshore, wind turbines, and drilling (Thomsen et al. 2009). Overall, during sand extraction source levels no higher than 180 dB re 1 μPa at 1 m can be anticipated, with the majority of the energy occurring continuously in the low-frequency region (i.e., <1,000 Hz) (Oslo/Paris Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) 2009; Thomsen et al. 2009; Reine et al. In prep). Another consideration is that TSHD operations differ from passing ships in that they work in a specific area, spending several hours at the borrow area, then transiting to the placement site. Depending on the distance between the borrow and placement areas, there could be several round trips per day.

3.3.6 Vessel Operations and Interactions

Dredges and associated vessels pose risks of strikes of slow-moving animals such as marine mammals and sea turtles. TSHDs do not require constant support vessels at the borrow area, crew and survey vessels are present at regular intervals. However, vessels are used at the pump-out, placement, or stockpiling site, assisting with connections to pipelines, surveying, etc. In contrast, CSDs require multiple support vessels for all operations. There have not been any reports of vessel strikes of marine mammals or sea turtles during OCS dredging operations.

3.3.7 Water Quality

The main concerns related to water quality during dredging operations (other than increased suspended solid concentration, discussed in Section 3.3.2) include the potential for low levels of dissolved oxygen in the water column and at the water-sediment interface, increased nutrients, and accidental oil spills. The creation of pits that refill slowly has the potential to cause a decrease in water mixing and result in lower oxygen exchanges and increased deposition of finer sediments, which may contribute to a localized depletion of oxygen. This result is of particular concern in areas that have seasonal water stratification, such as in the area of seasonal hypoxia in the Gulf of Mexico to the west of the Mississippi River.

In a study of the pit off Holly Beach, Louisiana 38 months after excavation, bottom dissolved oxygen concentrations both in and out of the 3-m deep dredge pit were measured to be 3.0-3.5 milligrams per liter (mg/L). The authors concluded that, although the sediments were finer grained, water quality was the same inside and outside the pit (Palmer et al. 2008). They did qualify this statement by saying that more temporal data would be needed to determine if there were periods of low dissolved oxygen; they only measured water quality on two sequential dates in June.

Release or resuspension of sediments during dredging operations could also affect water quality through the release of nutrients, as documented by Newell et al. (1999). They reported an increase in organic matter in the heavily screened overflow plume during aggregate (mostly gravels with lesser amounts of sand and mud) dredging off the U.K. due to pulverization of biota removed during dredging. No data were found on actual measurements of dissolved nutrients during OCS dredging operations.

All vessel operations pose some risk of accidental spills. Diesel is the most commonly used fuel, especially on the large CSDs, with some of the TSHDs utilizing heavier fuels, such as Intermediate Fuel Oil 180 and marine gas oil. Great Lakes Dredge & Dock indicated that all of their dredges use only diesel as fuel (W. Hanson, pers. comm., 2012). Fuel capacities vary with the vessel and can range from a low of around 380,000 liters (100,000 gallons) to a high of as much as 2.27 million liters (600,000 gallons) or more on a single vessel, with additional fleet capacity in the ancillary vessels surrounding the dredging operation. Fuel is stored in multiple tanks and moved among tanks as needed for power and ballast. Actual capacity carried on board will vary with the project and site conditions, as the more fuel on board means less cargo carrying capacity for the hopper dredges (A. Taylor, C.F. Bean, pers. comm., 2012).

Spills of diesel have the following general behaviors, fates, and effects in open waters:

- Diesel has a very low viscosity and is readily dispersed into the water column when winds reach 5-7 knots or with breaking waves.
- Diesel is much lighter than water (specific gravity is between 0.83 and 0.88), compared to 1.03 for seawater). It is not possible for diesel to sink and accumulate on the seafloor as pooled or free oil unless adsorption occurs with sediment.
- It is possible for the diesel oil that is dispersed by wave action to form droplets that are small enough be kept in suspension and moved by the currents.
- Diesel dispersed in the water column can adhere to fine-grained suspended sediments (adsorption) which then settle out and deposit on the seafloor. This process is more likely to occur near river mouths where fine-grained sediments are carried in by rivers. It is less likely to occur in open marine settings. This process is not likely to result in measurable sediment contamination for small spills.
- Diesel is not very sticky or viscous, compared to black oils. When small spills do strand on the shoreline, the oil tends to penetrate porous sediments quickly, but also tends to be washed off quickly by waves and tidal flushing. Thus, shoreline cleanup is usually not needed.
- Diesel is readily and completely degraded by naturally occurring microbes, under time frames of one to two months in open-water settings.
- In terms of toxicity to water-column organisms, diesel is considered to be one of the most acutely toxic oil types. Fish, invertebrates, and plants that come in direct contact with a diesel spill may be killed. However, small spills in open water are so rapidly diluted that fish kills have never been reported. Fish kills have been reported for small spills in confined, shallow water.
- Crabs and bivalves can be tainted from small diesel spills in shallow, nearshore areas. These organisms bioaccumulate the oil, but will also depurate the oil, usually over a period of several weeks after exposure.

- Small diesel spills can affect marine birds by direct contact, though the number of birds affected is usually small because of the short time the oil is on the water surface. However, small spills could result in serious impacts to birds under the “wrong” conditions, such as a vessel grounding right next to a large nesting colony or transport of sheens into a high bird concentration area. Mortality is caused by ingestion during preening as well as to hypothermia from matted feathers.

No oil spills have been reported during OCS dredging projects. The NOAA Office of Response and Restoration maintains a password-access-only website on which they post real-time information for all spills that they respond to (starting in 1978). This website was searched for any spill involving a dredge. Seven incidents were found where oil from a dredge was released. For two of these, the dredge sank while being towed by a tug in the Great Lakes. The other incidents included: a dredge under tow sank after the tug ran aground off New York; a stationary dredge was involved in a collision when a ship lost control in Newark Bay, New Jersey in 2008; a dredge sank in Charleston Harbor, South Carolina; and fuel was released from a dredge in New York harbor during a fuel transfer. The only offshore spill identified was in September 1999, when the dredge M/V *Stuyvesant* spilled 8,000 liters (2,100 gallons) of Intermediate Fuel Oil 180 near the mouth of Humboldt Bay, near Eureka, California after a dredge arm punctured one of its fuel tanks (California Department of Fish and Game (CDF&G) et al. 2007). It is clear that the risk of oil spills from offshore dredging operations is low.

3.3.8 UXO, Shipwrecks, and Other Hard Structures Temporarily Exposed During Dredging

Dredging of sediments potentially containing munitions and explosives of concern (MEC), unexploded ordnance (UXO), or discarded military munitions (DMM) became of concern in the late 1980s when these types of materials were placed on the shoreline during a beach nourishment project in Virginia, which required a very expensive effort to subsequently remove them from the beach. At-sea disposal of DMM was legal for many years, and such disposal sites are marked on nautical charts. However, there can be a risk of munitions in any offshore borrow area, with increased risks likely offshore of coastal military bases. Munitions have been encountered during dredging operations offshore of Surf City, New Jersey, where 1,078 DMMs were removed from 2.5 km of beach down to 75 cm immediately after the problem was identified, with additional removals over time for a total of 3,107 items (Brewer 2011). Upfront historical record searching is one of the most cost-effective means to prevent these kinds of problems. USACE requires Phase I environmental site assessments during the feasibility phases of project planning.

Because prevention is much more cost-effective than post-dredging removal action, in areas where munitions have been detected or suspected, dredgers may be required to place a screen over the draghead to prevent any UXO from entering dredge equipment and or being placed on the beach. The screen must be designed to prevent the passage of objects greater than 4 cm in diameter. There may be requirements to place screens at the outfall pipe and to conduct periodic inspections of screens and the sand at the placement site. As discussed in the section on sea turtles, screening at the draghead negates the effectiveness of screening prior to placement of sand into the hopper of TSHDs (inflow screening) for diagnosis and observation sea turtle takes, along with other species of concern.

OCS borrow areas along the Atlantic and Gulf of Mexico coast lie among the most heavily utilized shipping routes in the Western Hemisphere. Historical research associated with submerged cultural resource baseline studies has identified those navigation routes as high-density areas for shipwrecks (Research Planning Inc. (RPI) et al. 2004). Exposure of or damage to shipwrecks and other hard structures during dredging are mostly avoided through the requirement to conduct geophysical surveys prior to dredging so such items can be identified and protected from disturbance by placing buffers around them. However, there is always the possibility of damage resulting from power loss, influence of storms, or human error during dredging operations. Furthermore, there may be other mechanisms of damage. RPI et al. (2004) conducted a review of reports of cultural resources damaged by dredging operations and the reasons the damage occurred. They reported twenty documented cases where dredging damaged shipwreck remains, though nearly all cases were associated with developing and maintaining navigation channels. The types of offshore dredging activities that could affect shipwrecks include dragging of wires and chains associated with ground tackle across exposed shipwrecks outside the dredged area during vessel maneuvering or towing of barges from staging areas to the borrow area. For example, as summarized in RPI et al. (2004):

> Another example can be found in the *Hilton* Wreck, a small mid-nineteenth century schooner lost in the Northeast Cape Fear River at Wilmington, North Carolina. The *Hilton* Wreck was found virtually intact in 1988 during a remote-sensing survey for the Wilmington District Army Corps of Engineers... During the spring of 1989, an ocean-going tug towing a barge with a wire bridle dragging from the stern passed the site... The bridle fouled a telephone cable and pulled it across the stern of the *Hilton* Wreck, destroying most of the transom, part of the cockpit, and virtually all of the deck aft of the cockpit...

Dredging could expose shipwrecks and other hard structures that were not identified during the geophysical surveys, or there could be larger-than-expected slumping or changes to the shape of the dredged pit caused by the action of waves and currents and associated sediment transport processes. Nairn et al. (2005) conducted a study to address the issue of possible seafloor instability created by dredging deep borrow pits on the Louisiana OCS and the potential impact on pipelines and other oil and gas infrastructure. The goal was to develop guidance for site-specific buffer zones around an item during dredging. This work was updated by Nairn et al. (2007), who conducted a broader study of buried channels, finding different patterns of pit evolution in sandy versus muddy sediments. In muddy areas, pit margin erosion can extend for some distance beyond the edge of the pit (25 to 50 % of the width of the pit) and result in vertical erosion from several centimeters to 0.6 m (2 feet) or more.

Another concern is the formation of pedestals when a buffer zone extends completely around a surficial or buried feature. A "pedestal" is defined as the situation where the top of the required minimum depth of cover over a feature protrudes above the adjacent bed level after dredging. Over time, the sediment cover extending over the feature will erode and it could become exposed, with the potential for degradation. The foundation of the feature may become unstable, and the feature could lose its structural integrity. Such processes are expected to take a long time assuming that proper buffers were followed; however, for archaeological resources, long-term integrity is a requirement for their preservation.

4.0 FINDINGS BY RESOURCE TYPE

4.1 INTRODUCTION

In this section, we discuss the findings of our literature synthesis on the impacts of OCS dredging operations by impacting mechanism and effectiveness of mitigation methods for each resource type:

- Benthic resources, communities, and habitats (including ESA-listed coral species)
- Fishes and essential fish habitat
- Foraging seabirds
- Marine mammals
- Sea turtles

We also discuss the impacts of OCS dredging on the ecological interactions among biological resources.

4.2 BENTHIC RESOURCES, COMMUNITIES, AND HABITATS

Benthic invertebrate communities of largely sessile and discretely mobile species of several phyla (especially infaunal polychaetes, arthropods, mollusks, and echinoderms in sediments versus sponges, cnidarians, bryozoans, mollusks, hydroids, and polychaetes on hard bottom) are present on the seafloor across the entire OCS. Benthic communities vary in composition with the physical nature of the seafloor (hard versus sedimentary and, within sediments, by the granulometry of sand versus mud), water depth, and history of disturbance. Benthic seafloor habitats are dichotomously either sedimentary or hard bottom in nature. Among sedimentary habitats, sandy sediments on the Atlantic OCS harbor some commercially fished mollusks, such as surf clams (*Spisula solidissima*) and ocean quahogs (*Arctica islandica*) from Cape Cod to Cape Hatteras, sea scallops (*Placopecten magellanicus*) in the Mid-Atlantic Bight, Georges Bank, and the Gulf of Maine, and calico scallops (*Argopecten gibbus*) mostly off the Florida east coast, occasionally the Florida west coast, and rarely North Carolina. Although not fully investigated, the value of sedimentary habitat on the OCS should be judged mostly by its functional role in providing benthic prey to feed demersal fishes, crabs, and shrimps, which are groups of mobile predators of high importance because they include species that are harvested by commercial and/or recreational fishermen and because they are in turn prey for higher-order consumers such as seabirds, larger fishes, sea turtles, and marine mammals (Hill et al. 2011). This function of feeding predators of value is one type of promoting bioenergetic transfers up the food web. Sedimentary habitat such as sand shoals on the OCS can also function as breeding and spawning sites (Gelpi et al. 2009; Condrey and Gelpi 2010) and as a provider of structural habitat complexity that protects vulnerable juvenile life stages of fishes and crustaceans (Diaz et al. 2003). This structural complexity is provided physically by large bedforms on meso- and microscales (with wavelengths >30 cm and crests about 10 cm) in sand, and in most sedimentary habitats also indirectly by biogenic growth of erect, emergent invertebrates, such as sponges, bryozoans, hydroids, and other clonal animals, and by burrows into sediments or by reefs of tube-building polychaetes (Diaz et al. 2003).

Hard-bottom habitat on the OCS is generally valued more highly than sedimentary bottom for its role in providing emergent structural refuges for early life stages of fishes and crustaceans (e.g., Lindeman and Snyder 1999, although this study was done in State waters; CSA 2009) and in harboring food sources for many invertebrates, sea turtles, piscivorous and molluscivorous diving birds, and marine mammals. These contributions to higher trophic levels have led to hard-bottom habitat on the continental shelf of the U.S. being classified and managed by NOAA as EFH under the Magnuson-Stevens Fisheries Conservation and Management Act. Hard-bottom habitat on the OCS of the Florida Keys, the Florida east coast south of Boca Raton, throughout the Caribbean, and in the western Gulf of Mexico coast outside U.S. waters can also support staghorn (*Acropora cervicornis*) and elkhorn (*Acropora palmata*) corals. These are the only two marine benthic invertebrates along the Atlantic and Gulf coasts currently listed under the ESA, both presently as threatened but being considered as endangered. Seven more species of Atlantic and Caribbean corals, all of which predominantly occupy waters less than 30 m, although some can also range more deeply, are currently under consideration for endangered status—pillar coral (*Dendrogyra cylindrus*), boulder star coral (*Montastrea annularis*), mountenous star coral (*Montastrea faveolata*), star coral (*Montastrea franksi*), and rough cactus coral (*Mycetophyllia ferox*) or threatened status—Lamarck's sheet coral (*Agaricia lamarcki*), and elliptical star coral (*Dichocoenia stokesi*). Where staghorn and elkhorn corals may occur, this hard-bottom habitat is identified as Habitat of Particular Concern (HAPC), an even more protective habitat classification under the Magnuson-Stevens Act.

EFH designations are not uniformly applied to all sedimentary bottoms on the OCS, except along the entire Gulf of Mexico coast, where the entire continental shelf out to 110 fathoms is designated EFH to protect largely depleted populations of reef-associated fishes of high value. Elsewhere, EFH designations are applied to sandy bottoms where emergent hard substrata are common on OCS areas, such as the Frying Pan shoals off Cape Fear, North Carolina. Designations of sandy sediments around emergent hard-bottom rocky reefs in this area are in part a consequence of the observational and experimental study of Posey and Ambrose (1994) that revealed that demersal fishes occupying these emergent rocky ridges in Onslow Bay around Frying Pan shoals prey on sandy bottom invertebrates at distances ranging out 75 m from the rocky reefs. Emergent sand shoal features can also be considered for EFH designation because of their value in elevating the bottom above depths at which seasonal bottom-water hypoxia can develop (such as Ship Shoal in Louisiana) or where sand shoals are thought to function as mass spawning sites or juvenile settlement and foraging habitat for some demersal fishes of commercial or recreational value. Consequently, the sandy OCS features that provide important ecosystem services to fishes of value may require that any degradation associated with sand mining be minimal or short-lived.

4.2.1 OCS Borrow Areas as Important Benthic Habitats

Planning for and permitting of OCS sand mining must give special consideration of valuable seafloor habitats in five different contexts: 1) habitat for ESA-listed benthic species (the two acroporid corals and potentially seven more stony corals); 2) habitat that is designated as EFH; 3) sedimentary habitat that sustains commercially exploited populations of bivalve mollusks; 4) sand habitat supporting benthic invertebrate resources that serve either as emergent structural refuges for associated fishes and mobile invertebrates, which are prey to higher-order consumers, especially larger fishes, or else as critical prey sources for valuable demersal fishes, crabs, and shrimps, which in turn help feed diving birds, sea turtles, and marine mammals; and 5) topographically elevated sedimentary habitat serving to promote spawning and reproduction or possibly to guide migrations of important fishes, crabs, or shrimps. No study has demonstrated a topographical sand feature that serves as an irreplaceable "sign post" to guide migrations, but resource agencies raise this concern.

To properly consider impacts of sand mining on benthic communities, we first distinguish sand resources on topographically elevated sand shoals or ridges within ridge and swale complexes from resources present as surficial sand sheets or buried sand resources, the last of which are typically found in filled-in paleo river incisions or inlet sequences often capped by muddier surface sediments. Among sand shoals, those that are actively renewing may differ from relict shoals that no longer are actively supplied by a sand source because relict shoals would not be expected to recover their sand volume after repeated dredging, although shoal crest height may recover to pre-dredge elevations provided the depth cut is not excessive, there is sufficient along-shelf sediment transport, and the water depth is shallow enough and bed shear stress great enough to mobilize sediments (Dibajnia and Nairn 2011). In OCS ridge and swale complexes, patterns of benthic invertebrate abundance and diversity as well as patterns of abundance of fishes using those invertebrates as prey resources are not well established in the literature and include conflicting guidance to managing mining of sand resources. Diaz et al. (2004) demonstrated in sampling the ridge and swale complex off the Ocean City, Maryland inlet around Fenwick Shoals that the commercially valuable surf clam *Spisula solidissima* comprised 66% of the total biomass of shoal benthos, implying need for caution and mitigation in sand mining there. In studying macrobenthos on and around the Sandbridge Shoals off Virginia, Diaz et al. (2006) reported that sites off the ridge itself had about 2.5 times more benthic production than sites on the ridge. Nevertheless, the higher abundances of juvenile fishes associated with physically and biotically structured meso- and microscale habitat complexity on Fenwick and Weaver Shoals (Diaz et al. 2003) were benefiting from reduced rates of predation where structural refuges were provided in part by polychaete tubes. The fishes changed habitat occupation from day to night, with fish abundance during daylight higher around the more structured shoals but more abundant at night foraging heavily on benthos of the sand flats away from the shoals. Slacum et al.'s (2010) study of OCS shoals in the mid-Atlantic Bight confirmed greater fish abundances, richness, and diversity utilizing the flat, silty sand habitat than the pure sand of the shoals. Similarly, Ramey et al. (2009) sampled a system of rippled sandy bottom in 12 m of water off the New Jersey coast where ripple heights ranged from 5-15 cm. They showed that the troughs between ripples had higher density and diversity of benthic macrofauna as well as greater amounts of particulate organic carbon. Ship Shoal off Louisiana, where Gelpi et al. (2009) and Condri and Gelpi (2010) have shown such high importance of benthic prey resources to dense feeding and breeding aggregations of commercially valuable blue crabs, *Callinectes sapidus*, stands as a sharp contrast. It is possible that Ship Shoal and perhaps other similar shoals in that geographic area are atypical because of their value in elevating the benthos above the depth that experiences sustained summer hypoxia and anoxia from excess nutrient loading, and in elevating benthic microalgae into lighted waters where their enhanced production produces more oxygen (Grippo et al. 2009). Ridge and swale complexes can possess more structural complexity containing somewhat different sediment types and differing sediment dynamics, which translate into typical differences in benthic invertebrate composition (e.g., CSA et al. 2010; Dubois et al. 2009). In addition, sand shoals have been identified as important breeding grounds for crabs and fishes, such as the Ship/Trinity/Tiger Shoal complex off the Louisiana coast (Gilmore 2008; Gelpi et al. 2009; Condrey and Gelpi 2010), although Woodland et al. (2011) demonstrated that juvenile fish use of invertebrate prey on the flat, inner continental shelf bottom is essentially indistinguishable from juvenile fish usage of the estuarine bottom, perhaps the most highly valued fish nursery.

Sediment grain size plays a fundamental role in determining the composition of OCS benthic invertebrate communities (Gray 1974; Rosenberg 2001; Cooper et al. 2011). Although sand and mud typically differ dramatically in benthic species composition, there are no compelling studies that reveal which of these divergent benthic communities harbors more valuable prey for demersal fishes, crabs, and shrimps. Traditionally, the macrobenthos of sandy sediments has been thought to be dominated by suspension feeders, which benefit from stronger oscillatory bottom boundary flows associated with sand sediments because greater flow velocity implies higher influx of new particulate foods suspended in the water column (Rhoads and Young 1970). Muddy sediments have been traditionally characterized by greater numbers of deposit-feeding macrobenthic animals because of higher net deposition of organic-rich particulates where bottom flow velocities and shear stresses are lower (Sanders 1958; Rhoads and Young 1970; but see qualifications in Snelgrove and Butman 1994). Nevertheless, the main question about differences in function of OCS sedimentary habitats that remains unanswered is whether demersal fishes, shrimps, and crabs prefer sandy or muddy sediments as feeding grounds and whether their energetic rates of food intake of benthic invertebrates predictably differ between these two sedimentary environments. This issue has particular relevance to sand mining practices that may convert sandy bottom areas to mud or mud to sand. Excavating depressions into sandy habitats that fill with muddy sediments can occur where fine-grained overburden is removed and sand is dredged from underlying paleochannels (as in Louisiana), and where sand sheets (as in southeast Florida) or low-relief sand ridges (as in South Carolina) are dredged. In most other areas, the shoal profile and granulometry ultimately equilibrate under physically controlled environmental conditions. Where initially sandy sediments are removed and resulting topographic depressions are created (see Figures 2.9 to 2.12) and serve as a depositional basins, their refilling can occur by deposition of finer materials (e.g., Van Dolah et al. 1992, 1993). Consequently, the recolonizing macrobenthic community is likely to differ in composition from the original community. This switch in sediment character to oxygen-demanding mud can include generation of anoxia suppressing benthic recolonization. However, where current flows bearing oxygen are sufficient, the new benthic community may change from the replacement of sand with mud and thereby alter the value of its ecosystem service of feeding demersal fishes, shrimps, and crabs.

The potential for the recovering benthic community to differ from the initial community in part because of fining of sediments within a more quiescent dredge depression also can come into play after an overburden of muddy sediments is removed to access a buried sand resource (Palmer et al. 2008). Under these conditions, the initial overburden is judged to be too fine-grained to be chosen for mining, and it possesses a benthic community appropriate to fine sediments. The quiescent conditions inside the dredging depression after dredging ceases will also induce siltation, but the benthic community that develops will probably differ from the original one because the sediments may become even finer through the siltation of suspended particles from the water column and because this siltation rate within the topographic depression may be more intense, and thus represent an ongoing enhanced disturbance at least early in the refilling process (Palmer et al. 2008). The siltation and environmental changes related to it in Palmer et al.'s (2008) study were dependent upon the presence of large supplies of mud in the Atchafalaya and Sabine Basins in the central Gulf of Mexico, conditions that are extreme for the OCS on the Gulf and Atlantic coasts more generally. Additionally quiescent conditions inside the dredged depressions, when combined with organically rich siltation, may lead to stratification of the water column within the pits and near-bottom hypoxia; however, continuous sampling throughout the warm months, when oxygen demand is highest, has not yet been done to test this hypothesis. After natural infilling of the depression is complete, the siltation rate would presumably equal that of the surrounding mud bottom and similar granulometry of surficial sediments would ultimately be expected to develop, promoting recovery of a benthic community similar to that on the mud cap before its removal. Thus, over time frames of multiple years (Palmer et al. 2008), the sea-floor surface sediments within this type of mining site and their benthic community would be expected to revert back to the original conditions after dredging even without the costs of returning the initial overburden of muddy materials into the mining depression.

Refilling by transferring the muddy overburden back into the dredged depression, which is not currently required as a permit condition, would contribute to recovery merely by speeding up the process of filling the depression, while not greatly affecting the nature of the sediments or the composition of the benthic invertebrate community that would ultimately develop after natural infilling. This redirection of the muddy overburden would be impossible if this material has been already deployed elsewhere, as at Raccoon Island and Pelican Island, where the materials helped rebuild marsh platforms. In addition, redredging carries other costs, such as inducing another period of elevated turbidity and exposing sea turtles to potential additional takes. Disposal of the sediments in the surface mud cap onto the nearby sedimentary seafloor (see Figure 2.12 for the Sandy Point, Louisiana disposal area) would cause almost complete mortality of existing benthos from suffocation in the depositional footprint (Rhoads et al. 1978) as well as induce turbidity plumes that extend beyond the footprint with potential negative effects on nearby hard-bottom benthos and also on suspension feeders in nearby sedimentary habitat, as described below. These additional risks of disposal impacts can be avoided if beneficial use of some of the overburden can be made, as partially achieved at Raccoon and Pelican Islands.

4.2.2 Potential Direct, Indirect, and Cumulative Environmental Effects on Benthic Resources, Communities, and Habitats

The types of impacting mechanisms discussed in Section 3 that could potentially affect benthic resources, communities, and habitats are discussed in the following sections sorted by impacting mechanism. The spatial scope of direct effects of sand mining on benthic resources, communities, and habitats discussed here is extremely limited for all mechanisms, simply because the cumulative area dredged summing across all sites is miniscule when compared to the area of sandy seafloor habitat across the OCS depth ranges in which sand mining occurs. Even with anticipated expansion of demand for OCS sand resources, the proportion of the sandy seafloor habitat disturbed will likely remain fairly small. In addition, no process of great concern transports impacts far beyond the actual dredging footprint: turbidity and subsequent siltation covers a larger area than the dredging footprint, but concentrations of suspended particles decline quickly with distance and thus so do siltation rates. Mitigation measures are mentioned where applicable. Many of the mitigation measures presented in this section 4.2 are conceptual suggestions that have not been subject to testing nor examined for cost reasonableness.

4.2.2.1 Alteration of benthic habitat at the borrow area

Benthic habitat is defined by two components–its physical environmental conditions and its benthic biological resources. Understanding the impacts of the OCS dredging process on both components requires assessment of species-specific mortality rates as a function of type of environmental disturbance and resilience in their subsequent responses. Both the rates of species recoveries and the directional changes in community composition require study after dredging mortality. This need to incorporate understanding of recovery complicates and lengthens this section on sediment and benthos removal. Extensive study has been made on processes of recovery of benthic resources and communities, although unanswered questions remain. Somewhat less effort has been invested in studying recovery of bottom topography and sediment type, but informative data are available in some locations. Because the most immediate and drastic impacts on the benthic resources, communities, and habitats are caused by removal during dredging, the information on recovery processes and rates is presented in this section. Recovery also is mentioned and the processes discussed here are cited where appropriate under other mechanisms, such as increased sedimentation and deposition of fines (Section 4.2.2.3).

The most significant characteristics of the OCS habitats are bottom topography, substrate type, and hydrodynamic energy regime. For the relatively narrow range of water depths on the portion of the OCS that can be practically mined for sand resources, these three major physical environmental conditions largely determine the benthic biological community present and its recovery rate (Gray 1974; Snelgrove and Butman 1994; Foden et al. 2009). In addition, the recent history of disturbance to the benthic habitat, such as storms with intense wave action, dictates whether the benthic community is already in early stages of recovery or if it is comprised of more long-lived species characteristic of periods of environmental stability (Posey and Alphin 2002). Furthermore, the timing of disturbance relative to the reproductive seasons of the benthic organisms plays a role in the direction and speed of recovery (Diaz et al. 2004).

The most fundamental substratum distinction arises between hard-bottom and sedimentary substrates. In sedimentary substrates, sandy sediments differ from finer-sized silts and clays in environmental properties and in benthic inhabitants. Other environmental parameters that can influence benthic habitat suitability include water temperature, oxygen concentration, turbidity, pH, organic loading, and levels of inorganic and organic toxins. Benthic biological resources include microalgae and macroalgae where sufficient light penetrates to support photosynthesis, microbial populations of decomposers, and benthic invertebrates–both meiofauna, such as nematodes, harpacticoid copepods, and turbellarians, and macrofauna, such as polychaetes, mollusks, crustaceans, and echinoderms. The benthos can also have important feedbacks on the physical properties of the bottom habitat by providing sediment stability through microbially created biopolymers or by containing mats of tubes. Emergent large tubes of polychaetes can induce sediment erosion around the edges as boundary-layer velocities are amplified. Most evaluations of OCS benthic biological conditions include only macrofaunal abundance, biomass, species richness (and species diversity based on information-theoretic measures), and community composition, with body-size distributions, burrowing depths, and meiofaunal characterization only rarely considered. Studies of how dredging affects meiobenthic populations reveal large declines in abundance, but these studies were done in relatively shallow waters–not depths characteristic of the OCS (Rogers and Darnell 1973; Pequegnat 1975; Sherman and Coull 1980). Studies of OCS sand dredging impacts only rarely include meiofaunal responses (Brooks et al. 2004, 2006).

Only under some very rare circumstances would sand mining convert the seafloor from one to the other of the two major bottom types (hard and sedimentary). For example, some projects may involve sufficiently complete exploitation of thin, surface sand layers, or inadvertent overdredging of surficial sand, overlying a hard pavement to leave the hard-bottom habitat behind. In some instances, unconsolidated hard-bottom rubble that becomes exposed can even become harvested by the dredge. In these cases where surface hard bottom remains after dredging, epibiota would recruit to the hard bottom, enhancing biogenic structure through emergent clonal invertebrates such as sponges, brozoans, and hydroids, and thereby provide both physical refuges and foraging grounds for crustaceans and fishes, as well as for larger consumers such as sea turtles, diving piscivorous and molluscivorous seabirds, and marine mammals (e.g., Marsh et al. 1978; Turbeville and Marsh 1982). Whether this enhancement of biogenic structure and increased utilization by demersal nekton would persist would depend upon if and when active sedimentation returns the site to a bottom comprised of surficial sediments. Long Bay in South Carolina, for example, has large areas of the inner and some outer continental shelf that can be characterized as flat hard bottom alternately covered by a thin veneer of sand and then uncovered even in the absence of dredging for sand (Gayes et al. 2003). The alternative conversion from hard to sedimentary bottom may inadvertently result from siltation after dredging sand from a borrow area, or when depositing those sediments into, or later redredging them from, a rehandling site (addressed in Section 4.2.2.3).

Effects of the locally intense disturbance caused by extraction of sand and its benthos at a borrow site are best understood within the context of disturbance theory and ecological succession to produce a synthetic general model of benthic habitat impacts and their consequences to higher trophic levels (Oliver et al. 1977; Rhoads et al. 1978; Bolam and Rees 2003; Hill et al. 2011). The intensity of dredging during sand mining has the immediate effect of also removing the large majority (see below for data reviews) of benthic invertebrates at the borrow sites, defined as those locations actually dredged within the borrow area, as the dredge harvests the sediments. Notably, the loss of benthic organisms in sandy sediments could include one of four species of exploited bivalve mollusks: the surf clam or ocean quahog found from Cape Cod to Cape Hatteras, the ocean scallop in the Mid-Atlantic Bight, Georges Bank, or Gulf of Maine, or the calico scallop found regularly off the east coast of central Florida, occasionally off Florida’s west coast and rarely off North Carolina (last fished there in the early 1970s). Furthermore, deposition of these sediments at a rehandling area probably results in sufficiently deep burial to kill most existing benthos at rehandling areas as well. The physical disturbance in both cases eliminates such a large fraction of the resident benthos that this sets the stage for secondary succession to determine the development of the new replacement community of benthic invertebrates in the surface layers of the seafloor sediments. Thus, applications of ecological disturbance theory and an understanding of resilience as achieved by benthic community recovery guide the most compelling syntheses of empirical information and predictions of impacts of OCS sand mining (Rhoads et al. 1978; Newell et al. 1998; Bolam and Rees 2003; Hill et al. 2011).

The initial step towards recovery of local bottom topography and benthos occurs through slumping and slope re-equilibration of the sides of the dredge depressions (Cooper et al. 2007). Under oscillatory bottom boundary flows, rippling and other bedforms will develop, creating meso- and microscale structures that can facilitate larval settlement and benthic invertebrate recovery. Initially, the slumping simply redistributes benthic animals, although at a cost of additional mortality of benthic invertebrates by burial, suffocation, and abrasion (Barry et al. 2010, based on data in Kenny and Rees 1994, 1996). Lateral migration of juvenile and adult benthic organisms into the dredged depression may help accelerate recolonization (Brooks et al. 2004, 2006). Yet the majority of benthic biological recovery occurs through subsequent larval settlement and interactive community development, analogous to recovery from other mass perturbations of seafloor sediments and benthos at these depths where sand mining occurs.

Rhoads et al. (1978) along with McCall (1977) characterized the process of succession of macrobenthic invertebrates in seafloor sediments that have been subjected to intense disturbance (in this case, intense deposition of fine sediments in Long Island Sound) by describing the life history characteristics of colonizers during three successive temporal phases of succession. McCall 1977) conducted an experimental investigation into macrofaunal succession via larval colonization of defaunated fine sediments placed on the seafloor of Long Island Sound in a design that provided replication and ability to sample colonization over time. This study identified three phases of succession, each characterized by species of different life histories. Group I (characteristic species of Phase I in succession) species are small-bodied, quickly maturing, rapidly reproducing, shallow-borrowing, and surface-occupying species that possess an opportunistic life history. This Group I suite of invertebrates does not burrow to depth in the sediments, but they do begin the process of disturbing and oxygenating the surface sediment layers (Rhoads et al. 1978).

In contrast, Group III species are large-bodied, do not engage in multiple reproduction events annually, have low death rates, do not exhibit high recruitment rates, and tend to live deeper into the sediments. This set of life-history characteristics qualifies these species as equilibrium species typical of the climax stage of benthic community succession. Group II species have life-history characteristics that are intermediate between these two end members. Although the details identifying those species and taxa that belong to Group I, II, and III are specific to the colonization of fine sediments in Long Island Sound, the concept of species succession after a major defaunating disturbance applies generically to recolonization of all sedimentary seafloor habitats after a disturbance has eliminated most of the organisms. As such, this succession concept clearly applies to recovery of seafloor sediments from OCS sand mining and from deposition and redredging of sand from rehandling areas (Bolam and Rees 2003; Hill et al. 2011), each of which involves near defaunation of relatively large areas of seafloor sediments.

Recognition that a generic succession model matches the known dynamics of benthic macrofaunal recovery on the seafloor after major disturbances provides insights into predicting recovery rates. In particular, those environments in which intense bottom disturbance is more frequent will be characterized by benthic macrofaunal communities that are naturally maintained in earlier successional stages. As such, those more frequently disturbed environments are expected to recover their benthic community more rapidly after a major defaunating disturbance (Bolam and Rees 2003). Consequently, recovery of seafloor benthic communities in a given sediment type would be expected to be quicker in shallower sites because wave energy would be expected to produce energetic, oscillatory bottom boundary flows at shallow bottoms more frequently than at deeper bottoms. This pattern has been demonstrated off the Monterey, California coast and interpreted in the context of succession on the path to recovery (Oliver et al. 1977). Furthermore, at any given ocean depth, mobile sandy substrates may be expected to experience more frequent disturbances than muddy bottoms, implying that time to recovery for those defaunated sediments will be shorter. This set of theory-based predictions helps provide a framework with which to interpret the body of empirical evidence of recovery of seafloor benthos after a variety of defaunating disturbances. Dernie et al. (2003) conducted a disturbance experiment on the continental shelf in the U.K. by excavating pits and then following the recovery of their sediments and biota. Clean sand sediments exhibited the most rapid fill rates and benthic recovery, whereas more stable muddy sands had the slowest rate of topographic recovery and benthic faunal rebound, consistent with the predictions that more frequently disturbed seafloor habitats can recover more quickly from disturbance.

Empirical studies of benthic macrofaunal recovery after sand mining or temporary placement at a rehandling area have applied a range of different biological metrics as quantitative measures of degree of recovery. Recovery metrics can be divided into: 1) univariate metrics that combine multispecies data into one number, such as total density, biomass, number of species (species richness), species diversity (using an information-theoretic index), and evenness or equitability of species abundances; 2) multivariate measures, such as the similarity in community composition as defined as the suite of species abundances; or, far less frequently, 3) functional measures, such as secondary productivity or energy transferred to predatory trophic levels. Earlier studies of benthic community recovery after a major disturbance tended to use some selection of univariate metrics because development of software with powerful statistical methodologies to compare species compositions dates from only about 25 years ago. The major problem posed by using univariate metrics is that each value of a univariate metric can be achieved by an almost unlimited number of different community compositions.

A problem posed by using community composition is the inability to know how closely a community must resemble the pre-disturbance composition to be considered an adequate recovery of function. Explicit functional measures represent a closer approach to measuring what is valuable about the macrobenthos. This is especially true of a functional measure that quantifies the energy or biomass flow from the benthos to predatory fishes, crabs, and shrimps because that reflects the most highly valued ecosystem service of the benthos that environmental policy is designed to protect. The problem with applying this functional measure to evaluate recovery of macrobenthos is the lack of any practical method to measure energy, biomass, or materials flows from the benthos to the demersal predators, although knowledge of diet composition and energetic requirements of the predatory species may help. An alternative measure that may address functional value might be to measure secondary production of the seafloor benthos; however, this introduces many technical challenges when done on a suite of different taxa and, to our knowledge, no secondary production measure has ever been applied for this purpose. Nevertheless, an enduring question prevails in determining recovery of the benthos and that question is how the recovering community compares to the pre-disturbance community in its capacity to feed valuable demersal predators. A further major functional role of the seafloor benthos that is universally recognized is its microbially mediated biogeochemical services in sustaining critical elemental cycles that sustain the world's biosphere. This role is not usually considered in environmental assessment of sand mining because, collectively, all sand mining projects affect such a small fraction of the seafloor that modification of global elemental cycles is inconceivable.

The large number of empirical studies documenting how univariate measures of macrobenthic community status change during recovery from a major disturbance, when combined with the conceptual development of disturbance theory, in which recovery is understood as a process of successional change in life history of dominant species in seafloor sediments, has allowed construction of a widely accepted, generic recovery model for benthic invertebrate communities after dredging (Hill et al. 2011). In developing this model, Hill et al. (2011) synthesized results of studies of dredging impacts in the U.K., which target gravel and coarse sand, a different habitat from the sandy bottoms that provide sand for beach nourishment in the U.S. Hill et al. (2011) discovered that the immediate effects of dredging resulted in declines in macrofaunal abundance ranging from 40-95% and declines in species richness (numbers of species) ranging from 30-70%. Our similar compilation here of multiple studies of sand dredging in the U.S. OCS reveals immediate-to-short-term declines in macrofaunal abundance ranging from 45-88% (79%-Kaplan et al. 1974; 86%-Oliver and Slattery 1976; 74-88%-McCauley et al. 1977; 72%-Johnson and Nelson 1985; 45-75%-Bergquist et al. 2009a, 2009b) and in species richness ranging from 25-60% (60%-Oliver and Slattery 1976; 25%-Bergquist et al. 2009b) in borrow areas just after dredging. Thus, the immediate impacts of dredging on benthic macroinvertebrates are large and very similar in the U.K. and U.S.

When dredging ends, macrobenthic recovery begins. This process can be best understood as succession (Figure 4.1). Benthic abundances increase dramatically (Turbeville and Marsh 1982; Saloman et al. 1982; Deis et al. 1992 (cited in Greene 2002); Jutte et al. 2001a), as opportunists settle from planktonic larvae, attaining a maximum density in about six months that is typically much greater than the pre-dredging abundance level (Pearson and Rosenberg 1978; Rhoads et al. 1978). Then the abundance exhibits a sharp decline, representing high mortality of opportunists driven either by food shortage from overpopulation or by predation from demersal consumers, or both. Biomass and numbers of species also initiate rapid increases immediately after the dredging ends. Biomass exhibits a decline at the same time that abundance declines but, unlike benthic abundance, biomass exhibits a second increasing phase as the "equilibrium," long-lived species begin to grow in size and biomass and replace the dying opportunists.

If the environment is stable enough physically to support a Group III equilibrium community, then, as these slower-growing, longer-lived species replace most of the opportunists and dominate, numbers of species exhibits a final decline because of competitive exclusion of poorer competitors, as described by the intermediate disturbance hypothesis (Connell 1978). Biomass may also decline as the equilibrium community becomes fully established. This temporal progression of the key univariate community metrics during the recovery process is illustrated as Figure 4.1. Species diversity, as defined by information-theoretic indices, such as the Shannon-Wiener H', and evenness or equitability indices, may not exhibit predictable patterns through recovery, but the early phase of heavy settlement by opportunists is typically associated with low evenness because small numbers of a few species of opportunists are usually hyper-abundant. This low evenness suppresses H' even though numbers of species rise as more opportunists and growing numbers of equilibrium species colonize the dredged bottom. H' generally rises in the middle stages of recovery, but shows a decline at Phase III as the best competitors come to dominate. Although these patterns in univariate metrics generally apply to benthic recoveries after disturbance, this insight provides incomplete guidance to natural resource managers because the patterns do not reflect changes in value of ecosystem services (Hill et al. 2011).

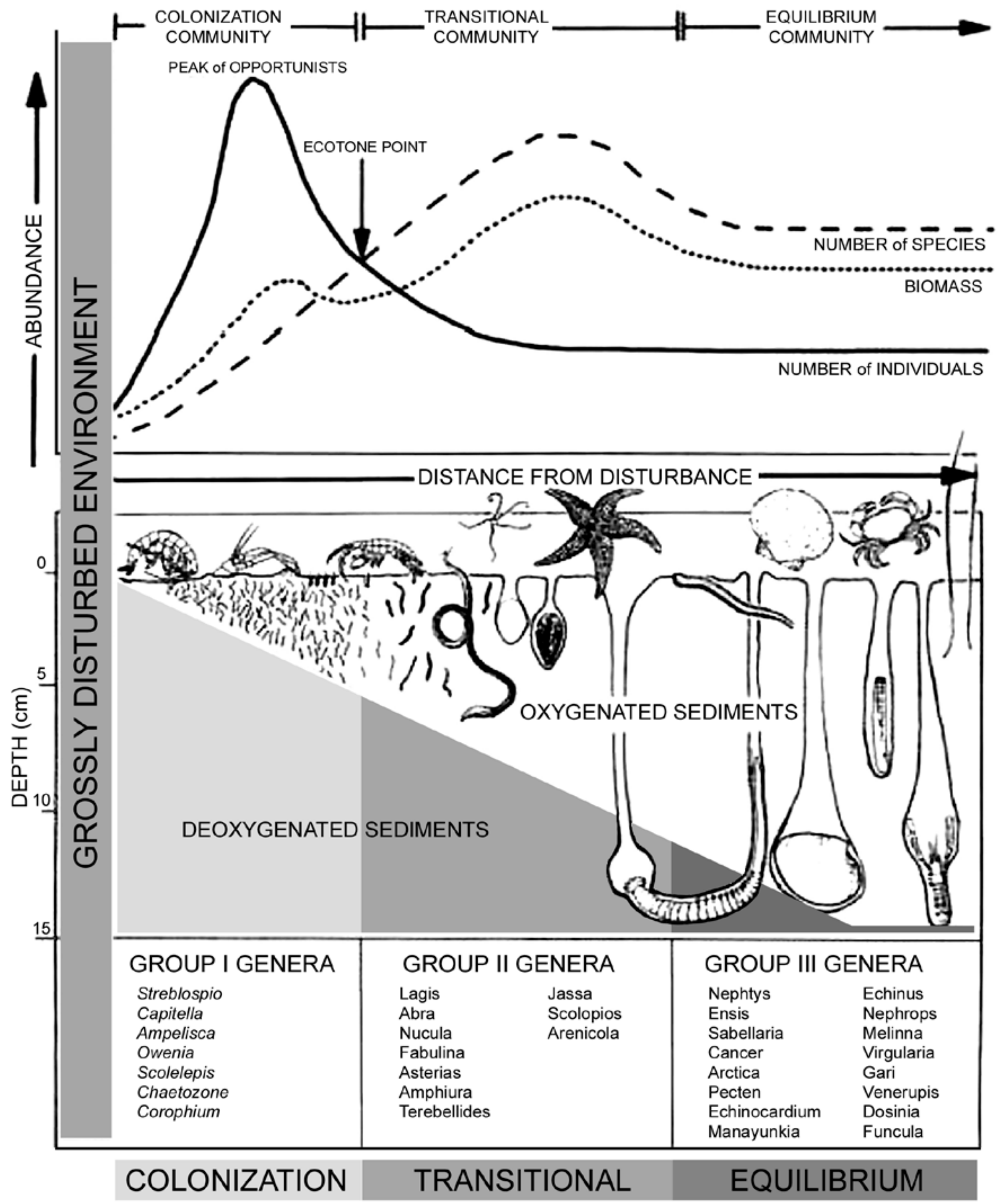


Figure 4.1 The process of 'Ecological Succession' in marine benthic communities through a gradient of environmental disturbance. From Newell (1998), based on Pearson and Rosenberg (1978).

If the initial community structure and benthic abundances and biomasses are restored, then it is reasonable to conclude that the functions and ecosystem services of the benthic community are also restored. However, the converse is not also true: benthic communities that differ dramatically in taxonomic composition may well provide the same or higher levels of the most widely acknowledged ecosystem services of value associated with sand shoals on the OCS–feeding demersal and even pelagic fishes, crabs, and shrimps, and providing important habitat for the reproduction and other critical functions of these economically important predators. Sand ridges can enhance feeding by demersal fishes by supporting high production of suitable benthic prey, whereas pelagic fishes may also benefit from greater foraging success if the physical structure of the ridge-and-swale geomorphology interacts with hydrodynamic processes to concentrate pelagic prey or otherwise enhance their availability to consumers (CSA et al. 2010). Van Rijn (1993) depicts the effects of bottom sand ripples on boundary-layer flows, as observed in both flume and field observations, showing formation of clear eddies. Eddies like these can be generated by boundary-layer flows interacting with fields of perpendicular sand ridges or bottom ripples of various scales. The flow dynamics of eddies concentrate and can induce deposition of suspended particles, probably enhancing the feeding ability and efficiency of planktivorous fishes. This expectation has yet to be tested, but could form an important mode of association between sand ridge topography, boundary-layer flow regime, and biological benefits. Despite this possible role of sand ridge topography in promoting coupling between demersal fishes and their prey, restoration of the same sedimentary habitat and benthic community is not a prerequisite for restoring original, valuable ecosystem services. In cases where long-lasting changes in bottom sedimentology occur as a consequence of sand dredging (as discussed Section 3.3.1), the benthic community that develops after dredging ends is unlikely to resemble the initial undisturbed sand community because benthic invertebrates are sensitive to sediment composition. It is conceivable, but as yet not tested, that energy flow to demersal consumers could be as great or greater on muddy-bottom communities than on sandy-bottom communities because of equal or greater value of the mud-bottom benthos as prey.

Cooper et al. (2008) examined and compared the effects of dredging for gravel in the U.K. on ecosystem functions using indices that characterize the richness and evenness of functions, rather than of species (as done in traditional techniques). Five analytical metrics that quantify ecosystem function were considered suitable for use with available data: Infaunal Trophic Index, Somatic Production, Biological Traits Analysis, Taxonomic Distinctness, and Rao's Quadratic Entropy coefficient (Cooper et al. 2008). All measures showed congruence with each other and with traditional univariate metrics except for Rao's Q, which implied a hard-to-explain increase in functional diversity during times of active dredging (Cooper et al. 2008). Such functional indices are not widely used and are still in a development and testing stage. In a subsequent but related study, Cooper et al. (2011) compared the Infaunal Trophic Index, Taxonomic Distinctness, and Rao's Quadratic Entropy coefficient to traditional multivariate analyses in examining the recovery of the functional diversity of sites that experienced intense, light, and no dredging for gravel, along with screening at the ship to remove fine sediments. This new study also assessed sediment size distributions in the two dredged and two reference sites. While the functional indices suggested functional recovery of the low-intensity dredge site after only two years, multivariate analyses of community composition using the same data indicated little difference between intense and light dredging treatments, with neither being functionally equivalent to the reference site four years after disturbance. Collectively, ecosystem function metrics were interpreted to indicate that functional diversity of dredged sites had not recovered even five years post-dredging (Cooper et al. 2011). The effects of dredging disturbance were augmented in this study by effects of screening the dredged materials at shipboard to remove fine sediments. This had the effect of rendering the granulometry of dredged sites dramatically different from reference sites by removal of both gravel (coarse) and fine sand and silts (fine) sizes of sediments. These sedimentological impacts were invariant over the four years of presumably post-dredging sampling, showing no recovery toward reference conditions, and thus providing a likely explanation for why the benthic communities remained different between dredged and screened sites and reference sites. The construction of these ecosystem function measures can be challenging because information required to compute them can be very data intensive and results can be highly specific to local taxonomic composition. Further development of functional measures may provide more universally applicable and conceptually compelling metrics of function, which would represent a welcome advance.

The nature of the disturbance can dictate some important specifics of the benthic community that ultimately develops, but the process of succession is common to all types of disturbance that virtually eliminate the existing benthos. This allows benthic studies of many different intense disturbance agents to be included in synthesizing information to produce a powerful conceptual model of how the biological aspects of benthic habitat will recover after sand mining and from sand deposition at a rehandling area. Listed chronologically by dates of publication of the key informative studies describing the process of marine benthic community succession, the types of well-studied disturbances that make the most substantive contributions to our conceptual understanding of benthic biological habitat resilience are organic enrichment (from paper mill wastes, Pearson and Rosenberg 1978), wave disturbance of bottom sediments along a depth gradient (Oliver et al. 1977, 1980), bottom disposal of dredged fine sediments (Rhoads et al. 1978), sand and gravel extraction at borrow areas to supply beach nourishment and aggregate (Newell et al. 1998; Hill et al. 2011), and intense and repeated bottom disturbance by commercial fishing gear such as shellfish dredges and trawls (Collie et al. 2000).

Important factors in recovery of dredged sites are the recent history of benthic disturbances prior to dredging, the season during which dredging is carried out relative to the timing of reproduction of the invertebrates comprising the benthic community, the depth to which the sediments are dredged, the hydrodynamic energy regime of the borrow area, the local availability and sedimentology of sediment sources nearby to fill the dredged site, and the topography after dredging. Benthic communities within the shallow continental shelf to depths of up to 30-35 m, which includes OCS areas of interest for sand mining, are influenced by natural disturbances, particularly storm-generated oscillatory waves. The effects of such waves influence communities at shallower water depths more than at depths below about 30 m, and where the disturbance of surface sediments is great enough, such events can induce mortality of benthic invertebrates and thus displace the benthic invertebrate community back towards an earlier successional stage (Posey and Alphin 2002). Although Posey and Alphin (2002) did not detect any dramatic effect of a hurricane on borrow area communities at 15-20 m depth off Cape Fear, North Carolina, wave energy from intense storms has potential to induce disturbances to the benthos and differentially affect the longer-lived Phase III species. Consequently, if sand dredging were conducted just after an intense natural disturbance, the time required for recovery of this Phase I or II benthic community could be shorter than if the community had been a Phase III community at the start. Similarly, because winter conditions are often stormier, there is often a seasonal cycle of benthic community composition on relatively shallow sand sediments in OCS waters below about 20 m depth (Boesch 1979).

Benthic invertebrates exhibit strong seasonality in reproduction, meaning that the recovery after dredging disturbance can be facilitated and the nature of recovery directed by appropriate seasonal timing of dredging for sand on the OCS. Not all benthic taxa reproduce most intensively during the same season, so timing of dredging can select for dominance of different taxa during the recovery process. Diaz et al. (2004) suggested that dredging around Fenwick Shoals in Delaware before spring/summer would allow the crustaceans to settle and grow afterwards, whereas dredging before fall/winter would favor polychaetes. Because juvenile fishes there strongly prefer crustacean prey, Diaz et al. (2004) further recommend that sand mining be completed before the spring/summer season. More generally, Hobbs (2002) recommends avoiding sand mining during the summer season when benthic invertebrate abundances and biomass are typically highest on sand in the OCS and juvenile fishes that seek benthic prey are most abundant.

When relatively deep pits are excavated in surficial sand deposits, they can exhibit low resilience by their slow recovery if located in areas with low hydrodynamic energy and low rates of bed load or suspended load transport of sediments along the bottom. Thus dredged depressions in shallow sand resources are expected to fill more rapidly than similar-sized pits on deeper sand formations. However, availability of sediments also affects rate of sand and topographic recovery. Some shallow sand sheets are located near riverine sources of fine sediments, providing potential for fill to differ in sedimentology from the sands that were extracted. As a consequence, such topographic depressions can become filled in, not rapidly by sand but slowly by deposition of silt and clay from suspensions in the water column. Deposition of fine sediments within the dredged sites can have short- and long-term impacts on the composition of the soft-bottom benthic community that recolonizes the dredged depressions. For example, one (on Joiner Bank) of two borrow areas dredged off Hilton Head, South Carolina in State waters, was only slowly filled by muddy sediments, exhibited higher turbidity levels, and within three months had developed a benthic infaunal community that differed in composition from the sand bank community and remained different after two years (Van Dolah et al. 1992; 1993). Subsequent mining of these same shoals (Joiner and Gaskins Banks) again resulted in a fining of surficial sediments and community shifts of the benthos within the borrow pits lasting at least seventeen months after a 1997 dredging event (Jutte and Van Dolah 1999; Jutte et al. 2001a, 2001b), and fourteen months post-dredging for a 2006 dredging event, even where the mining depth was reduced from 5-6 m to 2.4 m below grade at Joiner Banks (Bergquist et al. 2009a). Similar results were observed at two borrow areas in State waters for a Folly Beach, South Carolina project, where the sediments and benthos remained altered at for at least twelve months post-dredging (Bergquist et al. 2008), although faunal recovery did occur at one pit from which samples were collected after 24 months (Bergquist et al. 2009b). Presumably, oxygen levels remained high enough within the Joiner Banks pits to support necessary respiration of benthic invertebrates, but this is not always the case.

Unfortunately, the South Carolina examples have not been studied for a long enough time after dredging to know whether the conditions created represent long-term equilibria or transient states. Imposition of a major hurricane disturbance may reset the status, for example. In addition, such extensive study of the sedimentology within dredged depressions has not been repeated elsewhere, so replication, especially in OCS waters, is needed to develop the ability to predict resilience and recovery of dredged depressions. Multi-year suppression of recovery of sedimentary invertebrates in deeper dredge pits can occur under conditions that are not entirely predictable, but caused by insufficient oxygen in the sediments, stratified bottom waters, and consequent chemical inhibition (see Section 4.2.2.4). Such cases represent a more serious impact than what occurred on Joiner Banks, where the mud-associated benthic infaunal community that developed presumably served at least some, and maybe even most, of the trophic and biogeochemical functions of the initial sand-associated benthos. Prior to the Hilton Head study, Oliver and Slattery (1976) had demonstrated that even if oxygen levels remain high, the benthic invertebrate community that develops in sediments made muddy by dredging will differ dramatically from the pre-dredging community characteristic of coarser sandy sediments. A borrow area that refills through deposition of fine sediments instead of sand cannot be reused for subsequent extraction of sand resources for beach nourishment and other uses, implying that future sand extraction must be done in different locations, spreading the cumulative area of bottom disturbance (Van Dolah et al. 1998).

Another set of studies in coastal South Carolina helps develop some further understanding of how depth of dredging for sand and morphology of the dredging pattern may influence recovery of benthos. During a 1996-1998 beach nourishment project along South Carolina's Grand Strand, three sand shoals were mined in succession along the 96-km beach restoration site. All borrow areas had a 1-m target excavation depth. Sand from the Cherry Grove and Surfside Beach areas was excavated using a hopper dredge that left undisturbed strips between mined furrows throughout the borrow area; the undisturbed fauna that remained between 1-m-deep furrows was expected to facilitate recovery (Whitlatch et al. 1998; Jutte and Van Dolah 1999). A hydraulic dredge was used in the Cane South borrow area, where approximately 1.5 m of sand were removed from the area (Jutte et al. 2001a). The three borrow areas differed in recovery rate in a pattern that implied that energy regime and availability of nearby sediment supply were major determinants (Jutte et al. 1999; 2001a, 2001b). Dredge pit morphology remained evident at Cane South three years after hydraulic dredging; sediments were significantly finer three years post-dredging and benthic community composition remained altered for at least two years post-dredging, although faunal abundance, diversity, and species evenness were similar to the reference area after only one year (Jutte et al. 2001a). The slowest recovery rate among the three borrow areas occurred at Cherry Grove, where sediment character and the benthic community composition remained altered for three years post-dredging, although benthic abundance, diversity, and species evenness were similar to the reference area after one year (Jutte and Van Dolah 1999). Recovery at the other hopper-dredged area, Surfside Beach, differed greatly from that at Cherry Grove. Both sediment character and benthic community composition recovered within 3-6 months post-dredging at Surfside; however, a general decrease in faunal abundance was observed at both borrow and references areas, raising the question whether broad-scale changes in the benthos had occurred across the 96-km project region (Jutte et al. 2001b). Even if such regional changes were occurring during the study period, they do not affect the rigor of the conclusions about retarded recovery at Cane South and Cherry Grove because at each of those places the impacts were judged against reference area conditions and the benthic biology was matched by a persistent switch in sedimentology. Distinct differences in energy regime and sand availability were observed between the two areas mined by the hopper dredge. An abundant supply of mobile sand rapidly replenished the Surfside borrow area, while the benthic habitat remained significantly altered at Cherry Grove, where around the borrow area only thin veneers of sand were present atop the hard bottom that dominated that area (Jutte and Van Dolah 1999; Jutte et al. 2001b). Because more than one variable differed among the Grand Strand borrow areas, the mechanisms that determined recovery rates for sediments and benthos remain unclear. Nevertheless, borrow area recovery rates were generally more rapid on the Grand Strand project, where depth of sand extraction was limited to 1.5 m below original seafloor depths (Jutte et al. 2001b) than at Folly Beach and Hilton Head with their deeper excavations (Van Dolah et al. 1998; Jutte et al. 2001b). It is unclear to what degree differences in suspended sediment supply from riverine inputs may exist among the three study sites and confound these conclusions.

Bergquist and Crowe (2009) conducted meta-analyses to examine dredging impacts using the large database available from beach nourishment projects studied in South Carolina, which are summarized in Table 4.1. The majority of these borrow areas were in State waters. Their examination of sediment and benthic biological characteristics for thirteen borrow areas consistently showed a pattern of altered sediment character but variable benthic invertebrate responses without any compelling pattern. Bergquist and Crowe (2009) found statistically significant decreases in sand content and reductions in grain size, with increases in silt and clay content and in organic matter content. These responses were significant when assessed at 0-3, 6-9, and 12-15 months after dredging ended. The benthos was similarly modified over those same time frames with lower numbers of species, lower macrofaunal densities, and higher percentages of polychaetes among the taxa in each time frame after dredging.

At dredged sites where excavation depth was shallow (e.g., Grand Strand), the accumulation of fines was minimal (Jutte and Van Dolah 1999; Jutte et al. 2001a, 2001b), but at deeply excavated sites (Table 4.1), fines comprised over 25% of the material that refilled the resulting pit (Van Dolah et al. 1992; Jutte and Van Dolah 2000; Bergquist et al. 2008). Variance among sites in the relationship between excavation depth and the percentage of fines during subsequent filling of the depressions is probably driven by differences in sources of bedload and suspended sediments. Benthic communities varied in rate of recovery and in similarity to initial community composition (Bergquist and Crowe 2009). Similar variability in benthic recovery has been documented in Florida (e.g., Naqvi and Pullen 1982), although the replication of such studies is far less than in South Carolina. While benthic abundance uniformly decreased rapidly with dredging, recovery of abundance, diversity, and evenness relative to reference sites often occurred within one year (Naqvi and Pullen 1982; Jutte and Van Dolah 1999, 2000; Bergquist and Crowe 2009). Nevertheless, many of these studies documented development of benthic community differing in composition twelve to fifteen months after dredging, even though univariate benthic metrics become similar to those of reference sites (Jutte and Van Dolah 1999; Jutte et al. 2001a; Bergquist et al. 2009b).

Table 4.1

Recovery of sediments and benthos in borrow areas for beach nourishment projects in South Carolina (listed from north to south).

Fill Site/ Borrow Area	Borrow Location (km /nm offshore)	Date Dredged	Dredged Depth (m)	Proximity to Inlet/ River	Recovery of Sediments	Recovery of Benthos	Reference
Grand Strand/ Cherry Grove	nearshore in State waters/ <1.6 nmi	Sep 96 - Apr 97	~1	northern portion near Hog Inlet	Yr 1: fining; ↓ sand; ↑ silt/clay; ↑ phi size; ↑ OM; some in-filling of sediments, which may have been redistribution from ridges Yr 2: remained fine; slow in-filling and coarsening of sediments Yr 3: ↓ % sand v. pre and ↑ % $CaCO_3$ v. pre (both significant)	Yr 1: abundance, species diversity, evenness and community composition similar to reference site; shift to more polychaete dominance v. ref Yr 2: abundance, etc. recovered, yet community remained altered Yr 3: general recovery, diversity and abundance higher v. pre	Jutte et al. 1999 - Phase I
Grand Strand/ Cane South	nearshore in State waters/ <1.6 nmi	Apr-Jul 1997	1.0-1.5		Yr 1: fining;↓ sand; ↑ silt/clay; ↑ phi size; ↑ % $CaCO_3$; ↑ % OM v. ref and v. pre Yr 2: remained fine; slow in-filling Yr 3: poor recovery (but recovery rate faster than Joiner, Gaskin), dredged morphology evident	Yr 1: ↓ density and ↑abundances v. ref; diversity similar to ref & pre Yr 2: ↑polychaetes, community shift; great variability by seasons	Jutte et al. 2001 - Phase II
Grand Strand/ Surfside Beach	nearshore in State and federal waters/ <2.5 nmi	Sep-Nov 1998	~1		Yr 1: little change in % sand/silt/clay or CaCO3, similar silt/clay/OM to ref and similar $CaCO_3$ to pre); rapid recovery (3-6 mo)	Yr 1: recovered rapidly and similar to reference; community shift; both Surfside borrow and reference areas experienced decreases in abundance	Jutte et al. 2001 - Phase III
Grand Strand/ Little River	4.8 / 2.6	Sep-Nov 2008	0.1-0.3		Yr 1: ↑ OM (significant prior to 12 mo) v. ref	Yr 1: ↓ density, spp. evenness and diversity; composition variable thru time v. pre; ↓ abundance v. ref (significant prior to 12 mo)	Bergquist et al. 2011; McCoy et al. 2010
Grand Strand/ Cane South	4.8 /2.6	Aug-Dec 2008	0.1-0.3		Yr 1: ↑ silt/clay; ↑ phi size;↑ OM (significant prior to 12 mo), $CaCO_3$ unchanged v. ref	Yr 1: community composition altered v. pre; ↓ richness and density (significant thru 12 mo), evenness depressed, diversity differed little v. ref	Bergquist et al. 2011; McCoy et al. 2010
Grand Strand/ Surfside and Garden City	≤7.0 / ≤3.8	Nov 2007 -Feb 2008	0.1-0.3		no assessment	no assessment	Mc Coy et al. 2010
Folly Beach/ Site A (S end)	5.3 / 2.9	Apr-Oct 2005	3.0	near Charleston Harbor	Yr 1: fining; ↑silt/clay (3.4 x ref), ↑phi size, ↓$CaCO_3$ v. pre and v. ref Yr 2: ↑silt/clay, ↑phi size, ↓CaCO3 v. pre; ↓$CaCO_3$ v. ref	Yr 1: ↓density, ↓richness, ↑evenness v. pre and v. ref Yr 2: recovered univariate metrics	Bergquist et al. 2008

Table 4.1 Recovery of sediments and benthos in borrow areas for beach nourishment projects in South Carolina (listed from north to south) (continued).

Fill Site/ Borrow Area	Borrow Location (km /nm offshore)	Date Dredged	Dredged Depth (m)	Proximity to Inlet/ River	Recovery of Sediments	Recovery of Benthos	Reference
Folly Beach/ Area B; adj to 2005 site (NE end)	5.3 / 2.9	Apr-Jul 2007	3.0	near Charleston Harbor	Yr 1: fining;↑silt/clay, ↑phi size, ↓CaCO3 (75% < ref),↑OM (2x ref) v. pre and v. ref	Yr 1: ↓density v. pre; ↓density, ↓richness, ↑evenness v. ref; variable recovery; community altered	Bergquist et al. 2009b
Hilton Head/ Joiner Bank	4.0 / 2.2	1990	3.4	south edge of Port Royal Sound entrance channel	Yr 1: fining; ↓ 31% sand; sand cap over silt/clay lens	Yr 1: significantly altered	Van Dolah et al. 1992; Van Dolah et al. 1998
Hilton Head/ Gaskin Bank	3.7 / 2.0	1990	3.0		Yr 1: sand content almost recovered v. pre ; slow to fill	Yr 1: recovered	Van Dolah et al. 1993
Hilton Head/ Joiner Bank	4.0 / 2.2	May-Aug 1997	5.5-6.1		Yr 1: fining; ↓ 75% sand; 13%↑ silt/clay; ↑ phi size;↑ OM v. ref	Yr 1: community shift	Jutte and Van Dolah 1999, 2000
Hilton Head/ Gaskin Bank	3.7 / 2.0	May-Aug 1997	5.5-6.1		Yr 1: fining; ↓ sand, ↑ silt/clay, ↑ phi size,↑ OM v. ref	Yr 1: community shift	Jutte and Van Dolah 1999, 2000
Hilton Head Joiner Shoals	2.6 / 1.4	2006	2.4	south edge of Port Royal Sound entrance channel	Yr 1: severe fining,↑17x silt/clay, ↑phi size, ↑ 6 x OM (significant thru 12 mo) v. pre; ↑silt/clay, ↑phi size, ↑OM (significant thru 12 mo) and CaCO3 decreased over 12 mo v. ref	Yr 1: ↓richness, ↓diversity (significant thru 12 mo); ↓amphipods, ↑polychaetes, ↑mollusks (significant thru 12 mo) v. pre; ↓density (recovered by 12 mo), ↓evenness (not recovered by 12 mo), ↑diversity, community altered (thru 12 mo) v. ref	Bergquist et al. 2009a
Hilton Head Barrett Shoals	2.5 / 1.4	2006	2.7	near Calibogue Sound inlet		Yr 1: ↓richness, ↓diversity (significant thru 6 mo); ↓amphipods, ↑polychaetes (significant thru 12 mo) v. pre; ↓density (recovered by 6 mo), richness, evenness, diversity (depressed thru 6 mo), community altered (thru 12 mo) v. ref	Bergquist et al. 2009a

Legend: OM: organic matter; ref= reference site; pre= before dredging samples; $CaCO_3$= calcium carbonate

Mitigation measures most often recommended to accelerate recovery of benthic habitats and communities include: 1) dredging small portions of shoals that are expected to refill most rapidly and allow the feature to maintain its geomorphic integrity; 2) rotational dredging; 3) leaving some areas undredged; and 4) avoiding creation of deep pits. Within sand ridges, sand mining might best be directed towards morphologically specific features of the ridge and also done with a deliberate dredging design and cut-depth created to maximize recovery rate of the geomorphological shoal feature, the sedimentology, and presumably therefore the benthic biota. In particular, sedimentary depocenters and the leading edge or down-drift margin should be targeted for dredging, where sand is most actively accreting and thus where geomorphological recovery is likely most rapid (CSA et al. 2010; Dibajnia and Nairn 2011). The lee sides of sand ridges, relative to prevailing currents (e.g., the south side on Weaver Shoal (Diaz et al. 2003)), can be sites of abundant tubiculous polychaetes and shells providing important biogenic habitat structure and could be protected from dredging as a potential mitigation measure to minimize uptake of some of the highest densities of benthic resources and to sustain the surfaces of sediment stabilization by densely packed tubiculous polychaetes (CSA et al. 2010). The peaks of sand ridges and the topographically high portions of leading margins may be areas where dredging recovery of the geomorphology, sedimentology, and benthos is most rapid because bottom shear stress, turbidity, and sediment reworking is maximal in these higher positions, thereby supporting a benthic community of small, short-lived, rapidly reproducing invertebrates that have more rapid recovery rates than communities composed of larger, longer-lived invertebrates (Oliver et al. 1977, 1980; Newell et al. 1998; CSA et al. 2010; Hill et al. 2011).

Many suggest, although evidence is not yet compelling (Brooks et al. 2004; Brooks et al. 2006; Cooper et al. 2007; Barry et al. 2010), that dredging may be best done in strips with undisturbed sand ridges in between to provide multiple nearby local sources of sandy sediments and benthos to facilitate infilling and faunal recolonization via horizontal transport during slumping along the edges and active lateral movement of established adult and juvenile benthic organisms (Whitlatch et al. 1998; Jutte and Van Dolah 1999; Bergquist et al. 2011a; CSA et al. 2010). Unfortunately, rigorous testing the effectiveness of this particular suggested mitigation to speed up benthic recovery is technically difficult for dredgers and expensive. Given that slumping results in additional mortality of benthic invertebrates, that individuals that survive slumping do not represent net additions to replace the deaths but simply redistributions, and that areas modified by dredging are so large that larval settlement is ultimately needed to achieve recovery, there are good grounds to conduct a rigorous evaluation of this notion before accepting it.

Dredging might also be constrained to relatively shallow depths to avoid creating deep pits, some of which require long recovery times and fill slowly with fine-grained sedimentation (Palmer et al. 2008; Bergquist and Crowe 2009). The trade-off that accompanies shallow excavation of sand is the need for disturbing more surface area of the seafloor, where the benthos lives and functions, implying greater initial benthic mortality. Nevertheless, the slow pace of recovery in deeper dredge pits suggests that avoiding their creation may be advisable as a mitigation measure. A full assessment of the relative merits of this option and an accompanying test of whether the net biological impact over time is reduced or increased by changing cut-depth of dredging are still needed.

An effective means of minimizing loss of commercially valuable bivalve shellfish is the engagement of commercial fishermen to survey and then harvest any bivalve shellfish resource found on prospective borrow areas in advance of dredging. This practice does not prevent mortality but it avoids waste of commercially marketable stock. If the initial sediment character and the topography on meso- and microscales return after cessation of dredging, then the bivalve populations are also likely to recover, although perhaps with time lags because they do not necessarily exhibit large recruitment pulses every year.

4.2.2.2 Increased turbidity

In principle, elevated turbidity generated by persistent suspensions of finer particles during sand mining could have biological impacts, although field testing of this potential impact of dredging is essentially non-existent. Even the spatial extent of transport of turbidity plumes generated by dredging in the borrow areas is not well characterized, although the range of their extent is relatively small, measured in the thousands of m (Hammer et al. 1993). Higher turbidity reduces light transmission. Light energy failing to reach the seafloor can limit the production and growth of benthic macroalgae and microalgae, reducing primary production on the seafloor and potentially limiting energy available to support various food chains (Grippo et al. 2009). Cahoon and Cooke (1992) sampled the benthic and planktonic microalgae and measured primary production of each type on sandy bottoms from 14.6 to 41 m deep on the OCS of Onslow Bay, North Carolina, finding that the benthic microalgal production was on average virtually identical to the integrated phytoplankton production throughout the entire water column. This study emphasizes the valuable role of benthic microalgal production at the base of the food chain and the importance of sustaining light to support the production process on OCS sand habitats. In addition, the macroalgae growing vertically from hard-bottom habitat provide biogenic habitat for use by many mobile crustaceans and fishes. Sustained turbidity could stunt macroalgal growth and even kill macroalgae by sustained shading or siltation, thereby degrading an important bioengineered habitat and decreasing abundance of a food resource for green sea turtles and other herbivores.

Additionally, increasing turbidity has negative implications for individual growth and even survival of exposed suspension-feeding benthic invertebrates (Rhoads and Young 1970). Other biological responses to turbidity include reduced hatching success, slowed growth, abnormal development, tissue abrasion, and increased mortality (Wilber and Clarke 2001). Short-term exposure to elevated turbidity has less serious consequences than sustained exposures, such as are associated with a dredging project with a duration of weeks. As ocean currents change, however, any turbidity plume will be transported in different directions, thereby exposing different individual suspension organisms in new areas. By adding a thin superficial layer of fine sediments onto the seafloor, dredging can cause delayed incidents of enhanced turbidity over time in response to resuspension by wave-generated bottom shear stress during storms that recur probably over time scales of months after termination of dredging. Sand mining targets sites that can provide dredged materials that contain few fines, so the generation of turbidity is intentionally minimized by initial selections of clean sand resources.

The spatial scale of enhanced turbidity extends some distance beyond the dredging footprint, dictated by fine particle transport by ocean currents and fall velocities that increase as sediment size increases. Biological impacts of this enhanced turbidity in the OCS are expected to be minor. Nevertheless, if turbidity plumes extend to any nearby coral reef habitat, corals can exhibit suppressed growth under more turbid conditions (Dodge et al. 1974; Anthony and Fabricius 2000) and mortality from a combination of turbidity and siltation (Dodge and Vaisnys 1977). Such coral injury could be major if it affected either staghorn or elkhorn coral because they are federally listed species, or if it affected one of the seven additional corals proposed for listing as threatened or endangered. Such major effects can be avoided by separating borrow areas and rehandling areas at sufficient distances from any coral reef, and by taking bottom current speeds and directions and perhaps also silt content of sediments in borrow areas into consideration to model expected maximum transport at areas proposed for dredging. Because visually orienting fishes can experience reduced capacity to find and consume their prey in turbid waters (Manning et al. 2013), turbidity associated with sand mining could lead to sustaining higher population abundances of some benthic invertebrate prey because of suppressed predation. Such an indirect effect on the benthos is probably of trivial importance in relation to sand mining in the OCS.

4.2.2.3 Increased sedimentation and deposition of fines

The most severe injury to benthic habitat that can indirectly develop from OCS dredging activity is epifaunal (including stony corals) mortality from sedimentation upon hard-bottom habitat. In addition, sediment deposition can inhibit colonization by larvae of numerous invertebrate species that need hard surfaces to settle and develop (Thorson 1966; Rogers 1990). In one study, sedimentation by fine sand and silt affected 24.7 acres of reef, with abnormal sediment accumulations of 6-8 cm on the reef north of the borrow area and 3-5 cm on the reef south of a borrow area in Florida State waters; elevated sedimentation levels were observed 5 weeks into an 8-week dredging project at least 335 m from the borrow area and extending further to 400 m in some places (Blair et al. 1990a). Elevated sedimentation levels (up to 2.9 cm) were also documented 245-365 m from the borrow area of a Miami-Dade County project in 1997 (USFWS 2002, cited in USACE Jacksonville District 2003a). Moderate to heavy sedimentation was observed on hard and soft corals, sponges, and hard-bottom habitat near borrow areas for another Miami-Dade County project, South Government Cut (MDCDERM 2010, 2012). Sedimentation was measured at stations 120-670 m from the borrow area (PERA CS-14 2012). Most stations showed no or minor increases in sediment thickness on the seafloor and on coral communities; however, the two closest areas (140 m south and 180 m north) showed higher sedimentation (up to increase an ~30 mm above pre-dredging amounts: PERA CS-14 2012). Where sedimentation was greatest, coral mortalities were documented as well as bleaching (Dial Cordy and Associates Inc. 2012). Most stations returned to pre-dredging conditions within two weeks after dredging.

Sedimentation damage to hard-bottom reef habitat was documented by Courtenay et al. (1972, 1974) from monitoring a beach nourishment project in south Florida off Hallandale. Here, habitat damage was reported across distances of 130-220 m from the dredged site, with sedimentation covering a reef area of 2.5 km^2, which was largely attributed to resuspension caused during rehandling of the fill material. Macroalgae and epibiotic bivalves were killed and hard corals exhibited substantial damage from sediment cover, while soft corals, which are vertically emergent and do not offer a horizontal surface that can be readily covered by sediments, remained unaffected. Resurveying of the injured reefs at 8-12 m depth seven years post-dredging revealed apparent recovery of all the coral damage (Courtenay et al. 1980). Eggs of benthic invertebrates such as some polychaetes and gastropods that are placed on and develop on the seafloor (La Salle et al. 1991) and demersal fish eggs (Miller et al. 2002) are the most sensitive life stages to mortality from siltation because they require continuous, high oxygen concentrations. Depending on species, egg deposition and development can occur on hard-bottom or sedimentary habitat.

The substrate variable of most concern that can be regularly modified by sand mining is sedimentation onto hard-bottom habitat. The depth and extent of sedimentation onto hard-bottom habitat are measures of intensity and breadth of impacts. In addition to causing mortality of stony corals through sedimentation, locating borrow areas too close to hard-bottom reefs has also been shown to result in direct damage to corals and other epibiota from unintended contact with and scraping by dredging equipment in a beach nourishment project off Sunny Isles in Dade County, Florida (Blair et al. 1990b). The most obvious mitigation action to limit damage to benthic resources, communities, and hard-bottom habitat involves establishing appropriate buffer distances to separate dredging or rehandling areas from existing hard-bottom habitat. In the absence of site-specific knowledge about direction and speed of sediment transport in bottom current flows and reliable modeling of the spatial pattern of expected sedimentation in two dimensions, a single minimum buffer distance of around 400 m might be appropriate, based on the worst-case scenario documented by Blair et al. (1990a). Because impacts on staghorn and elkhorn corals could represent major impacts, buffer distances around colonies of either of these species may need to be even greater to serve as adequate mitigation. Blair et al. (1990a) suggest that buffer distances be scaled to silt and fine sand content of the sediments being dredged, with minima of 100 m with silt content <3%, 500 m with silt content of 5-9%, and 1,000 m (Griffin 1974) when silt content is >10%. Real-time monitoring of currents and sediment transport or advance modeling of expected sedimentation may be needed as mitigation for potential impacts to these two ESA-listed acroporid corals. The need for wide buffers to protect the two acroporid corals would apply only to southeast Florida and the Florida Keys, the only regions in the continental U.S. where these two species presently occur. If additional species of stony corals are listed, then the geographic area requiring mitigation may expand. Mitigation could also include water-quality (turbidity) monitoring and cessation of dredging when thresholds are exceeded outside a permissible mixing zone that does not include resident acroporid or other stony corals or hard-bottom habitat more broadly.

Impacts of deposition of resuspended sediments resulting from dredging, deposition at rehandling areas, and redredging sediments stored at the rehandling areas can also occur on soft-bottom habitats within and well beyond the dredged location. The increase in suspended sediment concentration, typically modestly enriched organically, and its subsequent deposition can have either positive or negative impacts on sedimentary benthic invertebrates. Sedimentation can bury and suffocate benthic organisms, especially filter feeders, although some buried organisms are able to migrate vertically to the new sediment surface (Peterson 1985; Maurer et al. 1986; Miller et al. 2002). Reductions in infaunal abundance and biodiversity have been reported well beyond the dredged footprint (Johnson and Nelson 1985), with decreases in short-term abundance of macrobenthos ranging from 34-70% up to 100 m from the dredging site (McCauley et al. 1977). In general, such reductions in abundance decrease with distance from dredging sites (Johnson and Nelson 1985; Desprez et al. 2010). In contrast to these smothering impacts, sedimentation of organic matter released by dredging and deposited on sedimentary bottom has been credited with increasing benthic faunal abundance and diversity down current from dredging sites (Poiner and Kennedy 1984; Newell et al. 2004b), with the level of enhancement decreasing with distance from the site. Increases in macrobenthic abundance and biomass in response to releases and deposition of organically enriched sediments have been documented during wind events (Walker and O'Donnell 1981), and from simulated storm events in a mesocosm (Oviatt et al. 1982), matching expectations from a review of the literature (CSA et al. 2010).

Temporary deposition of sand in rehandling areas is expected to cause virtually complete mortality of all the benthic invertebrates within the deposition footprint except for some more mobile species at the margins where burial depths are so modest that some mobile polychaetes, gastropods, and other organisms may survive. This expectation arises from observed mass mortality of macrobenthos after fine sediment deposition in Long Island Sound by McCall (1977) and Rhoads et al. (1978), and similar mass mortality of benthos after sand deposition on ocean beaches during beach nourishment (e.g., Peterson and Bishop 2005, Bergquist and Crowe 2009). Although fine sediments will have been partially winnowed out of these sands during dredging and shipboard handling, small quantities of silts and clays would be expected to remain to increase local turbidity (only increasing by less than 3 NTUs in the Bodge 2002 example near Cape Caneveral, Florida) and induce correspondingly minor siltation extending some distance away from the footprint of the rehandling area. Furthermore, because the rehandling area is most likely located in shallower water than the borrow area, some minor erosion, transport, and deposition of additional fines would be expected during the period between deposition and re-collection of these deposits for transport to the intended deployment location, typically an eroding ocean beach. The process of redredging of these sediments would similarly result in minor enhancements of local turbidity (Bodge 2002) and their transport and ultimate deposition in and outside the footprint of the rehandling area. At each of these stages during which turbidity is generated and siltation is induced, the effects should decrease with successive dredging and rehandling, and only to a small degree conform with the impacts of turbidity enhancement and siltation processes as described for the initial dredging in the borrow area. Cearly, the lower the fraction of fine sediments in the original borrow area, the less the loss of fines at each succeeding stage in the rehandling process.

4.2.2.4 Water quality

Water-quality parameters potentially modified by sand dredging that may influence the benthos include turbidity and concentrations of ammonia, oxygen, and hydrogen sulfide. We discussed the role of turbidity in Section 4.2.2.2. Over the short term (weeks), compounds such as ammonia and hydrogen sulfide can become mobilized into the water column by the disturbance of dredging and may represent chemical cues of habitat unsuitability, causing larval benthic invertebrates to avoid recently disturbed sites (Engstrom and Marinelli 2005). Low levels of oxygen and trace levels of chemical contaminants, including hydrogen sulfide, may also inhibit faunal recruitment after dredging, until sediment reworking and hydrodynamic forces restore healthy, oxygenated water-column chemistry (CSA et al. 2010).

Although excavation of deep pits has occurred in few OCS projects, sand mining that does create deep pits in environments with limited hydrodynamic energy can produce depositional basins that are very slow to refill and in which sand content is low, silt/clay content high, and organic content high (Culter and Mahadevan 1982; Van Dolah et al. 1994a; 1998). These conditions can lead to high biological and chemical oxygen demand with resulting low oxygen concentrations and often release of hydrogen sulfide, both of which suppress settlement and survival of colonizing benthic invertebrates (e.g., Saloman et al. 1982; Diaz and Rosenberg 2008). Without adequate rates of sediment transport to fill such depressions, recovery of initial bathymetry will be slow, which can extend the water quality problem and suppress benthic recovery for as much as a decade or longer (Van Dolah et al. 1998). Some suggest that some of the deeper dredged pits in areas of low energy may last indefinitely without benthic resource recolonization (Taylor Biological Company 1978, cited in Pullen and Naqvi 1983). Saloman (1974) describes the conditions inside a borrow area off Treasure Island, Florida, revealing thick deposits of organic-rich mud, low oxygen levels, and evolution of hydrogen sulfide three years after dredging. Taylor Biological Company (1978, cited in Pullen and Naqvi 1983) conducted further sampling of all four borrow pits used for the Treasure Island nourishment project and demonstrated that they exhibited very slow benthic infaunal recovery when assessed four years after dredging. In contrast, in the presence of strong longshore currents off Hillsboro Beach on Florida's east coast, similar water-quality problems and organic mud conditions did not develop even though filling of the borrow area depressions was incomplete after five years (Turbeville and Marsh 1982). Figure 4.2 shows a conceptual model of the possible scenarios of recovery of the benthic resources, habitat, and community composition following a sand extraction design that results in pit formation.

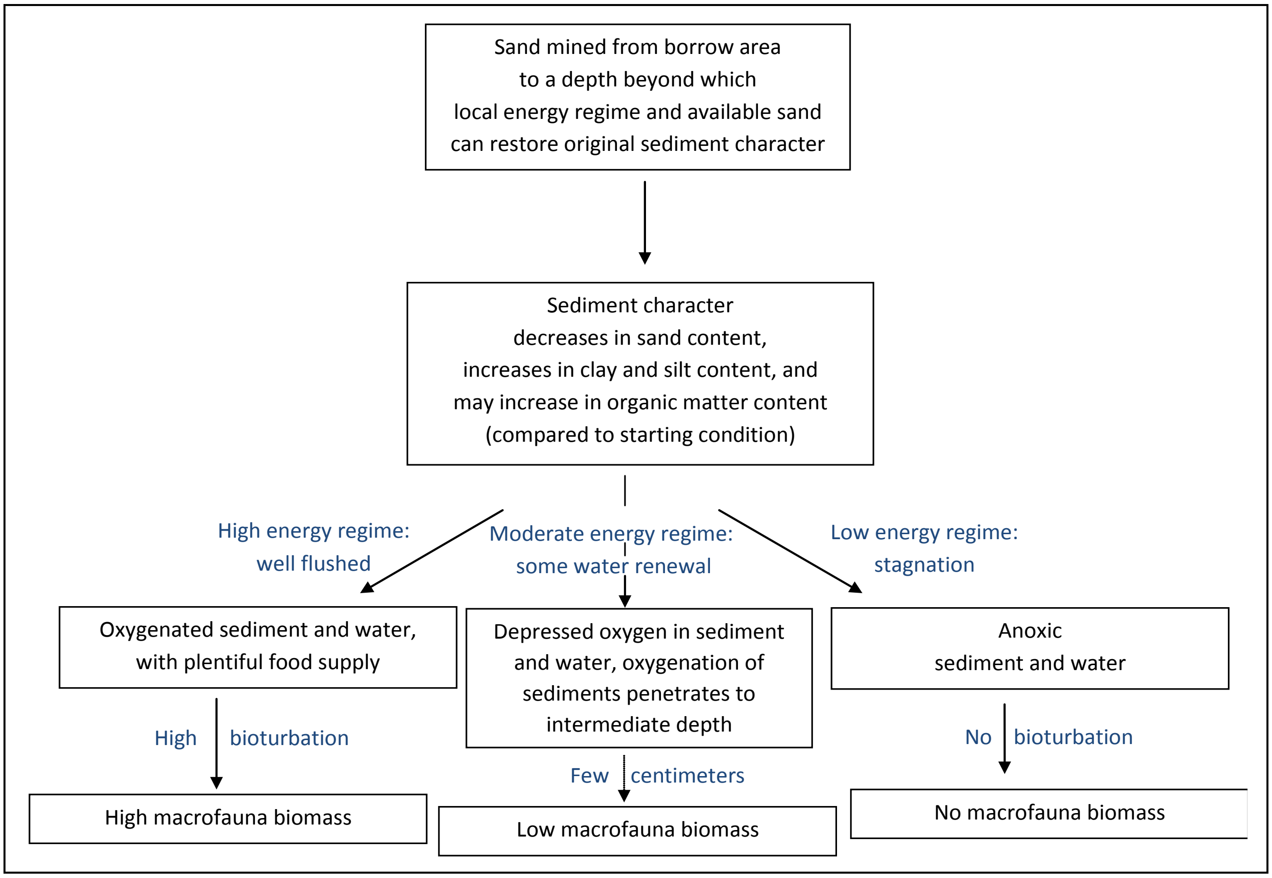


Figure 4.2 Simplified diagram of scenarios following mining of sand, pit formation, and modifications to the marine environment to water renewal. Modified from Pearson and Rosenberg (1978).

Avoiding dredging deep pits, selecting borrow areas that experience higher bottom boundary layer energy, and using borrow areas with good supplies of sand can reduce the risk of developing these water-quality problems (Murawski 1969; Taylor Biological Company 1978, cited in Pullen and Naqvi 1983; Saloman et al. 1982). These three constraints can thus serve as mitigation measures. Furthermore, compiling and mapping locations of borrow pits that did and did not develop water quality problems during recovery along with data layers of sedimentology, bathymetry, bottom boundary layer hydrodynamic energy, and sources of available sediments by type (sand vs. river-borne fines) may provide empirical guidance to locating borrow areas where recovery will not be slowed by development of water quality degradation. The coast of South Carolina would seem to be the best-sampled place in which this mitigation could be practiced because of the impressively large numbers of historic sand dredging projects for which data were gathered on initial conditions and short-term impacts, setting the stage for more insightful, longer-term research (see Table 4.1). However, most of these borrow areas were in State waters, so the issue of how well the results transfer to deeper OCS grounds would need to be addressed.

Any operations involving boats and equipment powered by fuel oil runs a risk of spills. Marine organisms, such as seabirds, that occupy and use the sea surface can suffer exposure to and resulting illness or even death from exposures to surface oil. Among benthic invertebrates, some species with surface eggs or larvae would run a small risk of mortality from oil spills. Transport of surface oil shoreward could result in coating intertidal habitats with oil, but the volatility of most fuel oils is sufficient that thick, suffocating coats of oil would not be expected. The most likely negative effects would be reductions in growth of suspension-feeders, which process large volumes of water during feeding. As a result, modest reductions in growth might be expected in oysters and clams in estuarine harbors if the spill is nearby. A small spill on the OCS would not be likely to be transported to shore in sufficient concentration to induce detectable harm.

Mitigation for such oil spills might be accomplished by routine inspections and maintenance of the vessels and dredges to be used in sand extraction. Immediate reporting of any spill to the U.S. Coast Guard also represents mitigation for more serious damage.

4.2.2.5 Entrainment near the seafloor

As a matter of definition, we consider the uptake of benthic invertebrates in and on the seafloor sediments as extraction along with the sediments during mining, and not as entrainment, and thus cover this process in Section 4.2.2.1. Uptake of more mobile, benthic megafaunal organisms may alternatively be defined as entrainment, meaning that uptake by the dredge of organisms such as sea stars, sea cucumbers, and octopi are defined as “entrained” by equipment during dredging operations, which can lead to their death (Johnson 1982; Saloman 1974; Culter and Mahadevan 1982; Reine and Clarke 1998; Nightingale and Simenstad 2001). There is also a vulnerable group of what are considered bentho-pelagic invertebrates, dominated by mysids, a crustacean that spends daylight hours buried in sediments and then rises up into the water column at night to feed like copepods on suspended particles in the water column. Mysids and invertebrate larvae can account for a substantial biomass of particle feeders on the continental shelf and, because they can be entrained while pelagic by suction from the dredge, we categorize the dominant mechanism of mysid and invertebrate larvae losses as entrainment (Nightingale and Simenstad 2001). Mortality of mysids is a likely effect of sand dredging on these part-time benthic invertebrates. Mysids and invertebrate larvae would seem unlikely to survive the process of entrainment and subsequent incorporation into the dredged sand mass for transport, possible deposition into a rehandling area and remobilization, or final application to the shore. Mysids represent a locally significant food source for larval and juvenile fishes, including early life stages of fishes of commercial or recreational value and of forage fishes (Miller et al. 1976; Simenstad et al. 1979; Jumars 2007). Because of the nightly migration upwards some substantial distance into the water column and then back downwards into the sediments near dawn, mysids would become mixed over large areas by directional currents at the dredge site. This implies that their depletion at the dredge site would not be expected to be total, but that the reduction in mysid numbers would affect areas some large distances away from the footprint of the dredging, as dictated by ocean currents. Recovery of mysids would probably be complete within a few months because of their short generation times, and thus risks to mysids would not rise even to a moderate level of concern.

4.2.2.6 Sound

A thorough review of primary literature and agency reports, including especially the proceedings of the BOEM workshop on Effects of Noise on Fish, Fisheries, and Invertebrates (Normandeau Associates Inc. 2012) revealed evidence of sound production and sound detection in some invertebrates such as snapping shrimp, cephalopods, and some bivalves; however, the role of sound in the ecology of marine invertebrates is largely unknown.

4.2.2.7 Vessel operations and interactions (including laying of pipelines)

There is no reasonable mechanism by which vessel movements could directly affect benthic invertebrates during sand mining in the OCS because of the typically substantial water depths. However, there could be some risk of damage to hard-bottom habitats in shallow, nearshore areas. For example, Blair et al. (1990a) monitored effects of dredging a borrow area in State waters about 2.4 km off the beach at 17-20 m depths to extract sand for nourishing a Bal Harbor beach in south Florida. Sampling afterwards revealed evidence of impacts on hard corals both from mechanical damage by inadvertent encounters with the dredge and from sedimentation. Over 100 m^2 of coral reef were affected by mechanical damage, within which 85 m^2 were denuded of benthic invertebrates. Maintaining sufficient separation distance between the borrow or rehandling areas and any hard-bottom habitat can serve to mitigate against hard-bottom habitat degradation caused by unintended encounters with the dredge.

Transportation of stored sand from the rehandling area to the final deployment site(s) may be accomplished by dredge or barge, but use of a pipeline could become more common as a means of optimizing the dredge fleet (Bodge 2002). A pipeline could impact benthos growing on hard bottom by crushing and abrasion along the pipeline route and extending outwards from its footprint if the pipeline is not firmly anchored in place to remain fixed during storms. For example, abrasion and breakage of hard corals and soft corals occurred in the South Government Cut project in 2012 (Dial Cordy and Associates Inc. 2012), resulting in impacts to 10.1 m^2 of hard-bottom reef habitat, affecting 11 hard coral colonies, including *Acropora cervicornis*. Damages were caused by contact from the tires and steel collars used to hold the pipeline in place and from where the pipeline itself contacted a topographic high stretch of shore-parallel reef. Further mechanical damage to benthic organisms would be expected to occur at locations where boating activity and anchoring occur to assemble and disassemble sections of pipeline, where booster pumps are installed, and where riser pipes are attached. Such impacts would be especially serious if they affected staghorn or elkhorn corals, as happened in the South Government Cut Miami-Dade project. Mitigation measures to protect not only corals but all other hard-bottom benthic invertebrates involve careful mapping of the potential alternative pipeline routes, leading to selection of a route that does not cross or minimizes contact with hard-bottom habitat and does not approach any such hard-bottom habitat closely enough to run a risk of substantial contact with the pipeline during installation, demobilization, or storms. The placement of a pipeline over sedimentary bottom would be expected to cause some direct mortality of sedimentary benthic organisms by crushing, but such effects would be relatively minor and recovery of the benthos would be rapid after pipeline removal (MESL 2007).

4.2.2.8 Exposed UXO, shipwrecks, and other hard structures exposed by dredging

Newly exposed structures provide habitat for the recruitment of benthic epibiotic organisms; thus exposing such structures is likely to enhance hard-bottom benthic communities, perhaps only temporarily if reburial occurs (see Section 4.2.2.1). Unless ordnance is detonated during dredging operations and directly kills epifauna or eliminates habitat, newly exposed UXOs will only harm existing benthic communities to the degree to which these exposures of non-biodegradable objects displaces benthic soft-sediment habitat.

4.2.3 Summary of Known Impacts on Benthic Resources, Communities, and Habitats due to OCS Dredging and Data Gaps

The degree of impact of OCS dredging on benthic resources, communities, and habitats is typically determined by measuring the magnitude of direct and indirect effects on and rate of recovery of four fundamental metrics of (typically) macrobenthos: abundances, biomass, species richness (sometimes also more complex measures of diversity), and community composition (Newell et al. 1998; Hill et al. 2011). The mechanism that most strongly affects the magnitude of direct dredging impacts on benthic resources, communities, and habitats is the extraction of sediments and, inadvertently, most benthic animals living in those sediments (Table 4.2).

The second most important process that contributes to determining the magnitude of benthic impacts is the indirect effect of deposition of fine sediments, accompanied by modest organic nutrient additions (e.g., Jutte et al. 2002), during and for a short time after dredging through the processes of local settlement of organically enriched silts and clays resuspended during bottom-disturbing dredging and settlement of fine materials washed overboard during handling of dredged materials at the ship (e.g., Kenny and Rees 1994, 1996) (Table 4.3A). Such siltation can result in some smothering of benthic invertebrates, which can be most serious when the siltation occurs on hard-bottom habitat, especially if ESA-listed staghorn and elkhorn corals occur on the hard bottom. Temporary storage/rehandling of dredge materials close to the placement beach is a different type of (intentional) sedimentation: impacts are listed in Table 4.3B.

Table 4.2

Impacting mechanism for OCS dredging on benthic resources: *Alteration of benthic habitat at the borrow area.*

Impact Pathway	*Potential Effects on Benthic Resources, Community, and Habitat*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☒	Direct Effects Immediate removal and mortality of 45-88% (U.S.) and 40-95% (U.K.) of benthic organisms[1] Removal of sand, creating topographic depressions Indirect Effects Creation of depressions that fill initially with finer sediments, can have poor water quality, and result in different benthic communities[2,3] Cumulative Effects Potentially long-term (perhaps >10 yr but unsubstantiated) change of benthic community from sand-associated to an impoverished, mud-associated community until surface depression is completely filled[4,5], when water quality is restored, and surface sediments return to original[3,6]	Impacts confined to the footprint of dredging plus area of slumping into the depressions along edges.	Recovery of abundance and biomass usually occurs within 0.5-2.5 years [2,7,8,9], but recovery of community structure can take longer[11,12]. However, if deep pits are formed where low sand availability and low energy produce low rates of sand transport, recovery of community composition may take >10 yr[4,5,6,7].	Recurs with each dredging event.	In most cases, the dredged area represents a small % of available habitat and recovery rates are relatively rapid[14], so impacts are usually considered minor; where pits persist and fill with fine-grained sediments, impacts could be moderate[3,4,5,6].
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Numerous short-term (1-2 years) studies on seabed impacts and on the impacts on biological invertebrate assemblages[7,8,9,10,11], but insufficient long-term monitoring to infer temporal duration of benthic community change.	Restrict dredging depths to <1.5 m where infilling by fine sediments is a risk; use borrow areas away from riverine sources of fine sediment and near sources of sand to avoid slow infilling by fine, organic-rich sediments that promote a different macrobenthic community [3,4,5,6]. Dredge depocenters or the leading edges of sand shoals to sustain shoal morphological integrity, crest elevation, and grain size; use seasonal dredging window to coincide with low benthic biomass[15] and promote return of taxa more valuable as fish prey by exploiting seasonality of settlement[16]; perhaps, dredge in strips to speed recovery by leaving undisturbed areas to speed recovery by slumping and short-range immigration[12,13].		Shallow mining depths and sufficient separation from inlet plumes delivering fine suspended sediments appear in SC studies to be effective in avoiding mud infilling and its consequences[3,13]. Inadequate study of whether dredging in strips speeds up sediment refilling and benthic recovery.	

[1]numerous: see section 4.2.2.1, [2]Cooper et al. 2007, [3]Bergquist and Crowe 2009, [4]Palmer et al. 2008, [5]Turbeville and Marsh 1982, [6]Van Dolah et al. 1998[7]Rhoads et al. 1978, [8]Pearson and Rosenberg 1978, [9]Newell et al. 1998, [10]Bolam and Rees 2003, [11]Hill et al. 2011, [12]Whitlatch et al. 1998, [13]Jutte et al. 2001a, [14]Naqvi and Pullen 1982, [15]Posey and Alphin 2002, [16]Diaz et al. 2004

Table 4.3A

Impacting mechanism for OCS dredging on benthic resources: *Increased sedimentation and deposition of fines.*

Impact Pathway	*Potential Effects on Benthic Resources, Community and Habitat*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☒	Direct Effects Sedimentation by fine, organic-enriched sediments resuspended during dredging[1,2,3,4,5] and overflows[6] or during rehandling[7], suffocating hard-bottom benthos and sensitive sedimentary benthos[1,2,3]. Indirect Effects Some increased settlement of mud-loving benthos and inhibition of larval settlement on hard-bottom substrata covered with silt[8,9]. Cumulative Effects Deeper pits and pits in sand sheets with quiescent hydrodynamics and little sediment transport accumulate fine organically enriched sediments over multiple years[10,11,12,13], often developing hypoxia/anoxia occasionally releasing ammonia and toxic hydrogen sulfide, suppressing benthic community, and leading to mud-associated, not sand-bottom benthos[11,12,13,14].	Across the entire footprint of dredging plus outside the footprint, and over the footprint of the rehandling area and beyond it[1,2,3,5,14], dependent on current flow speed and direction, and susceptibility to resuspension. Finer fractions of suspended sediments may deposit further away from the source[1,,2,3,5,14].	Coarse particles settle rapidly, while silts and clays remain suspended for hours to days[15]. The temporal duration of impacts depends on resource susceptibility, with eggs likely more sensitive to impacts[16,17]. Coral colonies can recover within months unless the entire colony is covered and suffocated. Absence of corals and other epifauna will persist as long as the substrate is covered by sediments.	Recurs with each dredging episode or if sediments are resuspended[6,18,19].	Causes suffocation of benthic epifauna on hard bottoms, including stony corals, while impacts on soft-bottom benthic species are limited to those that are unable to move up through the deposited layer[17,20,21]. Greatest impacts in soft sediments may be suffocation of bottom-attached gastropod, polychaete and fish eggs[17].
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Many demonstrations of organically enriched siltation on surface of dredged area and beyond as well of documentation of mud deposition into deeper dredged pits[10,11,12,13], but insufficient assessment of how to measure hydrodynamic energetics adequately to predict where sand transport will be great enough to fill dredged depressions rapidly and overwhelm siltation[22].	Avoid dredging deep pits; develop methods to predict which sand sheets experience high rates of coarse sediment bedload transport, and target those for dredging and avoid locations experiencing high transport of suspended fines from riverine sources to prevent slow filling of pits and persistence of fine sediment deposition. Establish appropriate buffers [3,4,5,22] from hard-bottom habitat and perhaps even greater separation from staghorn or elkhorn coral colonies or use models and field observations to predict extent of plumes.		Uncertain until completion of further compelling studies or necessary field tests. Additionally, development of transport models to predict site-specific patterns of sedimentation based on local measures of currents is needed and % fines in the borrow area[22], especially to protect acroporid corals, which are federally listed species, and seven other stony corals proposed for listing.	

[1,2]Courtenay et al. 1974; Courtenay et al. 1972, [3]Blair et al. 1990a, [4,5]MDCDERM 2010, 2012, [6]Hill et al. 2011, [7]Bodge 2002, [8]Thorson 1966, [9]Rogers 1990, [10]Culter and Mahadevan 1982, [11]Van Dolah et al. 1992, [12]Bergquist and Crowe 2009, [13]Palmer et al. 2008, [14]Johnson and Nelson 1985, [15]Gibbs et al. 1971, [16]La Salle et al. 1991, [17]Miller et al. 2002, [18]Rhoads et al. 1978, [19]Wilber and Clarke 2001, [20]Peterson 1985, [21]Maurer et al. 1986, [22]Blair et al. 1990a.

Table 4.3B

Impacting mechanism for OCS dredging on benthic resources: *Sediment rehandling*.

Impact Pathway	*Potential Effects on Benthic Resources, Community and Habitat*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☒	Direct Effects Suffocation and mass mortality of benthos by burial under deposited sand[1,2,3] Indirect Effects Increased turbidity from resuspension of fine sediments during deposition and subsequent rehandling, which reduces growth and can kill some suspension feeders on hard bottoms and in sedimentary habitat[4,5,6]. Siltation that arises from settlement of these fine particles induces mortality from smothering on hard bottoms and in sedimentary bottom, as well as transforming hard bottoms into unsuitable habitat for recruitment of epibiotic species requiring hard substrata[7,8,9,10] Cumulative Effects Impacts of losses of foraging use accumulate and sum to larger areas of habitat as first the borrow area habitat is impacted along with some surrounding margin, and then the storage site becomes degraded twice–once during deposition of sand and induction of turbidity[11] and then again when the sand resource is transported to its intended beach location[12]	Burial is restricted to the footprint of the rehandling area, while impacts of turbidity and siltation are most intense within the rehandling area boundaries but extend outward as a function of sediment sizes and composition, direction and speed of currents[12], and other factors.	Suffocation of existing benthos is a permanent state, although the surface of the stored sand will become colonized relatively rapidly by sand-loving benthos initiating succession. Turbidity arising from sand deposition and from its remobilization is relatively short-lasting on scales of days[11,13]. Siltation is longer-lasting but variable in duration dependent upon bottom shear stress that erodes these fine sediments.	Suffocation occurs once each time sand is deposited at a new location in the rehandling area, whereas induction of turbidity and its deposition as siltation recurs in a cycle driven by sequences of energetic wave events that drive another round or resuspension and deposition of fines until they become transported and more permanently settled into lower-energy depositional areas.	The mortality of benthos and consequent reductions in its value as foraging grounds for demersal predators are not permanent and the functionality of the sedimentary habitat area is restored by natural processes after the stored sand is removed[4]. The siltation onto nearby hard-bottom habitat, rendering it unsuitable for epibiotic species represents habitat degradation that may persist for relatively long periods of time,[6,7,9] reducing its value to demersal fishes, crabs, shrimps, and sea turtles.

Availability of Empirical Information	*Mitigation Measures*	*Mitigation Effectiveness*
Some studies document siltation of hard-bottom habitat and its impacts over time induced by rehandling, but assessment of such siltation effects on sedimentary habitat is unavailable. Impacts of temporary transformation by siltation on hard-bottom to sedimentary habitat on foraging by demersal predators and sheltering of sea turtles and other mobile vertebrates require more documentation.	Locate rehandling areas far enough from hard-bottom habitat that transport by currents of fine sediments is unlikely to travel far enough to settle as siltation on hard bottoms. Where feasible, re-use rehandling areas by sequentially storing sediments and then transporting them to their intended fill sites, so as to minimize total rehandling area and total benthos suffocated.	With characterization of the hydrodynamic environment at prospective rehandling areas and % fines in the prospective borrow area[8], probability of siltation on nearby hard bottoms could be modeled so as to make more rigorous siltation risk computations and establish effective buffer distance requirements.

[1]Oliver et al. 1977, [2]McCall 1977, [3]Bolam et al. 2006, [4]Rhoads and Young 1970, [5]Wilber and Clarke 2001, [6]Courtenay et al. 1974, [7]Courtenay et al. 1972, [8]Blair et al. 1990a, [9]La Salle et al. 1991, [10] Tillin et al. 2011, [11]see Table 4.2.3.2 A above, [12]Bodge 2002; [13]Gibbs et al. 1971.

Increased turbidity represents the third most important mechanism of impact on the benthos (Table 4.4), inducing mortality, especially of filter-feeders that are sensitive to turbidity (Tillin et al. 2011), and modifying the sedimentology of the bottom, which inhibits recovery of most sand-associated species that dominated the borrow area before sand mining. Like most indirect effects, this influence of the fining of surface sediments along with associated nutrient addition occurs with a time lag as these finer sediments, which include organic materials, prove more suitable for benthos with mud affinity than for sand-loving benthos during at least the initial phases of recolonization (Boyd et al. 2005).

Turbidity alone can harm and even kill stony corals as well as other epifaunal invertebrates on hard bottoms (Dodge et al. 1974; Dodge and Vaisnys 1977). Although each coastal state and the federal governments specify numerical turbidity standards in NTUs, no studies exist to test how impacts of turbidity itself change over a range of numerical levels and under conditions of temporally varying turbidity levels, as typifies the OCS waters where changing hydrodynamic energy levels can resuspend surface sediments. Water-quality degradation plays a role in inhibiting recovery of the benthos in deep dredge pits when they occur under conditions of low hydrodynamic energy at the seafloor and/or inadequate sand supplies so that coarse sediments are not rapidly transported by bedload transport to fill the dredge pits (Table 4.5). Under these conditions, low oxygen and high hydrogen sulfide concentrations develop in the sediments and overlying waters, suppressing any recolonization of benthos.

The multi-year and perhaps even decadal (Palmer et al. 2008 estimate of 7-8 years to benthos recovery in a pit off Louisiana) suppression of recovery of macrobenthic abundances, biomass, species richness (diversity), and community composition in the deeper dredge pits where water quality remains poor does reflect an impact of benthic resources, communities, and habitats. Nevertheless, this problem falls well short of becoming a major impact. Even in cases where deep pits are created by dredging and where bottom transport of coarser sediments is too low to refill them quickly with sand and they develop muddy substrata (Palmer et al. 2008), with or without local hypoxia and enhancement of hydrogen sulfide, such pits will fill in after varying periods of time. Ultimately they are expected to develop a surface sedimentology resembling the surrounding seafloor, promoting return of original benthos and its ecosystem services. In addition, limitations on dredging depths and developing protocols to limit dredging to locations with sufficiently high energy and available sand resources to promote rapid infilling by bottom-transported sand could serve as viable mitigation measures.

The uptake of benthic invertebrates in and on the seafloor sediments is considered as extraction along with the sediments during mining, and not as entrainment. Nevertheless, benthopelagic invertebrates, typified and dominated by mysids, as well as invertebrate larvae, would be affected by entrainment when they occur in near-bottom waters; they would also be subject to direct uptake by the dredge when buried (Nightingale and Simenstad 2001). We do not attempt to partition losses of mysids, other benthopelagic invertebrates, and invertebrate larvae between these two mechanisms and instead simply list mysids and invertebrate larvae under the entrainment category along with large mobile epibenthos, such as sea stars, other echinoderms, and octopi (Table 4.6). Because of short generation times of only a few months, recovery of mysid populations would be expected to be rapid.

Table 4.4

Impacting mechanism for OCS dredging on benthic resources: *Increased turbidity.*

Impact Pathway	*Potential Effects on Benthic Resources, Community and Habitat*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☒	Direct Effects Reduced growth and elevated mortality of corals, which are sensitive[1,2], along with growth reductions and some increased mortality of suspension-feeding benthos in sedimentary habitats within and beyond the dredging footprint[3,4] For dredging depths within the photic zone, decreased production of benthic macroalgae and microphytobenthos[5]. Indirect Effects Reduction in forage base for demersal predators that consume suspension-feeding macrobenthos. Reduction in emergent phytal habitat for crustaceans and other commensals and in forage base for benthic herbivores. Reduced ability of visually orienting demersal fishes to capture benthic prey[6]. Cumulative Effects Turbidity-induced reductions in feeding efficiency of visually oriented consumers could combine with lower abundances of benthic prey from their extraction to induce possible fitness declines in visually orienting demersal fishes[6].	Impacts extend beyond the footprint of the dredging and rehandling areas, with overall spatial exposure of benthos to elevated turbidity dictated by direction and speed of currents. Plumes typically extend to a few hundred meters from the source[7,8,9,10,11]., with the maximum distance measured in the thousands of meters[15].	The duration of a single plume ranges from hours to days[13], but plumes can be continuously generated during dredging, which may last for months, and plumes can be created for months after dredging ceases as waves and currents resuspend fine materials deposited on the seabed.	Recurs with each dredging event. In addition, resuspension of deposited surface fines can recur for an indefinite period as waves and currents erode and inject fine sediments into the water column over the dredge site[14].	Impacts are constrained in spatial scope and in temporal duration, so as to be limited in importance and will naturally abate without management intervention[1,10].
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Sensitivity of corals to turbidity is reasonably well established[1,2]. Turbidity effects on infaunal suspension feeders of sedimentary habitats are not well documented.	In coral habitat, monitoring of suspended sediment concentrations (NTU) with cessation of dredging if above threshold.		Likely effective, but monitoring results not readily available.	

[1]Dodge et al. 1974, [2]Dodge and Vaisnys 1977, [3]Rhoads and Young 1970, [4]Wilber and Clarke 2001, [5]Grippo et al. 2009, [6]Manning et al. 2013[7]Courtenay et al. 1974; [8]Courtenay et al. 1972, [9]Blair et al. 1990a, [10,11]Miami-Dade County Department of Environmental Resources Management (MDCDERM) 2010, 2012, [12]Anthony and Fabricius 2000, [13]Gibbs et al. 1971, [14]Peterson and Bishop 2005, [15]Hammer et al. 1993.

Table 4.5
Impacting mechanism for OCS dredging on benthic resources: *Water quality*.

Impact Pathway	*Potential Effects on Benthic Resources, Community and Habitat*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☐	Direct Effects Increases in hypoxia/anoxia and/or hydrogen sulfide can kill existing benthic invertebrates[1,2] and inhibit larval settlement[3,4,5]. Indirect Effects Elevated levels of ammonia and hydrogen sulfide or hypoxia deter mobile organisms from entering affected areas[3,4]. Small oil spills would kill some larvae at the surface[6,7]. Cumulative Effects Suppressed recovery of benthic prey could combine with fishing extractions to reduce production of commercially valuable demersal fishes, crabs, and shrimps[2].	Mostly confined to deeper dredge pits and/or where hydrodynamic energy is low or sand supply low or suspended fine particle delivery is high so pits fail to refill rapidly with sand, fill with fines, and insufficient oxygen is circulated into the pits[13]. Oil spill impacts for small spills from the dredge would only affect surface organisms[8].	The duration of reduced water quality is dependent on the rate at which the pit becomes filled largely by siltation rather than by coarser sediments, letting poor water quality persist perhaps for 10 yr or more[4,9,13]. Toxic compounds in spilled gas and diesel would be volatilized into the atmosphere in days to weeks and be widely dispersed and diluted.	Most likely in deeper pits where overlying muddy sediments had to be removed and infilling will take several years[10,11,13], but also occurs in pits excavated in sand where energy regime and sand supplies are low. Small oil spills would occur only rarely.	The severity of reduced water quality is dependent on pit depth, hydrodynamic energy to transport sand and sand supply in the vicinity of the borrow area, and character of sediments around the borrow area[4,12,13], and disturbance frequency that would allow re-oxygenation of borrow area sediments[9]. Impact is inhibition of benthic colonization and survival for years, perhaps more than a decade. The impacts of small oil spills from the dredge would be negligible.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Multiple studies demonstrate the generation of poor water quality in deeper dredge pits, but documentation of its duration is incomplete[12,13]. Inferences on benthic invertebrate and fish impacts now come from studies of "dead zones" not dredge pits. Many studies of oil spill impacts exist to guide expectations.	Avoid dredging deep pits; develop methods to predict where sand sheets experience high rates of sediment transport, and target those for dredging to avoid slow filling of pits and persistence of fine sediments without renewal of oxygenated water flows[3,4,5,9,14,15]. Regular inspection of fuel system on dredges and maintenance of available containment boom can mitigate oil spill impacts.		This water quality issue appears connected to excavation of deep pits, so avoiding that extraction design should be an effective mitigation. Development of a method to model and predict where bottom sand transport is large enough to refill dredge pits rapidly would be very beneficial. Oil spill mitigations should be effective for the small spills that may arise.	

[1]Rabalais et al. 2001a, [2]Diaz and Rosenberg 2008, [3]Saloman et al. 1982, [4]Van Dolah et al. 1998, [5]Engstrom and Marinelli 2005, [6]Tuvikene 1995, [7]Carls et al. 1999, [8]National Research Council (NRC) 2003, [9]Pullen and Naqvi 1983, [10]Louisiana Department of Natural Resources (LADNR) 2002, [11]U.S. Department of Commerce (USDOC) 2009, [12]Bergquist and Crowe 2009, [13]Palmer et al. 2008, [14]Murawski 1969; [15]Taylor Biological Company 1978

Table 4.6

Impacting mechanism for OCS dredging on benthic resources: *Entrainment near the seafloor*.

Impact Pathway	*Potential Effects on Benthic Resources, Community and Habitat*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☐	Direct Effects Mortality likely for benthopelagic organisms, such as mysids; invertebrate larvae; and large mobile megabenthos like sea stars, echinoids, octopi[1,2,3,4,5,6]. Indirect Effects Loss of an important prey for juvenile and small pelagic fishes, as well as juvenile demersal fishes, many of which concentrate in benthic structural habitats serving as fish nurseries[6]. Cumulative Effects None of significance	Impacts on large, mobile mega-benthos likely confined to the dredging footprint[4,5,6], but impacts on mysids would influence their abundance over wider areas of water column[4,5] determined by physical dispersion processes.	Impacts of megabenthos may last years because of the relatively long lifespans of these species, while mysids have multiple generations in a yr, leading to rapid replacement rates after dredging ends. Invertebrate larvae are produced by reproduction at least annually and typically more frequently.	Will occur throughout the period of active dredging operations.	Megabenthos cannot move faster than the approaching dredge head, so their mortality should be virtually complete within the dredging footprint. Mysids avoid entrainment at night when they rise up in the water column, although their entrainment probably approaches 100% where the dredge passes during day-time, and new mysids continue to descend into the dredged area every day.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	There is no information on entrainment rates of these organisms during OCS sand mining[4,5].	None known or proposed.		Not known.	

[1]Saloman 1974; [2]Culter and Mahadevan 1982; [3]Johnson 1982; [4]La Salle et al. 1991, [5]Reine and Clarke 1998; [6]Nightingale and Simenstad 2001

Impacts of vessel operations include inadvertent contact of the dredge with hard-bottom habitat, which can leave many square meters of damage to and losses of epifaunal invertebrates attached to the substrate surface. Such damage can be mitigated by careful mapping of all hard-bottom habitat in the vicinity of all potential borrow areas and establishing an effective separation distance as a buffer to minimize risk of accidental encounters. Pipeline corridors also carry risk of encounters between the pipeline itself or by the ship laying the pipe and hard-bottom benthos, with mortalities caused by crushing, scraping, and abrasion. Mitigation is achieved by establishing an effective buffer distance away from any hard-bottom habitat area (Table 4.7).

Unexploded ordnance, exposed shipwrecks and other marine debris simply become three-dimensional hard structures that provide habitat for the recruitment of benthic epibiotic organisms; the exposure of such structures is likely to enhance hard-bottom benthic communities, at least temporarily until reburial occurs (Table 4.8).

Although monitoring of the benthos has been done after dredging for sand resources, several information gaps exist that prevent confident determination of the importance of some benthic impacts from dredging. The most serious of these gaps involves the inability to infer how even large changes in benthic resources affect the functional value of the benthos to higher-level consumers in the broader ecosystem. Except for the geographically limited cases of two, and perhaps soon to be seven more, federally listed stony corals, and four species of fished benthic bivalves on sandy OCS bottom (surf clams, ocean quahogs, ocean scallops, and calico scallops), the benthic resources themselves are valued only for the ecosystem services that they provide. Development of functionally based metrics of significance to ecosystem services is in its infancy. Further studies need to be done to assess the severity of the changes in benthic resources after dredging and their effects on the feeding, production, reproduction, and migrations of fishes, crabs, and shrimps of value.

New studies are needed to assess what is most important about the benthos – sustaining functional values–especially the unique or special functional services of sand ridges and other seafloor habitats that are disturbed during sand mining. Several hypotheses that follow from our current understanding of benthic community response to and recovery from disturbance could serve to focus new functionally oriented OCS research. For example, the development of high densities and biomass of small, shallow-burrowing benthic invertebrates during Phase I of recovery (Fig. 4.1) could provide greater availability of food resources for demersal predators than are present in the original undisturbed bottom at the expense of reduced abundances of longer-lived, deeper-dwelling species like some bivalve mollusks. One could thereby hypothesize that provisioning of prey for valuable demersal predators could be enhanced by bottom disturbance associated with sand mining. Filling these knowledge gaps will often require use of experimental approaches, new technologies like acoustic tags with telemetry, and broader ecosystem-based approaches to replace or augment the traditional monitoring. In addition, some information gaps relate to developing and improving mitigation actions. For example, development of a reliable yet easy means of characterizing hydrodynamic regime at the seafloor and capacity for sand transport could improve ability to select borrow areas that show rapid recovery of pre-existing benthic resources and communities with reduced risk of leaving depressions that fill only slowly with organic mud and attendant problems of low oxygen and high hydrogen sulfide. Rigorous tests are needed of whether dredging in strips or an equivalent dredging design that preserves patches of unaltered benthos speeds up recovery of benthos despite the additional mortality associated with slumping and the recognition that larval settlement not immigration is required to recolonize areas the size of borrow areas.

Table 4.7

Impacting mechanism for OCS dredging on benthic resources: *Vessel operations and interactions (including laying of pipelines).*

<table>
<tr><th>Impact Pathway</th><th>Potential Effects on Benthic Resources, Community and Habitat</th><th>Spatial Extent of Impact</th><th>Duration of Impact</th><th>Frequency of Impact</th><th>Severity of Impact</th></tr>
<tr><td rowspan="3">Direct: ☒
Indirect: ☒
Cumulative: ☐</td><td>Direct Effects
Crushing of epifauna on hard-bottom habitat under the footprint of the pipeline and anchoring and pumping systems[1].
Accidental grounding of dredge vessels also crushing epifaunal.
Indirect Effects
Regrowth of benthos on hard bottom where crushing took place would initially involve short-lived opportunists before late-succession, longer-lived benthos would return[3].
Cumulative Effects
None known</td><td>Along the pipeline corridor and possibly covering a wider area where storms have shifted the pipeline position.</td><td>Lasting while the pipeline is in place, then for a period of years afterwards while the late-succession community of clonal, long-lived epibiota like sponges and corals recolonizes and grows back.</td><td>Depends on how many projects use pipelines to transfer sand from borrow areas or rehandling areas to the placement area and how successfully pipeline track avoids hard-bottom habitat.</td><td>100% mortality of hard-bottom epibiota along the footprint of the pipeline corridor[2,4]; recovery of late-succession community after pipeline retrieval would require years.</td></tr>
<tr><td>Availability of Empirical Information</td><td colspan="3">Mitigation Measures</td><td colspan="2">Mitigation Effectiveness</td></tr>
<tr><td>No data on impacts of laying temporary pipeline on benthos. Some studies that surveyed after dredges inadvertently grounded on hard-bottom habitat documenting losses of epibiota by crushing and scraping.</td><td colspan="3">Establishment of pipeline corridors that do not or only minimally intersect with hard-bottom habitat. Off-site introduction of reef-balls, artificial reef[5,6] or some other hard-bottom substrate that would be colonized by epibiota, or coral propagation and transplantation[7] which could compensate for the loss of hard-bottom benthos from injury to the time that the community recovers. Pipelines could be supported and suspended above hard-bottom areas[6].</td><td colspan="2">Evaluations of reef-balls in particular have been conducted, assessing epibiotic cover and fish utilization, revealing quantitative information supporting the use of this technique to provide compensatory resources of the same kinds lost to crushing by the pipeline and by dredge grounding.</td></tr>
</table>

[1]Johnson 1982, [2]Gage 2002, [3]Blair et al. 1990b, [4]Barnette 2001, [5]USACE 2003b, 2003a, [6]Coastal Planning & Engineering Inc (CPE) 2011a, [7]FDEP 2011

Table 4.8
Impacting mechanism for OCS dredging on benthic resources: Exposed UXO, shipwrecks, other hard structures temporarily exposed during dredging.

Impact Pathway	*Potential Effects on Benthic Resources, Community and Habitat*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☐	Direct Effects Settlement of epibenthos on exposed hard substrata and community development. Indirect Effects Small mobile invertebrates colonize and associate with the emergent epibenthic species providing habitat, and bottom flow field would be modified to include eddy development. Cumulative Effects Little cumulative impact except perhaps on rebuilding of depleted reef-fish populations through provision of more hard-bottom habitats and associated structural and prey resources.	Confined to the rare locations where such structures became exposed within footprint of dredging.	Dependent on rate of sedimentation, which is likely to rebury exposed hard structures within the dredged depressions, and on storms that are likely to impose a cycle of exposure and burial.	Because magnetometer surveys are done well prior to dredging, exposure of a large hard object is a rare event.	The effects of exposing large hard objects and replacing an area where only sediments previously were found on the bottom are largely beneficial biologically, with greater abundance, biomass, and diversity of benthic invertebrates and enhanced use by demersal and some pelagic fishes. However, any explosions of UXOs would harm fishes and damage to shipwrecks would degrade cultural resources.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	There are no studies on development of epibenthic communities and fish interactions with hard structures temporarily exposed during dredging.	Magnetometer surveys are required prior to dredging to identify and avoid such areas.		Should be effective, which could be documented by synthesis of past magnetometer surveys relative to later discoveries of buried hard structures.	

4.3 FISHES AND ESSENTIAL FISH HABITAT

Many OCS borrow areas are sand ridge and swale complexes (the ridges are often referred to as shoals in the literature), which are designated as EFH for migratory pelagic fishes (e.g., king mackerel, Spanish mackerel, cobia, and dolphin) by the South Atlantic Fishery Management Council (SAFMC). EFH is defined broadly under the Magnuson-Stevens Act as “those waters and substrate necessary to fish for spawning, breeding, feeding and growth to maturity”. These habitats, like analogous inner shelf habitats, also serve as a benthic nursery area, refuge, and feeding areas for a variety of ecologically important and commercially and recreationally harvested demersal fish and invertebrate resources. Hard-bottom reef habitat on the continental shelf is also identified as EFH for demersal reef fishes including the economically valuable snapper-grouper complex. We discuss the potential impacts to EFH (hard bottom and coral) from the turbidity and siltation that arises from sand mining and rehandling in Section 4.2.

Three separate processes involved in sand mining include dredging of the borrow area, conveyance, and rehandling of sand close to shore: each of these processes has the potential to cause direct, indirect, and cumulative impacts on biological resources and seafloor habitats (see Section 3). To date, only a few studies have rigorously assessed the potential biological and ecological impacts associated with sand mining on fishes and large motile epifauna (e.g., crabs, shrimp), and no attempts have been made to synthesize and integrate the nature of impacts across different borrow areas and operational activities using data from both domestic and international studies. This section synthesizes all relevant studies on the effects of dredging, conveyance, and rehandling of sand and gravel, where appropriate, on fishes and large motile epifauna.

4.3.1 Sand Ridge and Swale and Complexes as Important Fish Habitats

The SAFMC, Gulf of Mexico Fishery Management Council, and others (Diaz et al. 2004; Slacum et al. 2006; Vasslides and Able 2008; Slacum et al. 2010; Woodland et al. 2011; including recent environmental consultations by NASA 2010c and Coastal Planning & Engineering Inc (CPE) 2011a, among others) identify sandy shoals, ridges, and near-ridge areas as important fish habitats and forage habitats for migratory pelagic species (e.g., king mackerel, Spanish mackerel, cobia, and dolphin) and other recreationally and commercially important species (see for example Table 4.9). Some of these areas may also be visited occasionally by endangered species (Atlantic salmon, *Salmo salar*; Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*; Gulf Sturgeon, *Acipenser oxyrinchus desotoi*; shortnose sturgeon, *Acipenser brevirostrum*; smalltooth sawfish, *Pristis perotteti*) as the Atlantic and Gulf of Mexico OCS provides habitat to these species (National Oceanic and Atmospheric Administration (NOAA) 2012, http://www.nmfs.noaa.gov/pr/species/fish/).

Based on observations on the east Florida continental shelf, Gilmore (2008) described the importance of shoals based on three critical functions: refuge from predation, food supply, and spawning. Shoals, particularly those with relief, provide refugia habitats for schooling planktivores (e.g., anchovies, sardines, herring, menhaden and scad) that are key prey for pelagic predators (e.g., king and Spanish mackerel, sailfish, marlin, swordfish, dolphin, amberjack and almaco, wahoo, little tunny, bonita, and blackfin tuna). Similarly, resident year-round species found on sand ridge-swale and shoal complexes of the Mid-Atlantic region include dusky shark, lined seahorse, striped bass, tilefish, blackbelly rosefish, summer flounder, windowpane, skates,

Table 4.9

Fish and large motile invertebrate species commonly or temporarily associated with sand ridge and swale complexes, and the most likely function provided by these habitats. Not a complete list. Sources: CSA et al. 2010; Diaz et al. 2006; Slacum et al. 2006; Gilmore 2008; Vasslides and Able 2008, Slacum et al. 2010; NASA 2010c; and citations therein. (C) indicates species of concern (NOAA 2012, http://www.nmfs.noaa.gov/pr/species/fish/).

Species by Trophic Guild	Potential Habitat Function		
	Refuge	Feeding	Spawning
Planktivores	X	X	X
Atlantic menhaden (*Brevoortia tyrannus*)			
Atlantic thread herring (*Clupea harengus*)			
Bay anchovy (*Anchoa mitchilli*)			
Blueback herring (*Alosa aestivalis*) (C)			
Butterfish (*Peprilus triacanthus*			
Cuban anchovy (*Anchoa cubana*)			
Redear sardine (*Harengula humeralis*)			
Round scad (*Decapterus punctatus*)			
Scaled (*Harengula jaguana*)			
Spanish sardine (*Sardinella aurita*)			
Striped anchovy (*Anchoa hepsetus*)			
Benthic/demersal carnivores/invertivores	X	X	
American sand lance (*Ammodytes americanus*)			
Atlantic rock crab (*Cancer irroratus*)			
Atlantic sharpnose shark (*Rhizopriondon terraenovae*)			
Black drum (*Pogonias cromis*)			
Blackbelly rosefish (*Helicolenus dactylopterus*)			
Blue crabs (*Callinectes sapidus*)			
Butterfish (*Peprilus triacanthus*)			
Clearnose skate (*Raja eglanteria*)			
Dusky shark (*Charcharinus obscurus*) (C)			
Flounder (*Paralichthys* spp.)			
Lady crab(*Ovalipes ocellatus*)			
Little skate (*Leucoraja erinacea*)			
Lizardfish (*Synodus foetens*)			
Northern pipefish (*Syngnathus fuscus*)			
Red drum (*Sciaenops ocellatus*)			
Sand shrimp (*Crangon septemspinosa*)			
Sand tiger shark (*Odontaspis taurus*) (C)			
Scup (*Stenostomus chrysops*)			
Sevenspine bay shrimp (*Crangon septemspinosa*)			
Smallmouth flounder (*Etropis microstomus)*			
Spiny Dogfish (*Squalus acanthias*)			
Spotted hake (*Urophysis regia*)			
Summer flounder (*Paralichthys dentatus*)			
Tilefish (*Lopholatilus chamaeleonticeps*)			
Weakfish (*Cynoscion regalis*)			
Windowpane flounder (*Scophthalmus aquosus*)			
Winter Flounder (*Pleuronectes americanus*)			
Winter Skate (*Leucoraja ocellata*)			

Table 4.9 Fish and large motile invertebrate species commonly or temporarily associated with sand ridge and swale complexes, and the most likely function provided by these habitats. Not a complete list. Sources: CSA et al. 2010; Diaz et al. 2006; Gilmore 2008; Slacum et al. 2006; Slacum et al. 2010; NASA 2010c; Vasslides and Able 2008, and citations therein. (C) indicates species of concern (NOAA 2012, http://www.nmfs.noaa.gov/pr/species/fish/) (continued).

Pelagic carnivores/piscivores		X	
Almaco (*Seriola rivoliana*)			
Amberjack (*Seriola dumerili*)			
Atlantic butterfish (*Peprilus triacanthus*)			
Blackfin tuna (*Thunnus atlanticus*)			
Cobia (*Rachycentron canadum*)			
Dolphin (*Coryphaena hippurus*)			
King mackerel (*Scomberomorus cavalla*)			
Little tunny (*Euthynnus alletteratus*)			
Spanish mackerel (*S. maculatus*)			
Striped bass (*Morone saxatilis*)			
Swordfish (*Xiphias gladius*)			
Wahoo (*Acanthocybium solanderi*)			

hakes, drums, and sand flounders; transient species include pelagic species such as sharks (Squalidae and Carcharhinidae), herrings, anchovies, mackerels, Atlantic menhaden, cobia, bluefish, and butterfishes, among others (CSA et al. 2010). Slacum et al. (2010) and a review by CSA et al. (2010) further indicated that sand ridge and swale complexes off the coast of the Mid-Atlantic region are important habitat for a variety of benthic invertebrates and demersal and pelagic fish species. Tens of fish species use these habitats, although their composition, distribution, and abundance fluctuate seasonally (i.e., high late summer-fall diversity and low winter diversity, reflecting the presence/absence of the highly migratory boreal or warm-temperate/subtropical species). Sand ridge and swale complexes also harbor numerous invertebrates and smaller fish, which are a key food source for many demersal fish species (e.g., red/black drum, weakfish, spotted seatrout, and whiting), thus providing important habitat for the trophic base within these areas.

In addition to adult fish, larvae of at least 34 demersal fishes (e.g., sand dance, gobies, searobins, Atlantic croaker, smallmouth flounder, weakfish, and dusky pipefish) are found in sand ridge and swale areas, while others (e.g., anchovy, Atlantic menhaden, Atlantic mackerel, and searobins) spawn in the surrounding habitats (CSA et al. 2010). Observations based on the collection of eggs and larvae of schooling planktivores and gravid adults of several species (e.g., king mackerel, tripletail, red drum and goliath grouper) on shoals or adjacent areas indicate that these habitats are important spawning grounds for some fish species. Similar observations were also obtained via passive acoustics (Luczkovich et al. 1999) showing the use of ebb-tidal delta shoals as spawning grounds for weakfish, red drum, and spotted seatrout. Ship/Trinity/Tiger shoal complexes off Louisiana have also been identified as important breeding, spawning, and feeding grounds for blue crab (*Callinectes sapidus*) and fishes (Gelpi et al. 2009; Condrey and Gelpi 2010). Sand ridge and swale complexes are also important habitat for juveniles of species that utilize inshore estuarine and nearshore shelf areas as nurseries (CSA et al. 2010), and for many benthic fish species that rely on morphologic features as a part of a broader, cross-shelf habitat continuum whereby they complete their life cycles.

The ecosystem function of these habitats in the OCS and with regards to fishes has only been recently studied. Vasslides and Able (2008) documented that shoreface sand ridge and near-ridge habitats support a higher species abundance and richness than surrounding habitats on the inner continental shelf. Fish species most commonly found at the top of the ridge included a variety of juvenile fish, as well as many key prey (e.g., sand lances, anchovies, smallmouth flounder) of resident and transient piscivores (Vasslides and Able 2008). In contrast to the findings by Vasslides and Able (2008), Diaz et al. (2006) found a lack of fish association between Sandbridge Shoal and nearby areas off Virginia, possibly related to the low variation in sediment grain size and similar bed roughness between surveyed areas, and to the low occurrence of biogenic structure over the study area. Similarly, Slacum et al. (2010) documented greater abundance, diversity, richness of finfish on adjacent flat bottoms, as compared to shoal habitats, possibly resulting from greater availability of benthic forage in flat bottoms, or from differences on bottom depth between these habitat types. Reported differences in the relative ecological importance of sand shoals across studies may be the result of numerous variables, including sampling gear and methodologies, targeted species, spatial and temporal scales, and site-specific features.

Despite efforts to characterize the biological fish diversity in sand ridge and swale complexes, little information is available on the preferential use of specific shoal areas by fishes and various mobile species, and the ecological value of individual shoals as EFH has not been specifically addressed (Slacum et al. 2006). Nonetheless, understanding the ecological functions of these habitats, even at a site-specific scale, within the larger continental shelf is critical to properly manage and mitigate activities on the OCS. Overall, more systematic research is needed to understand the ecological roles of these habitats, as well as their fish abundance, diversity, richness, and their ecological value relative to adjacent habitats (e.g., muddy swales).

4.3.2 Potential Environmental Effects and Mitigation Methods on Fishes and Fish Habitats from OCS Sand Dredging by Impacting Mechanism

Conceptually, dredging of sand from the U.S. continental shelf has the potential to cause adverse consequences on fishes, fish habitats, and other marine resources. Consistently, in its fishery ecosystem management plan, the SAFMC raised concerns regarding the excavation of offshore shoals (SAFMC 2003, 2009) and indicated that these activities may disrupt the shoal’s ecological services (e.g., benthic nursery area, refuge, and feeding grounds) to a variety of fishery resources. This sentiment is shared not only by domestic researchers and institutions (Greene 2002; Johnson et al. 2008), but also by international groups (ICES 1992; Saunders and Roberts 2010; Tillin et al. 2011).

Most adult fish and mobile demersal fish species are less likely to be affected by dredging activities than shellfish resources as they may escape injury by avoiding areas of active sediment removal. However, higher risks of impacts are associated with: redeposition of sediments and smothering of fish eggs on spawning grounds; entrainment of eggs, early life stages, and benthic fish species by suction dredges; exposure of pelagic eggs to elevated suspended sediments; removal of benthos that provides an important food sources for demersal fish species; and loss of habitat for finfish (La Salle et al. 1991; Desprez 2000; Saunders and Roberts 2010; Tillin et al. 2011). Other assessments (SAFMC 2009) have also indicated that the disturbance and increased benthic prey availability created by dredging operations can attract resident and non-resident fishes, exposing them to adverse levels of turbidity, sound, and possibly resuspended pollutants present in the sediment being dredged. However, these disturbances are short-lived, and their effects are unlikely to cause long-term effects.

The most commonly idealized and perceived concerns on the detrimental impacts resulting from OCS dredging on fishes (Greene 2002; Johnson et al. 2008; SAFMC 2003, 2009) include:

- Decreased overwintering habitat for anadromous and coastal species;
- Decreased habitat availability for highly migratory species;
- Decreased spawning habitat for fishes with demersal eggs;
- Direct mortality, particularly of species present and even concentrated in offshore sand mining areas during the spawning season;
- Displacement of resources and removal of substrates that provide habitat for fishes and invertebrates;
- Reduced prey availability and removal of key food resources (e.g., small fishes, slow-moving/non-motile infauna, and epifauna);
- Conversion of habitats to potentially less supportive habitat (e.g., anoxic holes, formation of large furrows, and muddy bottom);
- Increased turbidity in the water column, which may reduce foraging efficiency and cause increased stress from gill clogging;
- Increased sedimentation causing smothering and burial of early non-mobile life stages and key food sources;
- Alteration of ecological interactions and energy flow caused by localized disruption of the food web base (invertebrates); and
- Release of toxic materials during sand mining operations.

In contrast, OCS dredging activities can also increase the complexity of the bottom topographic (increased vertical relief and roughness) (as seen in Figures 2.8 and 2.10), uncover structures, and remobilize carbon-rich substrates, providing habitat and forage opportunities for many species. Whether or not these operations cause negative or compensatory responses need to be judged on an area by area basis, with careful consideration of temporal and spatial scales.

A synthesis of the impacts of sand mining, both detrimental and compensatory, on fish and fish habitats is presented here by discussing relevant documents on the major impacting mechanisms discussed in Section 3.3, as well as by summarizing mitigation strategies. Section 4.3 discusses specifically fishes and large motile epifauna (e.g., crabs, shrimp), while other invertebrates are discussed in Section 4.2.

4.3.2.1 Alteration of benthic habitat at the borrow area

The main habitat distinctions identified for benthos (Section 4.2.2.1) also apply to fishes. Specifically, the fundamental dichotomy between hard and sedimentary bottom has dramatic effects on the abundances and species composition of fishes that use the bottom habitat. Within the hard-bottom habitat types, the more continuous rocky or carbonate reefs differ from hard bottoms comprised of gravel, cobble, or boulder-sized rocks. Furthermore, vertical relief off the seafloor is an important factor, modifying types of associated fishes and intensity of fish habitat use. In the inner shelf, the particle-size distinction between sandy sediments and muddy sediments represents a determinant of the resident benthic invertebrate community, which affects its productivity and desirability as a source of forage for fishes.

One of the greatest concerns regarding sand mining is the localized disturbance and potential loss of benthic habitat through alterations of the physical characteristics of the seabed (see Section 4.2). The degree of alteration of benthic habitats from mining depends on the intensity of OCS sand borrow extraction, the temporal and spatial extent of dredging, the resilience of the potentially impacted biological resources, and their recolonization ability from nearby and distant unimpacted populations. Dredging can reduce habitat viability, directly displace and/or remove the native community, and disrupt the spawning grounds of species with demersal eggs. Seabed alteration can also disrupt predator/prey interactions, resulting in negative impacts to fish and shellfish populations (Johnson et al. 2008). However, offshore sand mining operations can also modify the nature of seabed habitats by increasing surface area and bottom complexity/roughness and providing opportunities to species that may not have been capable of originally inhabiting the area.

Diaz et al. (2004) indicated that shoal removal would displace mobile fishery resources that would need to relocate to a new habitat. Gilmore (2008) argued that habitats used as nursery grounds, or occupied by abundant species, will saturate rapidly during recruitment, suggesting that many benthic organisms are habitat/space limited. Consequently, removal of a portion of a shoal habitat may proportionally reduce essential fish habitat for habitat-dependent fish species, depending on how habitat is defined. For instance, removal of habitat for abundant and dominant species (e.g., sardines, menhaden, and herring) may result in a reduction of food resources for commercially and recreationally important species (e.g., mackerel) (Gilmore 2008). Recent studies (Gelpi et al. 2009; Condrey and Gelpi 2010) showed that Trinity and Tiger Shoals off Louisiana and the surrounding area are important offshore spawning, hatching, and foraging grounds for blue crabs in the Gulf of Mexico at least from April to October, when mature female crabs appear to be in a continuously iteroparous spawning cycle. The authors cautioned that dredging large-volumes of sand over a long period could cause some decline in blue crab fecundity and condition factor (crab weight per unit of length) through a reduction in food availability and increased mortality of crab larvae from increased suspended sediments in the water column. Yet, given the size of these Gulf of Mexico features relative to the size of the borrow areas and that some of these areas are unlikely to be dredged because of conflicts with oil and gas infrastructure, more studies are needed to validate the factors and scale of operations that would result in detrimental effects on the blue crab population. At the time of this report, no dredging had occurred on Ship Shoal.

In contrast, Nedwell et al. (2004) recorded enhanced species richness, population density, and biomass of benthic macrofauna within an area (600 m wide and 2 km long) along the axis of tidal streams from the North Nab Production License Area dredge site (east of the Isle of Wight, U.K.), possibly from organic enrichment from sources within the dredged site. It is important to note that these extracted aggregates were predominantly gravel. Newell and Seiderer (2003) also pointed out that, under natural conditions, most fishes and mobile invertebrates (except for less actively mobile, more vulnerable life-cycle stages–eggs and larvae) are likely to avoid areas of disturbance or high turbidity, but indirect effects may result from reduction or alteration in the food available from benthic resources (see Section 4.7). However, there is little evidence (at least from studies in the U.K.), arising in part from the lack of rigorous assessments, that losses from the benthic invertebrate community in dredged areas have a detectable impact on the ‘carrying capacity’ of the surrounding seabed for mobile epibenthos and commercial fish resources (Newell et al. 2004a). Direct impacts on fishery resources, at least in studies from the U.K., are primarily due to losses of eggs and larvae of demersal species, rather than to direct impacts on commercially significant target species. In fact, one of the main concerns raised by the International Council for the Exploration of the Sea (ICES) Working Group on the Effects of Extraction of Marine Sediments on Fisheries was the potential dredging in spawning areas of commercially important species with demersal eggs (e.g., Atlantic herring *Clupea harengus*, and sand lance *Ammodytes marinus*) (ICES 1992). However, little is known regarding the direct impacts of sand extraction on spawning grounds, as well as their recovery to optimal spawning conditions particularly in the OCS. Recommended mitigation strategies aimed at minimizing impacts on spawning grounds include seasonal restrictions and/or exclusion zones (environmental windows), accompanied with robust monitoring programs that measure the effectiveness of such strategies (Sutton and Boyd 2009).

Although impacts on spawning/nursery grounds from dredging activities are difficult to assess and have not been comprehensively evaluated, some generalizations can be made for groups of species whose life histories are well documented. Dredging may potentially impact members of the family Sciaenidae (drums, croakers, weakfish), known for utilizing offshore sandy habitat to spawn (Wilson and Nieland 1994; Luczkovich et al. 1999; Roumillat and Brouwer 2004 cited in Nairn et al. 2007), and several flounder and eel species (e.g., *Ophichthidae, Ophidiidae*) known for utilizing offshore sand/mud habitat as nursery grounds. Again, the potential impacts on those vulnerable populations must be evaluated relative to their spatial distribution and spawning/nursery seasonality in relation to the scales, both spatial and temporal, of OCS sand dredging operations.

Another important aspect regarding sand extraction and habitat alteration deals with the recovery of the functional habitat to pre-extraction conditions. Some researchers have indicated that the temporal impacts of sand mining on mobile demersal fisheries resources are related to the recolonization rate of benthic communities, particularly preferred prey crustaceans, as well as to the availability of similar prey in nearby areas (Diaz et al. 2004). A large proportion of juvenile fishes feed on epifaunal and infaunal crustaceans and, therefore, sand mining activities that enhance the production of crustaceans in the sediments could improve habitat quality of the overall demersal fish community. For instance, removing at least the top meter of sand from Fenwick Shoal, Maryland prior to the spring/summer recruitment would favor the recolonization of crustaceans, which would reach sufficient levels to support the returning demersal fishes (Diaz et al. 2004). By contrast, dredging activities that end prior to the fall/winter recruitment would favor annelids, which are not the preferred prey of demersal fish species (Diaz et al. 2004).

As suggested above, and relative to inherent seasonal and storm-driven changes (Posey and Alphin 2002), long-term impacts to fishes and large mobile invertebrates depend on the recovery of the benthic community, which is a function of the rate at which the excavated area refills and the grain-size distribution and water quality during refilling. Results from studies on physical recovery rates following sand and aggregate mining are variable because of the site-specific characteristics and variability in extraction intensity (see Section 4.2.2.1 for details).

Modification of benthic habitat can also occur through sediment rehandling in nearshore habitats. Temporary placement of sand in nearshore environments directly impacts benthic organisms, though the effects of burial depends on the vertical and horizontal dimensions of the placement layer, rate of sediment placement, and size and behavior of benthic organisms. Burial can lead to mortality of fishes and large motile invertebrates with low mobility, while more motile species and those that are able to burrow can survive by moving outside the placement area, or to the surface of the placement layer. Studies have examined the effects of sediment burial on estuarine invertebrates (Maurer et al. 1986; Hinchey et al. 2006) and found species-specific abilities to migrate through sandy sediment. The same studies reported that survival from sediment burial depends on an organism's tolerance to hypoxic conditions while buried, overburden stress (i.e., the force exerted on organisms by sediment burial, and a metric that incorporates the bulk density of the sediment and the depth of burial), burial time, and behavior (motility, position within the sediment layer, physiological adaptations). These studies, however, focused on species that are inherently tolerant to high sedimentation rates compared to species inhabiting offshore and nearshore placement habitats, which would probably experience low sediment deposition and burial. Based on the observations above, it is likely that most fishes and large epibenthic motile invertebrates will temporarily avoid areas of fill placement, while their eggs may succumb to the effects of sediment burial. Most benthic macrofauna will also be killed by burial from sand deposited on rehandling areas. However, such areas usually represent very small percentages of available habitat.

Although the potential direct effects to fisheries from sand dredging are generally unknown, some monitoring studies related to OCS sand dredging indicate minimal or nonexistent impacts to fisheries (Van Dolah et al. 1992; Van Dolah et al. 1994b; Hackney et al. 1996; Nairn et al. 2004). The conclusion of no-to-minimal impacts is based on the general assumption that most fishes inhabiting sand ridge and swale complexes are typically wide-foraging or migratory species, spending only part of their life cycle in the area. Furthermore, these borrow areas often represent a small fraction of the available sand shoal habitat (Nairn et al. 2004). In addition, the degree to which any fish makes exclusive use of sandy habitat of the sort targeted for sand mining is unknown. Thus, it is inferred that habitat alteration would probably have minimal effects on the overall fish community. However, cumulative effects of offshore sand mining projects have not been adequately assessed (SAFMC 2003), and only a limited number of field studies have examined impacts on individual species or fish populations (Nairn et al. 2004). Unfortunately, there has been equally limited study of different species' preferential use of or site fidelity to spawning or foraging grounds.

Researchers have attempted to quantify the potential impacts on fisheries from sand mining in offshore waters of the U.S. (Burlas et al. 2001; Nairn et al. 2004). Nairn et al. (2007)'s estimates of impact of the Holly Beach, Louisiana dredge pit on mobile invertebrate production and demersal fishes were based on known data regarding loss of benthic production per unit area, the extent of the affected area, recovery rates, and bioenergetic efficiencies. Potential loss of prey, expressed as an annual estimate, averaged 8.3 g m^2/year (range: 0.9-19.8 g m^2/year), multiplied by the estimated extent of impacts (190,600 m^2), yielded an estimate of benthic production loss of 1,582 kilograms (kg)/year (range: 172-3,774 kg/year). Under the assumption of 11 years for complete benthic recovery at the site, the total loss from this area was estimated at 17,402 kg (range: 1,892-41,415 kg). These estimates, combined with a 20% bioenergetic efficiency of the local predator community, yielded 3,480 kg (range: 378-8,283 kg) loss of demersal fish and invertebrate production over the 11-year period, equivalent to ~316 kg/year. It is noted that these calculations are static and do not take into account migration and natural variability of benthic biomass, among other dynamic variables. Overall, the anticipated loss of demersal fish production from dredging is relatively small when compared to the harvest of commercially important species. However, the scalability or synergistic effect is especially poorly understood. Burlas et al. (2001) reported no evidence of a dramatic change in the fish assemblage structure or catch per unit effort (CPUE) among borrow areas after dredging, and offshore finfish did not show a large change in their assemblage composition or abundance in relation to borrow mining. Analysis of the food habits of winter and summer flounder at an offshore borrow area did not change appreciably between the baseline time period and during dredging, and one and two years post-dredging data indicated no shifts in quantity or composition of diets in relation to dredging. However, these study results are only applicable to species with feeding plasticity that are able to find supplemental food in nearby unimpacted areas, while the effects of these activities on species with more restricted diets, or the overall impacts of larger sand mining areas on demersal fishes, were not discussed.

Studies in the U.K. have also provided insights into the effects of continuous aggregate extraction on fishery resources. Ware et al. (2011) described mitigation and annual monitoring efforts at two aggregate extraction sites in the U.K. The Hastings Shingle Bank has been licensed for aggregate extraction since 1989. Monitoring programs based on fisheries information were established to identify the impacts from aggregate dredging on two important local fisheries: Edible crab (*Cancer pagurus*) and sole (*Solea solea*) (Ware et al. 2011). Although one study (Bannister, 2004 cited in Ware et al. 2011) indicated that the decline in crab catches could not be directly attributed to dredging, it did not conclusively rule out dredging as a potential cause. CPUE of sole was not different between the extraction and control sites, and mitigation strategies were enforced to prevent extraction activities during the spring sole migration. It is worth noting that, although there is no evidence linking dredging activities to the decline in sole catches, the continuing decline in landings at Hastings, but not at other ports in the area, is a source of concern (Rogers and Nicholson 2002; Sutton and Boyd 2009). Although a seasonal dredging restriction (environmental window) during the spring spawning period (February-April) was put in place to minimize the impacts on sole onshore fish migration, the effectiveness of this mitigation measure has not yet been confirmed (Sutton and Boyd 2009).

The Inner Owers license area, located south of the coast of Sussex, is an area that has been licensed for aggregate extraction since 2005. The monitoring strategy at this site included the assessment of impacts to a nearby reef site and to nesting sites of black sea bream (*Spondyliosoma cantharus*) located inshore from the aggregate extraction area (Ware et al. 2011). This monitoring effort did not detect changes in nest densities relative to baseline densities, nor did it show accumulation of fine sediment in the area directly upstream of the dredging areas (Ware et al. 2011). In one study (Cooper 2005), anglers reported significant declines in catches of smooth hound (*Mustelus mustelus*) in an area east of the Isle of Wight, U.K.; as a large proportion (75%) of this area was dredged in 2001, dredging could not be ruled out as a causative factor. Studies indicated that this species is attracted to the presence of key prey species common in the area (mussels, soft-shelled crabs, and bass) (Plumb, 1996 cited in DEFRA 2001). Because a large proportion of the seabed was continuously dredged between

1993 and 2001, much of the benthic prey mussel bed may have been removed, possibly contributing to its decline in abundance.

These observations are in agreement with the accounts by anglers who reported declining catches of this species since the beginning of dredging operations in the area, potentially resulting from impacts on the mussel bed (Cooper 2005). Consistently, surveys of fishermen from different localities also indicated that, unlike the decline of fishery resources (e.g., lobster) from fishing activities, the decline in the brown crab fishery and juveniles of this species may have been associated with dredging in the area, particularly as these activities removed food sources that are limiting to crabs (Cooper 2005). The fact that the distribution of sensitive habitats (i.e., suspected spawning ground and migration route for female crabs) overlaps the extent of cumulative dredging activities (1993-2001) also points at the potential contribution of these activities to the decline of this species. However, other concurrent factors (such as overfishing and changes associated with natural variability) may have also contributed to the overall declines, and no supporting scientific evidence is available to assess the unconfounded impact of dredging on the brown crab.

Stelzenmüller et al. (2010) recently developed a marine spatial risk assessment framework for the U.K. continental shelf, in which sensitivity maps of eleven fish and shellfish species were overlaid with the occurrence of aggregate extraction activity to describe their vulnerability to dredging. Factors considered in sensitivity estimates included species' geographical distribution, threat status, importance to fisheries, habitat vulnerability, trophic guild, degree of association with the sediment bottom, and reproductive biology. The highest sensitivity to dredging occurred in coastal regions as well as in offshore areas known to be important nursery and spawning grounds. This framework, which incorporates relevant spatial scales, can be applied to offshore marine planning where species distribution data and ecological and biological knowledge are available and accurate (Stelzenmüller et al. 2010).

Valuable information has also been generated through pre-dredging assessments, data that can be used as a reference against which to assess changes in the fish assemblage structure during and after mining. In one study (Slacum et al. 2006), benthic and fisheries pre-construction data (composition, abundance, and biomass, plus baseline diet of resident fish species) was collected from two borrow areas (0.8 and 3.2 km offshore) and reference sites of similar characteristics located off Kitty Hawk, North Carolina one year prior to sand dredging. Fish had similar species richness, abundance, and community diversity between the borrow and reference areas, and catches at the borrow areas were lower in the summer and fall, and higher in spring. Although the authors indicated that shoals were not preferred over the reference site habitat, additional studies are needed to determine if these complexes provide a unique ecological function relative to adjacent habitats. This study also provided information on the link between resident fish species and benthic fauna, whereby Atlantic croaker stomachs contained polychaetes and shrimp, southern kingfish stomachs contained polychaetes, shrimp, and bony fish, and spot stomachs contained bivalves and bony fishes. Post-dredging reports were not available for this project; thus, potential changes to the fish community are not described here.

Gilliam et al. (2011) studied the abundance, density, and richness of fish species associated with coral reef communities off Broward County, Florida before (2000-2004), during (2005), and after (2006-2010) beach nourishment projects. All fish metrics were significantly lower during the construction phase compared to both pre- and post-construction periods. During construction (i.e., emplacement of sand, possibly similar to nearshore rehandling), the abundance of several species (tobaccofish, reef butterflyfish, redband parrotfish, bicolor damsels, and juvenile grunts) and one family (scarids) were dramatically reduced. However, given lower averages of these species over previous years, and substantial inter-annual variation in fish assemblages, the apparent 2005-2006 fish disturbance period may have not been entirely attributable to construction activities. Although this assessment did not completely resolve the link between changes in fish assemblage and dredging/fill operations, it showed that having several years of pre- and post-construction data are essential to better interpreting the impacts associated with a nourishment project in light of natural sources of variation (seasonal and inter-annual). Another study (Courtenay et al. 1980) assessed fish populations adjacent to borrow areas (nearshore reef areas impacted by rehandling of fill material and adjacent areas within 0.5 km from the borrow area) off Hallandale, Broward County, Florida seven years post-dredging. Overall, the fish fauna was rich, but one species, the dusky jawfish, was adversely impacted by a combination of dredging activities and beach erosion. This species excavates permanent burrows on the reef platform, which are abandoned under conditions of considerable stress. The reduction in the abundance of this species was the result of substrate and habitat alteration from deposition of fill materials (i.e., high deposition of fine-grained sediment that is unstable for burrowing). Two other species (belted cardinal fish and roughhead blenny), common in preassessment surveys, were not observed post-dredging on reefs adversely affected by incursion of sediments from dredging activities and erosion of the beach fill.

Mitigation strategies aimed at ensuring the long-term habitat functions of offshore sand ridge and swale complexes (Atlantic Coast of Maryland Shoreline Protection Project) have included a maximum dredging limit of 5% of the total volume of any shoal, uniform dredging over a wide area of each shoal (<3 m thick), maintaining the shoal profile and existing depths of the seafloor, avoiding the crest to maintain maximum relief off the seafloor, and dredging on the updrift/downdrift (accreting) side of the shoal ensuring the stabilization of the shoal as a geomorphic feature (USACE 2008a, Annex E). Similar mitigation strategies (Wallops Island, Virginia) also recommended uniform dredging to avoid the formation of pits (<2 m), dredging in areas that are accreting and avoiding erosional areas, and avoiding dredging along the entire length of the shoal (NASA 2010c). Site-specific mitigation strategies aimed at reducing impacts to EFH from sand rehandling in nearshore waters may include similar strategies to those implemented during previous OCS dredging and beach nourishment projects including: avoiding infill areas within 15 m of coquina or worm rock outcrops (USACE 2009); implementing a safe dredging distance of a minimum of 122 m from any significant hard-ground areas (USACE Jacksonville District 2011b); and continue monitoring programs (e.g., sedimentation and turbidity) during these activities (USACE 2009, USACE Jacksonville District 2011a; USACE 2011a). Many of these mitigation strategies also indicate the need for additional coordination and consultation with NMFS throughout the life of the project to avoid and minimize impacts on EFH. Additional recommendations (based on outputs from numerical modeling) were also given by Dibajnia and Nairn (2011) to ensure the regrowth of the shoal post-dredging conditions: maintain the ratio of the shoal height to its base depth to >0.65, and avoid dredging those shoals with <0.5 ratios; target shoals with <30 m base depths, but consider those at greater depths depending on their ecological importance; avoid taking material from the entire length of the shoal crest, but rather from the leading edge of the shoal; site specific conditions may limit material removal from the shoal-crest; modifications to suggested guidelines may need to be considered based on shoal-specific characteristics (e.g., wave and currents, depth, shoal morphology). Although these recommendations are not specific to fish, these are intended to preserve the integrity of the shoal, thus protecting the use of this habitat by fishes and large motile invertebrates.

Overall, there is relatively little quantitative information on the effect of habitat alteration from sand mining on fishery resources. Many studies agree that most fishes and large mobile invertebrates are capable of escaping injury by avoiding active dredging areas, and that the most likely impacts would result from changes to the food supply from benthic resources, loss of spawning habitat, and loss of eggs and larvae of demersal species. However, the degree to which loss of the early life stages impacts fish populations associated with sand ridge and swale complexes is largely unknown. Evidence from analyses of empirical data suggests that below-average recruitment results in lower spawning populations (Myers and Barrowman 1996). Yet, direct impacts on fishes from habitat alteration are difficult to quantify because large amounts of data are needed to differentiate sand dredging impacts from natural sources of variability. Existing mitigation strategies are primarily targeted to prevent large-scale degradation of benthic resources and are addressed in detail in Section 4.2.2.1.

4.3.2.2 Increased turbidity in the water column

The biological and ecological effects of short-term exposures to sediment plumes are generally anticipated to be minimal, likely the case in most OCS operations (see Section 3.3.2), with exposure duration of adult fishes and motile epibenthic macroinvertebrates lasting from minutes to hours, unless dredging is confined to an area with restricted water exchange and circulation (Saunders and Roberts 2010; Tillin et al. 2011). At persistently elevated suspended-solid concentrations, fish gills can be clogged or damaged (Hammer et al. 1993), and any irreversible impacts on these structures can have negative consequences (e.g., limiting predation avoidance capacity, increasing energy demands, and inducing suffocation). To compensate for gill obstruction, fish will produce excess amounts of mucus and increase “coughing” or “gill flaring,” reactions that are associated with high metabolic costs; thus, persistent gill blockage may have detrimental effects on the overall fish’s energy budget. Sediment plumes at the water surface can reduce water clarity and could lead to short-term avoidance of the working area and the impacted habitat by some fish, birds, and marine mammals (Saunders and Roberts 2010; Tillin et al. 2011). Reduction in water clarity by sediment plumes can also have non-life threatening and temporary impacts on the feeding efficiency of predatory fish species (e.g., mackerel and turbot) that rely on visual cues to detect prey (ICES 1992; USDOI MMS 2009), by leading to temporary avoidance of turbid waters (e.g., little tuna and yellowfin tuna) (Barry 1978). Conversely, plumes can also attract other fishes and large invertebrate species to areas with increased supply of crushed benthos. However, in practice, the impact of plumes is of a greater concern to animals on the seabed/water interface that are not able to avoid increased turbidity (less mobile species, sessile organisms, and sediment-dependent life stages). In these species, more severe effects (stress, tissue damage, reduced growth, mortality) may occur. Redeposition of suspended solids onto the surface of the seabed can bury and smother demersal fish eggs on spawning grounds and larvae (decrease spawning and emergence success), and suffocate filter-feeding benthos (ICES 1992), effects that are addressed in Section 4.3.2.3.

There have been relatively few studies on the effects of increased suspended particle concentration and turbidity on pelagic fishes following offshore sand mining; most studies have focused on the physiological and pathological responses to these stressors in freshwater and estuarine species. The large majority of studies on the biological effects of increased turbidity and suspended solids in the water column from dredging operations have been on nearshore, enclosed waters and estuaries (La Salle et al. 1991; Newcombe and Jensen 1996; Clarke and Wilbur 2000; Wilber and Clarke 2001; Clarke et al. 2002), where generally, the sedimentary material is much finer-grained and turbidity plumes are longer-lived. Conceptually, the coarser material, common to most, but not all OCS borrow areas, should contribute to smaller and shorter-lived turbidity plumes. A brief synthesis of such effects is provided below. It is important to note that fish species inhabiting nearshore and estuarine habitats likely have greater tolerance than offshore species to elevated suspended solids in the water column, and turbidity effects may be less important in offshore areas because of the composition of the sediments (mostly sand), water depths, and open-water currents. Therefore, the synthesis below represents worst-case conditions unlikely to occur in most OCS borrow areas.

Effects from exposure to high suspended-sediment concentrations in estuarine and freshwater fish (e.g., reduced hatching success, slowed growth, abnormal development, tissue abrasion, and increased mortality) generally occur after prolonged exposures to concentrations in the 270-7,000 mg/L range. This range of suspended solids on the Atlantic and Gulf OCS is unlikely, except in areas prone to fine-grained sediment loading and turbidity maxima such as the Louisiana inner shelf. Fish exhibit great variability in their sensitivity to suspended solids (from no effects to behavioral changes to mortality) depending on the species and life stage, as well as on the characteristics of the sediment. Generally, eggs and larvae of both marine and estuarine fish species appear to be more sensitive to suspended-sediment exposures (Newcombe and Jensen 1996; Wilber and Clarke 2001) than adults. Developmental delays and increased mortality rates of eggs and larvae have been documented at exposures ranging from 100 mg/L for 1 day (reduced egg hatching in striped bass and white perch) to <500 mg/L for 3-4 days (increased mortality in striped bass, American shad, yellow perch, and white perch) (Clarke and Wilbur 2000 and citations therein; Wilber and Clarke 2001).

Newcombe and Jensen (1996) conducted a meta-analysis on the effects of suspended solids on fishes in freshwater streams and estuaries and developed a quantitative approach for assessing the severity of effects (from no effects to lethal effects) of suspended sediments on a variety of fish species based on exposure duration and suspended-sediment concentration. Although this work does not propose specific thresholds, it provides a quantitative and conservative approach to estimating the severity of impacts on fishes associated with exposures to suspended solids (Figure 4.3). A similar approach was also developed for the impacts of excessive turbidity (in NTU) on fish species not adapted to high turbidity conditions (e.g., clear water fish) (Newcombe 2003). This approach may be more suited to comparisons of increased turbidity during offshore sand mining operations, as offshore species are not likely subject to naturally increased levels of turbidity, except during the spring freshet offshore Louisiana. Newcombe and Jensen (1996) also attempted to model impacts of suspended solids from dredging activities to marine organisms, but concluded that the available lethal and sublethal information was not currently sufficient to assess effects, a conclusion widely supported by subsequent reviews (Nightingale and Simenstad 2001; Wilber and Clarke 2001).

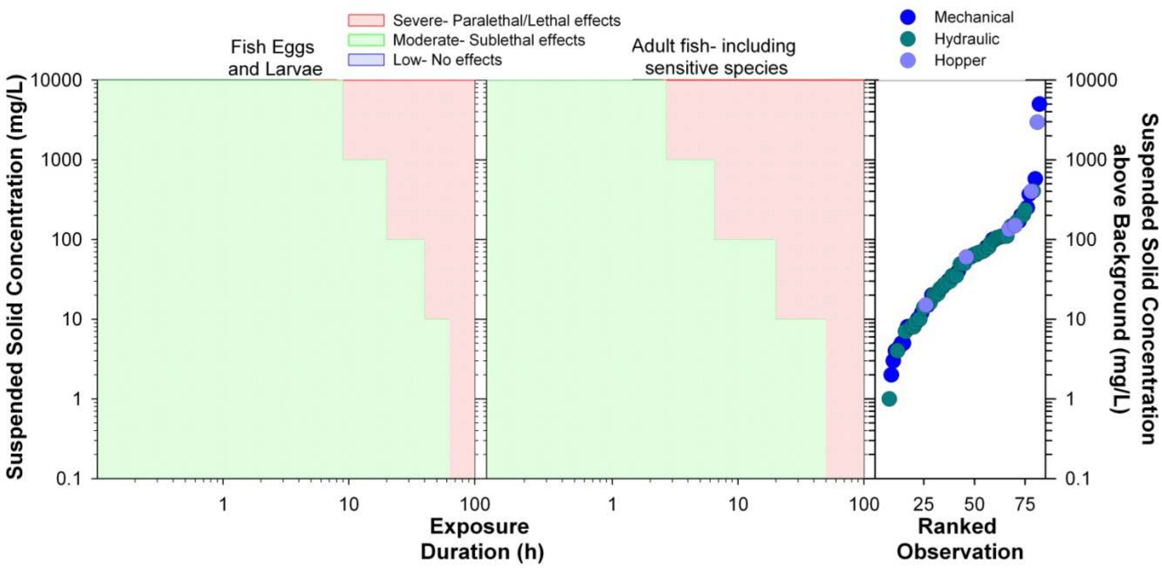


Figure 4.3 Models of the potential effects on fish eggs and larvae (left panel; recreated from data in Newcombe and Jensen 1996) and adult estuarine fishes (center panel; recreated from an updated version of the original model, W. Berry, USEPA, pers. comm.) from exposures to suspended-solid concentrations at various exposure durations. Moderate sublethal effects include reduction in feeding rates, physiological stress, habitat degradation, and impaired homing; severe-paralethal/lethal effects include reduced growth rates, reduced fish densities, mortality (0-100%), and severe habitat degradation. The far right panel shows the reported suspended-solid concentrations from navigation dredging operations in fine-grained sediment trapping river mouths, estuaries and nearshore areas (modified from Anchor Environmental 2003). These values were measured at distances from near-field to 98 m from the source, and are relative to study-specific background levels (range: 1-175 mg/L). Note that these concentrations are unlikely to occur in the sand-rich OCS.

Concentration of suspended solids at the water surface associated with nearshore rehandling operations are also much lower than in bottom plumes and are in the 0-100 mg/L range for hopper dredges and in the 0-150 mg/L range for cutterhead dredges (La Salle et al. 1991). Information on the spatial extent of suspended-solid plumes shows surface and bottom plume lengths associated with hopper dredges of 0-700 m and ≤1,200 m, respectively, and surface and bottom plume lengths associated with cutterhead dredges of 0-100 m and ≤500 m, respectively (La Salle et al. 1991). A monitoring report of the Broward County Shore Protection Project (Prekel et al. 2008) documented turbidity levels at control and experimental transects (water depths, bottom types, and wind and wave conditions were not provided) several months after beach construction, with average levels below 3.0 NTU. A related monitoring report (CPE 2011a) modeled turbidity plumes created by hopper dredges from the excavation of the proposed Borrow Area S1 in State waters, Panama City, Florida. Sixty minutes after the start of dredging operations, a modeled turbidity plume was confined to ~600 m across, with high turbidity levels (>19 NTUs) occurring in an area ~244 m across. Thirty and sixty minutes after cessation of dredge overflow, modeled turbidity levels dropped to <10 NTUs and <5 NTU, respectively, indicating that the episodic plume was mostly confined to the borrow area. Van Dolah et al. (1994b) reported turbidity values at the pipeline outfall in the surf zone of Folly Beach, South Carolina during beach nourishment of 100 NTU under calm conditions and of over 200 NTU under strong winds and rough seas (comparable to the field data in Manning et al. 2013). In both cases, relatively high turbidity levels (100 NTU) persisted within 1,000 m of the outfall, and values near 100 NTU were commonly observed. By comparison, the models by Newcombe (2003) suggest moderate sublethal effects on juvenile and adult fish at turbidity levels exceeding 400 NTUs. More recently, Manning et al. (2013) found depressed feeding rates of key prey, *Donax* and *Emerita*, in pompano fish exposed to turbidity levels (101 and 74 NTUs in feeding trials with *Donax* and *Emerita*, respectively) representative of levels measured in nearshore waters months after the completion of a beach nourishment project. This study further indicated that a slight increase of fine sediments in the water column impedes feeding rate by visually orientating predatory fishes.

Matsumoto (1984) simulated plumes of suspended solids (68-87% clay by weight) from a deep seabed mining operation and reported average particulate concentrations in the shipboard discharge of 5,800 mg/L, with a maximum near-surface particle concentration of 900 µg/L creating a relatively short lived plume (<5 hours) which was larger per unit area than full-scale commercial mining operations. This study suggested that changes in feeding behavior from exposure to dredging plumes by adult tunas is not likely of concern, given the transitory nature of these plumes and the ability of tuna to avoid and swim through turbid waters, and probable effects of mining on tuna and billfish eggs and larvae appear to be negligible. However, additional studies are needed to confirm these initial observations. Although a related study (Jokiel 1989) found reduced feeding activity of pelagic mahi mahi (*Coryphaena hippurus*) and larval death following 2- and 24-hour exposures to high concentrations of suspended solids (8,000 mg/L), eggs and larvae were not extremely sensitive to the levels of suspended-solids produced during sand mining activities. By contrast, experiments with suspended-sediment concentrations typical of sediment plumes (2-5 mg/L; mostly clay and limestone) (Westerberg et al. 1996) reported avoidance thresholds of ~3 mg/L (5 NTU) in 1-hr duration trials for benthic cod and pelagic herring at, as well as a strong negative linear correlation between the loss of egg buoyancy (from particle adhesion to the egg surface) and exposure duration at various suspended-sediment concentrations. Based on these studies, Westerberg et al. (1996) suggested that sediment plumes can be detrimental to pelagic and demersal eggs, and spawning success of fish and shellfish can be adversely affected by causing temporary avoidance of spawning grounds.

The most common mitigation strategy implemented for increased turbidity, both at the borrow and placement site, is water-quality monitoring (e.g., Prekel et al. 2008; CPE 2011a). Specific examples of required turbidity monitoring include: 1) background monitoring at the borrow's material rehandling area in nearshore waters at approximately 65 m from shore and 150 m up-current from the fill discharge or placement location (USACE 2011a); and 2) turbidity monitoring at background and compliance locations every six hours during offshore sand dredging and nearshore placement/construction activities (USACE 2011c). In both instances, for projects in Florida, work would be suspended if turbidity levels exceeded the state's water-quality standards (29 NTU above background).

From the information currently available, it is clear there are gaps in the general understanding on the effects of increased turbidity on fishery resources from offshore sand mining operations, and sampling strategies and methods need to be developed, when appropriate, to address these data gaps. Based on limited information specific to the conditions typical of the OCS, and mostly extrapolated from inherently different setting (estuarine and nearshore), it appears that increased suspended-solid concentrations may not persist for a sufficient amount of time to cause moderate or severe adverse effects. In some OCS environments, where the grain size distribution is >95-98% fine to medium sand, turbidity plumes are often not observed at the surface, except during overflow operations. In the absence of empirical data on relevant species exposed to typical turbidity occurring during offshore sand mining operations, efforts could be made to collect information on the concentration of suspended solids and duration of these events typical of OCS dredging operations, so that assessments can be made based on known responses from studies with nearshore species.

4.3.2.3 Increased sedimentation and deposition of fines

Benthic fishes and mobile macroinvertebrates (e.g., crabs and shrimps) encounter some degree of sedimentation under natural conditions, and most can migrate vertically through substantial overburdens. However, their tolerance to increased sedimentation and overburden beyond natural levels is largely unknown. One of the main risks associated with sand extraction is the redeposition of sediment within the extraction site, which can cause smothering of demersal fish eggs on spawning grounds, suffocation of filter-feeding benthos, and smothering and suffocation of crabs, which become lethargic while brooding (ICES 1992). However, the magnitude of effects from sediment redeposition depends largely on the amount of sedimentation, the nature of the local fauna, and their tolerance to increased sedimentation. Species particularly sensitive to smothering include fish species with demersal eggs that remain on the bottom until larval hatching. The sand eel, for example, is an important fishery resource of Denmark and the U.K. and a non-migratory species that plays important ecological roles (e.g., key prey for gadids). This species lays eggs on sand and, when covered with fine material generated during dredging and sorting, embryo development can be disrupted and hatching success reduced (ICES 1992). Similarly, herring (*Clupea harengus*) has a relatively small spawning ground and lays demersal eggs on gravel. Because of high site association and suspected fidelity of spawning grounds, this species can be directly affected by marine gravel mining, particularly in the English Channel (ICES 1992). Concerns have also been raised regarding the potential effects of increased sedimentation on the spawning habitat of winter flounder (*Pseudopleuronectes americanus*), which occurs mostly within state waters from the Middle Atlantic to the Gulf of Maine. In laboratory studies, very few recently spawned flounder eggs, which are benthic and adhesive, hatched when buried under >3 mm of sediment, and delayed hatching was observed in eggs buried under ≥1 mm of sediment (Berry et al. 2011). Numerical modeling was used to investigate the potential egg burial from elevated sedimentation in shoals of Newark Bay subject to 38,230 m^3 dredging material (Lackey et al. 2009). Under normal dredging conditions, little sediment was likely to deposit on the winter flounder spawning habitat, and the maximum estimated deposition (0.03 mm) was below the egg burial screening level (0.55 mm, or ½ of an egg's diameter) (Lackey et al. 2009). Because of the susceptibility of this species to increased sedimentation, mitigation strategies are typically put in place to avoid dredging in areas that represent important spawning grounds, or to limit the extraction volumes during the spawning season (seasonal windows) of this and related species. However, the effectiveness of these measures has not been completely resolved (Office of the Deputy Prime Minister 2005).

While dredging of borrow areas and material rehandling have the potential to alter the bottom habitat, conversions between sedimentary and hard bottoms are unlikely outcomes of the processes involved at most OCS sand borrow areas. Although there is a potential for temporary conversion of hard bottom to sedimentary habitat, caused by deposition of fine sediments suspended during the dredging process, the currents that prevented natural sedimentation would eventually remove the sediments. Habitat conversions can also occur in the opposite direction if most of the sand from a location on top of a pavement of hard rock is removed. In addition, sand extraction can lead to exposure of large rubble boulders and limestone debris, which can attract fish use, as demonstrated on the slopes of borrow pits created in 1972 off Hillsboro Beach in south Florida (Marsh and Turbeville 1981).

Lindeman and Snyder (1999) indicated that early life stages (newly settled, early juvenile, and juvenile) of fish species typically associated with hard bottoms (particularly cryptic species) may succumb to burial because of their low mobility. Potentially greater impacts from burial may occur prior to and during peak periods of larval recruitment (spring and summer). In their assessment, a hard-bottom habitat was impacted by dredge fill placement from a nearshore sand dredging operation and had a significant reduction in species abundance compared to controls. Furthermore, impacts from dredging on hard-bottom habitats are likely great because density-dependent processes (e.g., space and food availability) may limit the ability of displaced species to occupy nearby hard-bottom habitats. However, OCS dredging operations typically implement buffer zones, which are sites-specific, to minimize impacts on hard-bottom habitats.

Although turbidity and increased sedimentation from dredging has been shown repeatedly to have lethal and sublethal impacts on reef corals (Dodge and Vaisnys 1977; Bak 1978), even deaths of those corals would not convert the habitat type into a sedimentary seafloor. Nevertheless, loss of living corals could readily affect the habitat quality and its use by invertebrates and fishes. Lindeman and Snyder (1999) demonstrated by careful sampling before and after beach nourishment at Jupiter in south Florida how important nearshore shallow reef habitat is for fishes. Sand redistribution from the nourished beach buried hard-bottom habitat (half of which contained *Phragmatopoma lapidosa* polychaete reefs), reducing the number of fish species present (from 54 to 8) and their abundance (from 38 to <1 individuals per transect) after burial of the hard bottom. A large majority of these fishes was comprised of recently settled juveniles, indicating the importance of such nearshore reefs as a fish nursery. Depending upon depth of sedimentation and potential for erosion of those sediments covering the hard substrate, reef habitat damage from sedimentation may ultimately recover. Courtenay et al. (1980) resurveyed a 2.5 km^2 area of hard-bottom reef in 8-13 m water depths off Hallandale, Florida that had been damaged seven years previously by sedimentation originating from sand dredging 130-220 m away, revealing no sign of the extensive coral damage. Courtenay et al. (1980) also resurveyed the fish community on this reef at 8-13 m depths and on the nearshore reef at 5-7 m depths. The inner reef exhibited consequences of sedimentation by fine particles, presumably arising from erosion and transport from the dredged materials placed on the beach. The most evident consequence of this habitat modification on the inner reef was the complete loss of a fish that was common in an earlier pre-nourishment survey, the dusky jawfish. Courtenay et al. (1980) suggested that the deposition of fine sediments converted the soft bottom into materials that would not sustain the burrow structures this species makes into sedimentary bottom adjacent to hard reef. Although this particular example involves deposition of sediments derived from the beach, analogous sedimentation from rehandling operations have the potential to induce the same effects on sand burrowing fish species. However, the effects associated with rehandling activities in nearshore areas may be difficult to distinguish from natural sedimentation or sedimentation from episodic storm events, unless careful studies are designed to address and control for environmental sources of disturbance. Monitoring reports for the Miami Dade Southern Government Cut (MDCDERM 2010, 2012), where hard-bottom habitat (characterized by octocorallia, sponges, and scleractinians) was located in proximity of the borrow area (see Figure 2.11), found light to moderate sedimentation and signs of coral stress in surveys post-dredging, though these areas are episodically covered with sand remobilized by natural processes. These monitoring efforts did not include assessments of potential impacts on fish species or populations.

Although there are no mitigation strategies aimed at reducing impacts of increased sedimentation and deposition of fines on fish and large motile invertebrates, site-specific considerations should be taken into account during the planning of sand mining activities. A number of mitigation strategies that protect hard bottom habitat (and consequently fish and large invertebrates) from this impacting mechanism were discussed in detail in Section 4.2.2.3. Despite concerns for smothering of fish eggs on spawning grounds for key species, few quantitative data are currently available on egg survival and hatching success relative to the sedimentation levels typically observed following OCS sand dredging, which also need better characterization. A better understanding of the potential species at risk and their habitat utilization (spatial and temporal), combined with information on habitat requirements and tolerances (e.g., egg tolerance to overburden and survival under different sedimentation regimes) and modeling efforts (as demonstrated by Lackey et al. 2009) may provide the data needed for more quantitative assessment of potential impacts associated with increased sedimentation.

4.3.2.4 Water Quality

Aside from increased suspended-solid concentration (discussed in Section 4.3.2.2), one of the main concerns related to water quality during and after dredging operations is the potential for decreased dissolved oxygen in the water column and the water-sediment interface particularly when depressions are created by these activities. An example of these depressions is depicted in Figure 2.12. Nairn et al. (2007) stated that most fish, including all sharks, are highly mobile and can avoid impacts from dredging, including the avoidance of low-oxygen areas. Fish and invertebrates without sufficient mobility (e.g., cusk eels, tonguefishes) may not be able to avoid bottom hypoxia, but may be capable of tolerating such conditions for short periods of time (i.e., <1 day). Significant bathymetric changes and the formation of pits and furrows have the potential to cause a drop in current strength and water flow, resulting in lower oxygen exchanges and increased deposition of finer sediments, which may contribute to localized depletion of oxygen. Oxygen depletion is also more likely in deep pits or furrows with marked water stratification. Low dissolved oxygen can be particularly problematic in spawning grounds; however, oxygen depletion is more likely to occur during the dredging of substrates containing highly organic material, or after dredging creates deep furrows or pits with poor water circulation. However, dredging of substrates with high organic content is uncommon as these often do not meet grain size and percent fines requirements. By contrast, dredging activities that leave undisturbed high ground topographic features of shoals can project benthos fishes and large motile invertebrates high enough into the water column and into the upper mixed-zone stratum where anoxia does not develop. A study by Gelpi (2012) showed that Ship Shoal, Louisiana, an area surrounded by seasonally hypoxic waters (Rabalais et al. 2001b), has fairly high dissolved oxygen concentrations (5-6 mg/L) during most of the year providing an hypoxia refuge for important species including blue crabs. Furthermore, the high relief of this and similar nearby shoals (Trinity and Tiger) provide important spawning and foraging habitat for blue crabs. Consequently, sand mining could negatively impact the fecundity of blue crabs not only by reducing prey abundance, but also by lowering the topographic relief of these shoals and reducing the ability of surface wave action to oxygenate the water column, and facilitating the intrusion of hypoxic waters (Gelpi 2012). The data compilation by Rabalais et al. (2001a) indicated that nekton and demersal and benthic fishes and invertebrates off the southeastern Louisiana shelf generally exhibit signs of stress and even mortality when exposed for extended periods to oxygen concentrations in the 0-2 mg/L range. This study (Rabalais et al. 2001a) also highlighted individual species limits to low oxygen concentrations, with infaunal species exhibiting a higher tolerance than epibenthic invertebrates, while larger demersal invertebrates and fishes exhibiting avoidance of water masses with low oxygen concentrations.

Accidental spills of light refined oils in offshore areas are expected to rapidly disperse by natural weathering processes (as discussed in Section 3.3.7). In terms of toxicity to water-column organisms, diesel is considered to be one of the most acutely toxic oil types. Fish and invertebrates that come in direct contact with a diesel spill may be killed. However, small spills in open water are so rapidly diluted that fish kills have never been reported. Fish kills have been reported for small spills in confined shallow water. During the spill of an estimated 8,000 liters of an intermediate fuel oil from the dredge *Stuyvesant* 1-6 km off Humboldt Bay, California, water-quality models were used to estimate that oil concentrations above toxicity thresholds were limited to the upper 2 m of the water column (California Department of Fish and Game (CDF&G) et al. 2007) Thus, near-surface fishes and shrimps would be most at risk of exposure to toxic levels.

Mitigation strategies aimed at reducing the impacts associated with water quality include avoiding sand dredging and operations that create deep anoxic pits within the borrow area. The formation of pits and the potential impacts on localized water quality during OCS sand dredging activities is of great concern. Consequently, some OCS projects specifically state the maximum sediment depth of dredging activities. Examples include borrow excavation not to exceed a depth of 2.4-3 m over the life of the borrow area (USACE 2011b), uniform dredging over a wide area of each shoal not to exceed 3 m in depth (USACE 2008a, Annex E), and uniform dredging avoiding the formation of pits greater than 2 m in depth (NASA 2010c). Although other mitigation strategies may also include the implementation of pollution control plans, site-specific considerations (e.g., Gelpi 2012) should be taken into account during the planning of sand mining activities.

4.3.2.5 Entrainment near the seafloor

Entrainment occurs when organisms, particularly sedentary or slow-moving fauna and benthic eggs, larvae, and fish, are removed from the top layer of the seabed by the suction field generated during dredging. Both demersal and pelagic fish eggs and larvae (e.g., Atlantic herring, and sand lance) are susceptible to entrainment by suction dredges, a concern noted by the ICES Working Group (ICES 1992).

Little work has been carried out regarding entrainment rates of fishes and large mobile invertebrates during OCS dredging operations or their survival rates post return through the overflow process, and most of the currently available information comes from nearshore and shallow-water operations (La Salle et al. 1991; Reine and Clarke 1998; Nightingale and Simenstad 2001). A literature review in the late 1990s (Reine and Clarke 1998 and references therein) compiled entrainment rates for a variety of species during dredging of estuarine and riverine sites with hopper, pipeline, and clamshell dredges (Figure 4.4).

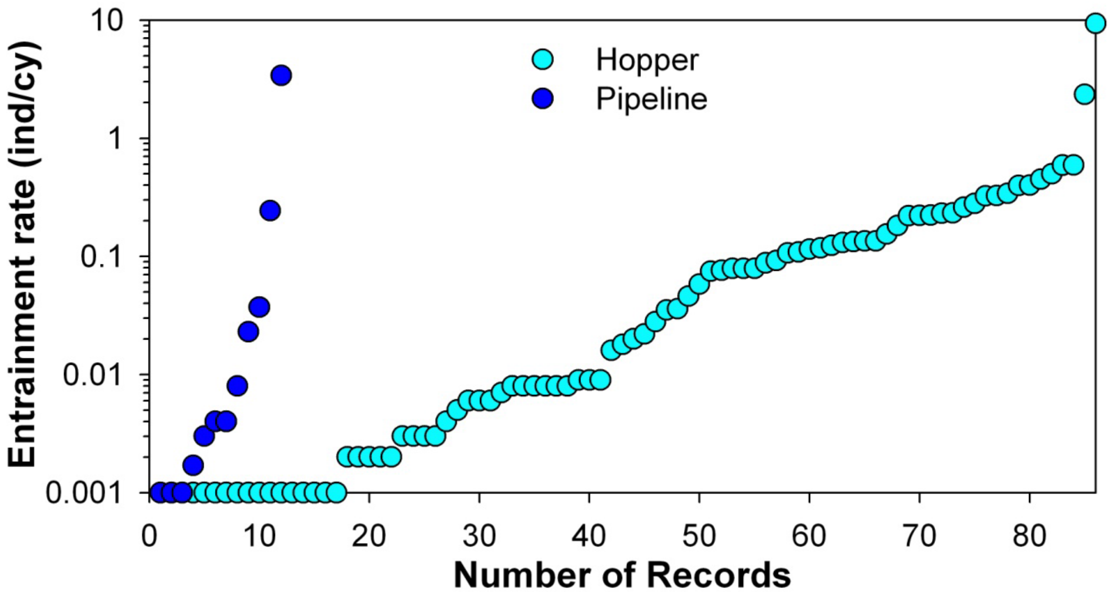


Figure 4.4 Reported entrainment rates (individuals/cubic yard of extracted material) for hopper and pipeline dredges from dredging operations in estuarine and nearshore environments (data from Reine and Clarke 1998 and references therein). These rates are not corrected for the abundance of individuals in the water column prior to dredging operations. Note that it is currently unknown if these entrainment rates occur during dredging operations on the OCS.

Several factors appeared to have influenced entrainment rates including bottom depth, hopper dredge speed or cutterhead advance rates, flow-field velocities generated at the draghead or cutterhead, volume of dredged material, and direction of dredging in relation to tidal flow. Entrainment rates were particularly high in hopper and pipeline dredges compared to mechanical dredges, because hydraulic dredges create a suction field that is unavoidable by many organisms (Reine and Clarke 1998). For Dungeness crab, adult mean entrainment rates for hopper dredges (all in estuarine or riverine sites) ranged from 0.040 to 0.592 crabs/yd^3 of dredged material, while juvenile crabs were entrained at a significantly higher rate (range 0.32 to 10.78 crabs/yd^3). Mortality during hopper dredging increased with increasing size from 5% for smaller crabs (7-l0 mm) to 86% for larger crabs (>75 mm) (Larsen and Laubek 2005 in Reine and Clarke 1998). For sand shrimp (*Crangon* spp.) entrainment rates by hopper dredges ranged from 0.063 to 3.38 shrimp/yd^3 of dredged material (Armstrong et al., 1982 in Reine and Clarke 1998). Fish entrainment rates, regardless of fish size, ranged from 0.001 to 0.135 fish/yd^3 for both pipeline and hopper dredges, with a mortality rate of 37.6% (Armstrong et al., 1982 in Reine and Clarke 1998). La Salle et al. (1991) also indicated that the only species consistently entrained at moderate levels (0-18.89 fish/yd^3) was the bottom dwelling sand lance (*Thaleichthys pacificus*). Whether or not these entrainment rates and effects occur during dredging operations on the OCS is not currently known, and cannot be adequately evaluated with the existing information.

However, none of these studies reporting entrainment rates discussed the influence of the relative abundance by species and/or life stages on their assessment of entrainment risks. Nightingale and Simenstad (2001) noted that, within U.K. waters, demersal species (e.g., sand lance, sculpins, pricklebacks, flatfish, and sand eels) were most likely to be entrained by aggregate dredgers. Similarly, overwintering crabs are likely to be susceptible to direct uptake, as they typically exhibit low activity during overwintering periods and would be unable to avoid a dredge suction field. By contrast, Van Dolah et al. (1992) estimated the mortality of postlarval shrimp from entrainment at a site offshore of Hilton Head Island, South Carolina, (~1,883 shrimps/day) and concluded that, given the high reproductive output of female shrimp (1 million eggs per spawn), the number of entrained shrimps was considered negligible. The same study concluded that the likelihood of adverse impacts from entrainment by dredges of economically and recreationally important planktonic species, including early life stages of shrimps, crabs, and fishes, was low given their low relative abundance. A recent study on the projected entrainment rates of benthic fish by hopper dredgers at a sand and gravel dredging site in the U.K. Drabble 2012) found rates (based on species, site-specific dredging assumptions, and monitoring data) as high as 0.02/m^3 for bib (*Trisopterus luscus*). This study identified species traits that increase their susceptible to entrainment, including poor sensitivity to dredge noise, limited avoidance reaction, low burst speed, high burial behavior, and low fecundity. Although the author acknowledged that even though population level impacts are generally perceived to be low, entrainment surveys should be integrated into monitoring strategies during marine aggregate dredging.

A synthesis sponsored by the Marine Aggregate Levy Sustainability Fund (MALSF) points to the lack of information on entrainment and survival rates of fish from marine aggregate extraction operations (Tillin et al. 2011). Only one study (Lees et al. 1992) looked at the outwash of the suction trailer dredger *Arco Tyne* from aggregate extraction in the English Channel. This work found that some components of the fauna, including most fishes and small crustaceans, appeared physically undamaged and would have been washed back to sea, though the study did not assess their subsequent survival. By contrast, worms and many crustaceans appeared to be highly susceptible to physical damage from dredging operation, and heavily shelled, mobile macroinvertebrates species (hermit crabs) would likely die because they are retained within the hopper (Lees et al. 1992).

Based on the available information on entrainment rates, the most sensitive resources are demersal fishes and eggs; larger demersal and pelagic juvenile and adult finfishes are likely to avoid dredging areas during operations. Entrainment and survival rates are largely unknown, representing a data gap in our understanding of the impacts of offshore sand mining on fishery resources. No mitigation strategies specific for fish and large motile invertebrates were identified for this impacting mechanism.

4.3.2.6 Sound

BOEM recently sponsored a workshop to help identify critical information needs and data gaps on the effects of sound generated by the energy industry on fish, fisheries, and invertebrates, with focus on the U.S. Atlantic and Arctic OCS (Normandeau Associates Inc. 2012). Although key elements of that work are briefly mentioned in the synthesis below, the reader should refer to the original work for details.

Processes associated with marine sand extraction contribute to increased sound levels above background and have different sound characteristics that vary depending on operational (i.e., dredge type, propulsion power) and environmental factors (i.e., seabed type, depth) (DEFRA 2003; Thomsen et al. 2009; Saunders and Roberts 2010; Tillin et al. 2011). Key sources of sound include pump driving, transport, deposition, draghead movement over the seabed, and sound generated by ships and machinery (DEFRA 2003).

The majority of fish species do not have known hearing specializations and only detect sounds in the 500-1,000 Hz range, with the best hearing acuity from 100 to 400 Hz. Hearing-specialist fish (fishes with specialized structures that enhance hearing sensitivity and bandwidth, thus having a broader hearing range, e.g., goldfish, catfish) can detect sounds up to 3,000 Hz (best hearing from 300-1,000 Hz), while some clupeids (American shad, menhaden, blueback herring, alewives) can detect ultrasonic frequencies of 200 kHz (Popper and Hastings 2009) (Figure 4.5). Because clupeids have hearing structures (prootic auditory bullae), these species are highly sensitive to acoustic stimuli and are most likely to be impacted by increased/loud ambient sound compared to non-hearing specialists species and species without swim bladders (e.g., plaice, dab, sculpin), which exhibit moderate and low sensitivity to sound, respectively (Nedwell et al. 2004). Recent work (reviewed in Normandeau Associates Inc. 2012) indicated that fish species may also be sensitive to particle motion fields in sound detection. Because the frequency and sound levels emitted by dredging overlap the frequency spectrum and the bandwidth of the hearing of some fish species (see Section 3.3.5), at a close range, dredging has the potential to impact fish by inducing adverse behavioral reactions and causing physiological damage (Popper and Hastings 2009; Thomsen et al. 2009).

As highlighted by Popper and Hastings (2009), disruption of a fish's ability to detect biologically relevant signals has the potential to cause deleterious effect on the survival of fish and the health of fish populations. Sound can influence fish behavior (e.g., feeding or reproductive behavior of some reef species), and it has been hypothesized to interrupt migrations (USDOI MMS 2009; Thomsen et al. 2009). Normandeau Associates Inc. 2012) also indicated that behavioral changes induced by sound may have major effects on fish populations; however, these are more likely the case when fish are exposed continuously to an intense sound source at levels well above ambient noise. Some pelagic reef fish larvae have shown to respond to sound stimuli as a navigational cue to locate reef habitat (Tolimieri et al. 2000); therefore, alterations of background sound may impair their ability to locate optimum settling substrates. Similarly, sound also plays an important role in fish communication, including sound produced during agonistic encounters (protection or establishment of territories) (Myrberg 1981; Ladich and Myrberg 2006), courtship, and spawning (Myrberg 1981; Myrberg and Lugli 2006). For example, members of the Sciaenidae family (drums and croakers) use sound for communication, such as in the case of adult male weakfish (*Cynoscion regalis*), which makes specific drumming sounds (127 dB re 1 µPa) that attract females to spawning grounds (Luczkovich et al. 1999). Consequenttly, increased noise above normal backgrould levels has the potential to mask and interfere with social forms of interaction and communication.

High-intensity sounds can cause hearing loss and permanently damage fish hearing and can disrupt swim bladders (Nightingale and Simenstad 2001; DEFRA 2003; USDOI MMS 2009), although fishes without swim bladders can also suffer tissue damage at 180 dB re 1 µPA (DEFRA 2003). Studies have also indicated that sound increases above background levels and sudden changes in sound pressure can lead to stress in many fish species (Popper and Hastings 2009 and references therein), increasing their vulnerability to predation.

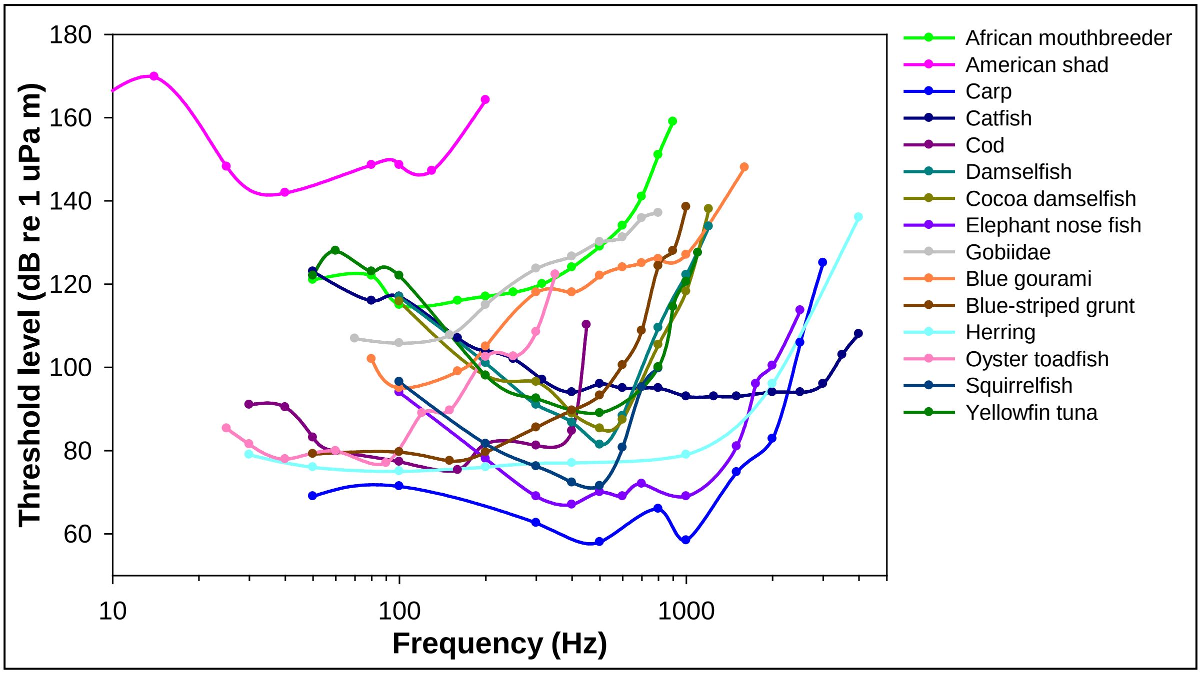


Figure 4.5 Audiograms of selected fish species (data from Nedwell et al. 2004 and references therein). Sound levels generated during OCS operations were 161.3-178.7 dB re 1 µPa at 1 m from the source, with peak frequency in the 80-3,000 Hz range. From Reine et al. (In prep).

Audiograms analyzed by DEFRA (2003) showed that clupeids are sensitive to acoustic frequencies in the 10-1,000 Hz range, with a maximum sensitivity to acoustic stimuli of 75 dB re 1 µPa at 1 m at 200 Hz. Sound levels 90 dB above the maximum sensitivity to acoustic stimuli for 20 minutes could lead to a temporary hearing loss, while sound level 110 dB above the maximum sensitivity to acoustic stimuli would cause a permanent loss of some hearing capability. As noted by Popper and Hastings (2009), the amount of hearing loss is related to the level of the sound above a species' hearing threshold, or the lowest sound detectable. At the frequency of maximum sensitivity (200 Hz), a clupeid would experience temporary and permanent hearing loss at sound levels of 165 dB re 1 µPa and 185 dB re 1 µPa, respectively, when exposed continuously over a 20-minute period. Analysis of sound recorded from U.K. dredgers showed that at a distance of 50 m the majority of the sound energy was less than 1,000 Hz range, decreasing to undetectable levels at 500 m from the dredger. Under full dredging activities measured sound levels ranged from 117 dB at 200 Hz to a maximum sound levels of 126 dB occurring at 400 Hz. Species studied by DEFRA 2003) (e.g., salmonids) would be aware of the dredging activities, which in turn can impact their behavior, whereas flatfish (e.g., plaice and dab) would be sensitive to low levels of seabed vibration. However, the sound produced by offshore dredging operations (typically in the range of 168-186 dB re 1 μPa) (OSPAR 2009) fall below the permanent threshold levels for clupeids (DEFRA 2003) (Figure 4.6).

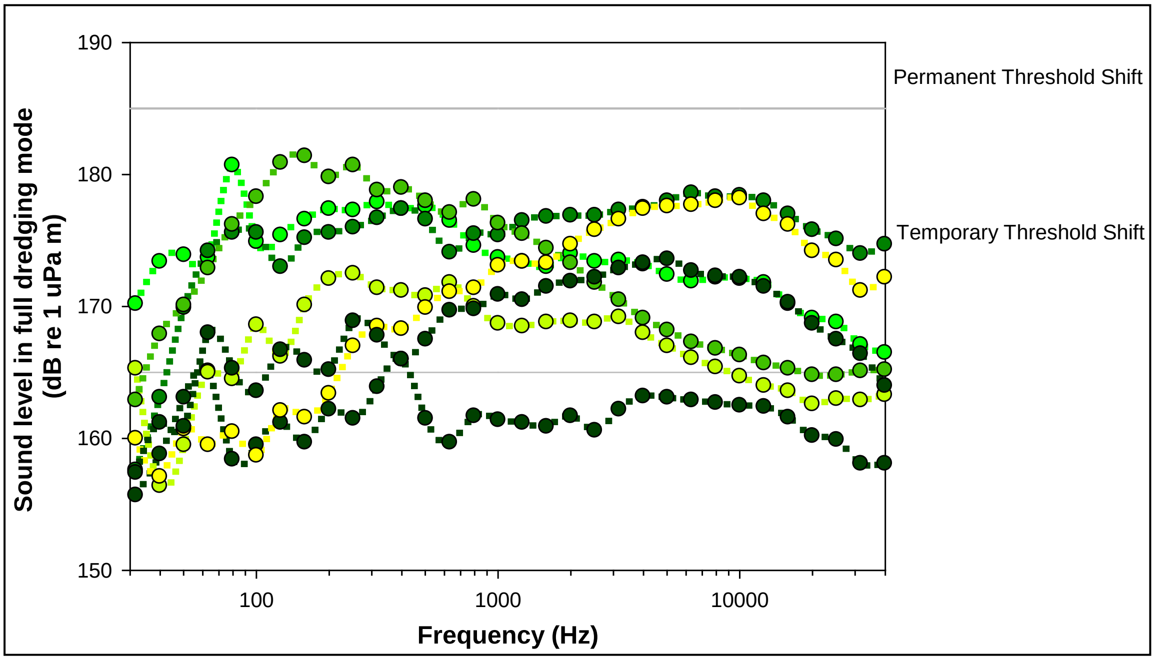


Figure 4.6 Sound generated by dredges during different aggregate extraction operations in the U.K. (green shade dots) (data modified from Robinson et al. 2011) in relation to the range of sound that would cause temporary (blue area) and permanent (red area) threshold shifts in clupeids when exposed continuously (at least 20 minutes) and at a short distance (a few m) from the source (data from DEFRA 2003). Note that the maximum sensitivity of clupeids to acoustic stimuli was reported at 75 dB in the 200 Hz frequency range (DEFRA 2003), and that hopper dredges can generate sound in the 70-1,000 Hz frequency range, with a maximum sound pressure level of 186 dB re 1 μPa at 1 m. From Thomsen et al. (2009).

Sound levels generated by three dredges during various phases of dredging (transit, sediment removal, pump out) of an OCS sand shoal, during the Wallops Island, Virginia project (Reine et al. In prep), were in the 161.3-178.7 dB re 1 μPa at 1 m range (peak frequency range 80-3,000 Hz), which overlaps the audiogram of the American shad shown in Figure 4.5. As a reference, the criteria for marine mammal behavioral disturbance/harassment from a continuous sound source (e.g., dredging) is currently 120 dB rms @ 1 μPa (see Section 4.5.2.4).

Other studies have also reported temporary hearing loss from fish exposure to sound. Fathead minnows experienced temporary hearing impairment following exposure to 142 dB re 1 μPa across frequency bands from 19-15,000 Hz (peak frequency 13,000 Hz) for 2 hours (Scholik and Yan 2002); whereas goldfish (*Carassius auratus*), a hearing specialist fish, exhibited physiological stress in response to sound, had lower hearing threshold after ~10 min exposure (160-170 dB re 1 μPa broadband 0.1-10,000 Hz), and had a maximum temporary hearing loss occurring within 24 hours of exposure. The work by Smith (2004) also indicated that fishes with lower baseline audiograms (hearing specialists) are more susceptible to sound-induced hearing loss, but that exposure to intense anthropogenic underwater sound may not cause permanent physiological injury or hearing loss.

As pointed out by the Normandeau review (Normandeau Associates Inc. 2012) the development of sound threshold criteria, for both behavioral and physiological responses protective of most fish populations, is imperative to support OCS operations; however, the development of these criteria faces great challenges, including the current scientific state of knowledge and the lack of carefully and rigorously designed experiments. To date, the interim criteria for the onset of physiological effects, which have been received with great resistance, are (see Normandeau Associates Inc. 2012 for details):

- Zero to peak sound pressure level: 206 dB re 1 μPa
- Cumulative sound exposure level from exposure to multiple sources: 187 dB re $1\mu Pa^2 \cdot s$ for fishes >2 grams and 183 dB re $1\mu Pa^2 \cdot s$ for fishes <2 grams

From the limited empirical evidence available to date, increased sound levels have the potential to induce behavioral (e.g., site avoidance) and physiological (e.g., temporary or permanent loss of hearing) changes (Popper and Hastings 2009). Assuming that fishes and other marine resources have similar sound sensitivities, and considering that sound generated by dredging may be audible to fishes over distances up to several kilometers from the source, injury due to underwater sound from OCS operations is unlikely to result in major impacts on fishes. Furthermore, based on the information currently available, dredging operations generally produce low levels of sound energy that are of short duration; therefore, the impacts of underwater sound on fish populations are expected to be temporary and localized. However, more research is required before the effect of dredging sound on fishes can be fully assessed (Thomsen et al. 2009; Nightingale and Simenstad 2001; Normandeau Associates Inc. 2012, and references herein). Although studies have documented behavioral and lethal effects from pile-driving sound (see OSPAR 2009), the conclusions are tenuous at best, given potential issues with experimental designs, an issue that applies to the effects of sound from seabed mining. While offshore sand dredging likely contributes to the overall anthropogenic ocean sound (see Section 3.3.3.5), little information exists on the effects of elevated sound levels above background on fishes, invertebrates, and developing eggs and larvae. No information is currently available on the effects of substrate vibrations during dredging on bottom dwelling flatfishes. Furthermore, extrapolations of the observations from the limited laboratory studies to fishes in the wild are not practical and, consequently, the impacts of sound on fishes from OCS operations including dredging constitute a data gap.

No mitigation strategies were identified with regards to sound. However, passive listening (similar to those designed for marine mammals) to detect the presence of vulnerable species may be an important mitigation strategy, along with planning the timing of OCS operations to avoid them (Normandeau Associates Inc. 2012).

4.3.2.7 Vessel operations and interactions

Relevant studies were not identified on the effects of offshore vessel operations and interactions with fishes and large mobile invertebrates. However, a handful of studies (Morgan et al. 1976; Holland 1986; Killgore et al. 1987; Killgore et al. 2001) have simulated the impact of shear stress or turbulence (a function of propeller speed) common of rivers and waterways, which are probably worst-case scenarios of impacts on fishes compared to those resulting from

OCS operations. Others (Gutreuter et al. 2003; Killgore et al. 2011) have also looked at the entrainment rates by propellers in river systems.

Laboratory studies with early life stages of five fish species (Killgore et al. 2001) reported increased mortality with increased shear stress (range tested: 634-4,743 dynes/cm^2) from propellers, with mortality being more likely a function of developmental stage (larvae more sensitive than eggs) and life-stage size (larvae <10 mm more susceptible). This study indicated that reducing boat traffic and speed during peak periods of larval fish abundance may be a practical way to reduce mortality. Killgore et al. (1987) also showed that high shear stress (6,320 dynes/cm^2) caused 80-87% mortality of paddlefish larvae compared to 3-13% mortality under low shear stress (1,838 dynes/cm^2), while Morgan et al. (1976) suggested that fish egg and larval mortality increases with shear force and time of exposure. In the channel of the upper Mississippi River, Holland (1986) found reduced ichthyoplankton catches and increased fish egg damages after the passage of loaded barges. Vessel interactions with fish during OCS operations are likely insignificant when compared to those of vessels in rivers and waterways, as the area of operations is less restricted, and the water column depth much greater (>5 m), limiting the impacts associated with propeller shear stress.

Work by Brown and Murphy (2010) indicated that vessel strike mortalities, particularly of female Atlantic sturgeon (*Acipenser oxyrinchus*) in the Delaware estuary, may be detrimental to the long-term viability of the population. This study recommended the implementation of reduced vessel speed during this species' spawning season for vessels transiting through spawning areas to reduce vessel strike mortalities. This species may occasionally inhabit sand shoal areas (particularly in offshore areas between the New York and the Middle Atlantic Bights), and therefore, interactions with this species are possible, but unlikely.

Although there is no specific information available on vessel interactions with fish during OCS operations, any interactions may be similar to those experienced with other types of offshore vessels, but likely less detrimental than those documented in river channels and waterways. No mitigation strategies specific for fish and large motile invertebrates were identified for this impacting mechanism.

4.3.2.8 Exposed UXO, shipwrecks, and other hard structures temporarily exposed during dredging

Relevant studies were not indentified on the effects of exposed UXO, shipwrecks, and other hard structures temporarily exposed during dredging on fishes and large mobile invertebrates. These structures could to provide temporary habitat to species common of hard-bottom substrates, including demersal and semi-pelagic species, and they could promote colonization opportunities for epibenthic assemblages, which in turn provide feeding opportunities for several fish species.

4.3.3 Summary of Known Impacts on Fishes and Essential Fish Habitat due to OCS Dredging and Data Gaps

Given the available information, some general conclusions can be drawn regarding the potential impacts of OCS sand mining on fishery resources. Overall, potential impacts may arise from offshore sand mining activities on fish, but most observations are based on the ecological and biological understanding of the link between stressors and responses, on laboratory tests and studies with limited applicability to field conditions, or on studies mostly derived from nearshore and estuarine assessments. Relative to these sources of information, much less knowledge has been gained from field data collection efforts during OCS mining activities. Furthermore, conclusively identifying the effects on fishes and large motile invertebrates (e.g., crabs and shrimps) from OCS sand mining and related activities is complicated by the myriad of factors that influence the severity of adverse biological responses to stressors (e.g., seasonality, species interactions, other sources of stress), and the large variability of spatial and temporal baseline conditions. The following sections summarize what is known about the potential impacts and effectiveness mitigation measures by impacting mechanism, discussed in order of greatest to least impact: sediment and biota removal, increased sedimentation/deposition of fines, entrainment, sound, and water quality. The other potential impact mechanisms (vessel operations and interactions and UXO and temporary exposure of hard structures) are considered to have minimal to inconsequential impacts or have no available data to discuss.

Of all major impacting mechanisms evaluated here, alteration of benthic habitat (Table 4.10) has the potential to cause the most detrimental direct, indirect, and cumulative impacts on fishes and large motile invertebrates, and is a commonly cited source of concern in EFH assessments and NMFS conservation recommendations. Localized alteration of benthic habitat by dredging could reduce habitat viability, displace and/or remove the native community, cause loss of eggs and larvae of demersal species, and disrupt predator/prey interactions. These concerns are warranted and environmental evaluations of OCS sand dredging sites review these impacts to fishes and large motile invertebrates. However, cumulative impacts have not been adequately assessed, and the relative lack of scientific peer-reviewed information suggests that this constitutes a major data gap. Similarly, the lack of carefully designed studies that address natural variability and other confounding factors has not allowed a quantitative assessment of the implications of habitat alterations on entire populations. However, population traits that may influence the severity of impacts from sand dredging may include: species with spawning grounds restricted to sand shoal habitats that also exhibit narrow spawning windows and low fecundity, and species with critical life stages restricted to sand shoal habitats. To err on the side of caution, mitigation strategies in the U.K. recommend the implementation of environmental windows (seasonal and spatial) and exclusion zones, though their effectiveness has not been confirmed.

Impacts from increased sedimentation and deposition of fines (Table 4.11) could also cause direct impacts, primarily on species sensitive to smothering, including species with demersal eggs that remain on the bottom until larval hatching (e.g., sand eel, herring, flounders). However, species-specific tolerances to increased sedimentation beyond natural levels, and egg survival and hatching success relative to the sedimentation levels typically observed following OCS sand dredging, are largely unknown. Precautionary measures, particularly in areas with high congregation of sensitive species, may include the careful selection of rehandling areas.

Table 4.10

Impacting mechanism for OCS dredging on fishes and essential fish habitat: *Alteration of benthic habitat at the borrow area.*

Impact Pathway	*Potential Effects on Fish*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☒	Direct Effects Alteration of foraging and spawning grounds Indirect Effects Altered forage base for species that feed on benthos in the borrow area; Abandonment of foraging areas Cumulative Effects Reduced fitness due to reduced foraging and less successful reproduction	Impacts likely confined to the footprint of activities. Effects may extend to nearby areas where density-dependent exploitation of the foraging grounds could lead to reduced feeding.	Recovery begins after cessation of activity, but impacts on loss of spawning grounds may persist depending on species, and recovery rate of topographic features by local sediment dynamics. Impacts on food source availability may persist depending on the recovery rate of the benthic prey and their substrate[1], but most species would likely find prey in nearby undisturbed areas. Fish displacement may lead to density-dependent competition leading to reduced feeding.	Unknown for specific shoals. There are plans for repeat dredging of shoals at five-year intervals.	The impacts may be higher for species with limited spawning grounds and spawning occurring during extraction activities[2]. There is limited information on the impacts of food availability disruptions, and most of the available studies have reported minimum impacts, possibly resulting from the lack of rigorous assessments.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Studies are available on seabed impacts and on the impacts on benthic invertebrate assemblages, but little information exists on the impacts of mining on foraging and spawning grounds.	Restricted seasonal and spatial mining periods, and establishment of exclusion areas[3, 4,5] adequate in size, thus provide refuge for fish species that may promote recolonization of the borrow area. Measures to speed recovery of benthic communities and habitats would reduce potential impacts to fish; leaving undisturbed patches of habitat, and limiting activities to non-critical spawning periods would be beneficial to fishes and large motile epifauna. Dredging guidelines to ensure the long-term habitat functions of offshore sand ridge and swale complexes, including a maximum dredging limits of the total shoal volume[6], avoidance of pit formation[6,7,8], maintaining the relief of the seafloor[6], implementation of safe distances from EFH[9, 10], and implementation of monitoring programs[9,11,12], sand mining targeting areas on the shoal that are accreting, while avoiding erosional areas, and portions of the crest[7,9], other measures that maintain the integrity of the shoal[13].		Seasonal restrictions and environmental windows not definitively confirmed by monitoring data[3,4]. Unknown at this time.	

[1]Diaz et al. 2004; 1992; [3]Sutton and Boyd 2009; [4]Office of the Deputy Prime Minister 2005; [5]NMFS and similar agencies in response to EFH consultations; [6] U.S. Army Corps of Engineers (USACE) Jacksonville District 2003a , Annex E; [7]NASA 2010c; [8]U.S. Army Corps of Engineers (USACE) 2011b; [9]U.S. Army Corps of Engineers (USACE) and Minerals Management Service (MMS) 2009; [10]U.S. Army Corps of Engineers (USACE) Jacksonville District 2011b;[11]U.S. Army Corps of Engineers (USACE) Jacksonville District 2011a; [12] U.S. Army Corps of Engineers (USACE) 2011a; [13]Dibajnia and Nairn 2011

Table 4.11

Impacting mechanism for OCS dredging on fishes and essential fish habitat: *Increased sedimentation and deposition of fines.*

<table>
<tr><th>Impact Pathway</th><th>Potential Effects on Fish</th><th>Spatial Extent of Impact</th><th>Duration of Impact</th><th>Frequency of Impact</th><th>Severity of Impact</th></tr>
<tr><td rowspan="3">Direct:
☒
Indirect:
☒
Cumulative:
☒</td><td>Direct Effects
Burial of benthic eggs on spawning grounds
Indirect Effects
Burial of prey sources important for many juvenile fishes
Cumulative Effects
Reduced population fitness from impact on early life stages; Reduced prey from direct impacts on benthic invertebrates</td><td>Sediment deposition may extend beyond the footprint of the activities, but the spatial scale depends on the composition of the substrate and the hydrodynamic flow regime transporting the sediments in suspension. Sediments with a finer fractions may deposit further away from the source.</td><td>The temporal duration of impacts depends on resource susceptibility, with eggs likely being more sensitive to impacts.</td><td>Unknown at this time, but directly related to the frequency of dredging activity.</td><td>Minimum impacts on fishes and mobile benthic species that are able to move through the deposited layer, while greater impacts may occur on eggs.[1,2] Greater impacts may occur if dredging activities and rehandling occur in the proximity of hard-bottom habitats[3].</td></tr>
<tr><td>Availability of Empirical Information</td><td colspan="2">Mitigation Measures</td><td colspan="2">Mitigation Effectiveness</td></tr>
<tr><td>Little information is currently available on the tolerance of fishes (particularly eggs) and invertebrates to sediment overburdens, as well as on survival and hatching rates of buried eggs.</td><td colspan="2">None specific to fish; however, measures to minimize impacts to benthic habitats would also reduce potential impacts to fishes and large motile epifauna; restricted rehandling in areas where sensitive species are found would minimize the impacts of sediment deposition[4].</td><td colspan="2">Unknown at this time.</td></tr>
</table>

[1]ICES 1992; [2]Berry et al. 2011;[3] Lindeman and Snyder 1999; [4]Office of the Deputy Prime Minister 2005

The potential for impacts to fishes and large motile invertebrates from entrainment near the seafloor (Table 4.12) is of concern for slow moving organisms and benthic eggs and larvae. Most information on entrainment has been generated from nearshore and shallow water operations (Reine and Clarke 1998; La Salle et al. 1991; Nightingale and Simenstad 2001), and only a few studies have characterized potential impacts from offshore dredging (e.g., Lees et al. 1992; Van Dolah et al. 1992). Despite lack of general knowledge on the ecological consequences of this impacting mechanism on fishes and large motile invertebrates, most mobile organisms may be capable of escaping the suction field and, therefore, the anticipated impacts are likely to be low.

Although no mitigation measures were identified to limit or minimize the impact of entrainment on fishery resources, site-specific mitigation strategies could be considered if the dredge area was found to include an important spawning area for sensitive species and life stages (i.e., eggs, larvae) or a federally protected endangered or threatened species.

There is little empirical information on the impact of sound generated by OCS dredging activities on fishes and large motile invertebrates (Table 4.13). However, it is known that dredging operations generally produce low levels of sound energy that, although audible over considerable distances from the source, are of short duration. Consequently, the impacts of underwater sound on fish populations are expected to be temporary and localized. However, as highlighted in the review by Normandeau Associates, Inc. (2012), the hearing abilities of many fish species remain unknown, and further efforts should focus on developing audiograms under natural and relevant-anthropogenic noise levels for priority species including herring, mackerel, cartilaginous fishes (sharks, skates, and rays), and jawless fishes (hagfish and lampreys). This review also highlighted large data gaps on the hearing abilities of larval fishes and on the hearing changes related to ontogeny.

One of the most important sources of reduced water quality from sand dredging (Table 4.14) is the potential formation of pits and furrows that, in sufficiently deep pits (see for example Figure 2.12), have the potential to cause a drop in current strength and water flow resulting in lower oxygen exchange and localized depletion of oxygen. Small oil spills from dredges in open water are likely to have minimal impacts. Aside from the following oil spill prevention plans, the only other mitigation strategy to reduce impacts to water quality is to avoid formation of deep pits in areas important to demersal fish species.

Table 4.12

Impacting mechanism for OCS dredging on fishes and essential fish habitat: *Entrainment near the seafloor.*

<table>
<tr><th>Impact Pathway</th><th>Potential Effects on Fish</th><th>Spatial Extent of Impact</th><th>Duration of Impact</th><th>Frequency of Impact</th><th>Severity of Impact</th></tr>
<tr><td rowspan="3">Direct: ☒
Indirect: ☒
Cumulative: ☐</td><td>Direct Effects
Mortality/injury of entrained animals
Indirect Effects
Mortality/injury of prey sources
Cumulative Effects None likely</td><td>Impacts likely confined to the dredged area footprint.</td><td>The temporal duration of impacts depends on resource susceptibility. Large crustaceans, shelled invertebrates, benthic eggs and larvae are likely more sensitive to impacts[1,2].</td><td>Unknown at this time, but related to the frequency of dredging activity.</td><td>Minimum impacts on mobile species that are able to escape the suction field, while greater impacts are expected in large crustaceans, shelled invertebrates, benthic eggs and larvae[2].</td></tr>
<tr><td>Availability of Empirical Information</td><td colspan="2">Mitigation Measures</td><td colspan="2">Mitigation Effectiveness</td></tr>
<tr><td>There is very limited information on entrainment rates during offshore sand dredging operations.</td><td colspan="2">None specific to fishes and large motile epifauna; temporal/spatial environmental windows may be implemented to protect sensitive stages of species susceptible to entrainment[3].</td><td colspan="2">Unknown at this time.</td></tr>
</table>

[1]ICES 1992; [2]Lees et al. 1992; [3]Office of the Deputy Prime Minister 2005

Table 4.13

Impacting mechanism for OCS dredging on fishes and essential fish habitat: *Sound.*

<table>
<tr><th>Impact Pathway</th><th>Potential Effects on Fish</th><th>Spatial Extent of Impact</th><th>Duration of Impact</th><th>Frequency of Impact</th><th>Severity of Impact</th></tr>
<tr><td rowspan="3">Direct:
☒
Indirect:
☒
Cumulative:
☐</td><td>Direct Effects
Masking of sounds which can interfere with fish communication, predator/prey interactions; temporary loss of hearing
Indirect Effects
Behavior changes, displacement
Cumulative Effects
Not likely</td><td>Increased sound above background can be detected a few kilometers from the source, but the greatest sound levels are in close proximity to the source[1, 2].</td><td>Increase sound above background is confined to the extraction period.</td><td>Unknown at this time.</td><td>Behavioral changes and temporary loss of hearing may occur on some fish species[3], but these effects are most likely to occur over prolonged exposures at short distances.[5] Most fishes are likely to avoid noisy areas, and little risk of hearing damage exists for most species[4].</td></tr>
<tr><td>Availability of Empirical Information</td><td colspan="2">Mitigation Measures</td><td colspan="2">Mitigation Effectiveness</td></tr>
<tr><td>There is limited information on the hearing thresholds of most fish, and little is known regarding the response of fishes to sound under field conditions. Little information exists on the sound effects on large motile invertebrates (e.g., shrimp and crabs).</td><td colspan="2">None specific to fishes and large motile epifauna; dredge selection and reduced loading and on-site time on site may minimize impacts[6].</td><td colspan="2">Unknown at this time. However, passive listening may be implemented to detect vulnerable species[10].</td></tr>
</table>

[1]Greene 1987; [2]Clarke et al. 2002; [3]Nedwell et al. 2004; [4]Department for Environment Food and Rural Affairs (DEFRA) 2003; [5]Thomsen et al. 2009; [6]Office of the Deputy Prime Minister 2005; [10]Normandeau Associates Inc. 2012

Table 4.14
Impacting mechanism for OCS dredging on fishes and essential fish habitat: *Water quality.*

Impact Pathway	*Potential Effects on Fish*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☐	Direct Effects Mortality/injury from low dissolved oxygen (at the seafloor) and spilled oil (at the sea surface) Indirect Effects Behavior changes, displacement; Loss of prey in pits where anoxia and poor water quality persist Cumulative Effects Not likely	Low oxygen would be mostly confined to the footprint of any deep pits; Oil spills could occur at any vessel locations.	Deep pits can take years to completely fill. [1,2] The risk of spills would be during any vessel operations.	More likely to occur when deep anoxic pits are created and slowly fill. Unlikely at offshore sand ridge and swale complexes, with the exception of shoals affected by hypoxia in the Gulf of Mexico.	Impacts, if any, may be minimal, except where pits are formed.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Although environmental monitoring is required, there is little data availability to assess the effects of sand mining on water quality.	None specific to fishes and large motile epifauna; however, implementation of pollution control plan will reduce potential impacts on fishes and large motile epifauna. Avoiding the creation of deep anoxic pits in areas where natural sand transport rates are slow.[3,4,5] Implementation of pollution control plans.[6] Careful design of sand mining operations that minimize changes to hydrodynamics, grain size distribution, feature depth, sediment transport etc.		Unknown at this time.	

[1]ICES 1992; [2]Van Dolah et al. 1998; [3]USACE 2011b; [4]U.S. Army Corps of Engineers (USACE) Jacksonville District 2003a, Annex E; [5]NASA 2010c; [6]Gelpi 2012

The impact of turbidity to fishes and large motile invertebrates has been relatively well documented following dredging operations in estuarine and nearshore areas. A handful of studies with pelagic fishes have suggested the potential for short-time reduction in feeding from increased sediment suspended solid concentration (e.g., Matsumoto 1984; Jokiel 1989), while others have suggested that sediment plumes can be detrimental to pelagic and demersal eggs, and that spawning success can be adversely affected by causing temporary avoidance of spawning grounds (Westerberg et al. 1996). Consequently, and based on limited information, the potential for impacts to fishes and large motile invertebrates from increased turbidity in the water column (Table 4.15) is considered to be of short temporal duration, and likely of low impact to most of these resources. Mitigation strategies that could minimize the impact of turbidity on fishes and large motile invertebrates, aside from continued monitoring during operations at the borrow and placement areas and comparisons relative to state standards or exceedances of background (NTU) conditions, could include strategies that limit the spatial and temporal extent of turbidity plumes, particularly in areas near sensitive habitats/species. One approach may include the adoption of site-specific maximum plume dimension along the vertical and horizontal gradient from the source, above which dredging should be temporarily suspended. Another approach may include an assessment of the dredge cycle and speed/volume of the outflow such that turbidity levels fall within acceptable limits. These or similar strategies require site-specific information on sediment characteristics and local hydrodynamics.

Overall, limited quantitative information is currently available on the potential impacts of most impacting mechanisms to fishes and large motile invertebrates. Based on the available information, potential effects are likely minimal (except possibly for sediment/biota removal and increased sedimentation as shown in Tables 4.10 and 4.11), though precautionary measures could be taken to prevent any unforeseen impacts on these resources. To date, no quantitative measures of mitigation strategy effectiveness were identified with respect to fishes and large motile invertebrates.

One of the greatest challenges in assessing the impacts of sand dredging on fish populations and fishery resources is the limited information on the importance of sand ridge and swale complex habitats. Although the ecological value and function of shoals and shoal complexes has been previously discussed (e.g., Greene 2002; Diaz et al. 2004; Vasslides and Able 2008; Slacum et al. 2010, and numerous environmental documents submitted to BOEM), there are substantial knowledge and data gaps on the degree of association of fishes with these habitats, as well as their importance as spawning and nursery grounds for benthic and pelagic species. Limited scientific information has been generated through the various sand dredging monitoring projects in the U.S. on the cumulative and long-term impacts of offshore dredging and nearshore sand placement on demersal and pelagic fish, and large motile invertebrates (e.g., reduced prey availability, displacement, loss of habitat) (see Section 4.7). The understanding of the ecological value of these habitats is further complicated by the fact that their quality and uniqueness are not homogeneous in space. The Ship/Trinity/Tiger shoal complex in the Gulf of Mexico has been found to provide unique habitat for blue crabs (Gelpi et al. 2009; Condrey and Gelpi 2010). However, Dibajnia and Nairn (2011) identified more than 180 offshore sand ridge and swale complexes in the mid-Atlantic region. Without a better understanding of the relative importance of these shoals in the mid-Atlantic, it will be difficult to develop a long-term strategy that

Table 4.15

Impacting mechanism for OCS dredging on fishes and essential fish habitat: *Increased turbidity in the water column*

Impact Pathway	*Potential Effects on Fish*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☐	Direct Effects Clogging of gills and other reversible physiological changes Indirect Effects Behavior changes, displacement; reduced visibility to predatory fish; Cumulative Effects None likely	Impacts may extend beyond the footprint of the activities, but the overall spatial scale depends on the composition of the substrates, and the hydrodynamic flow regime. Plumes can extend to a few hundred meters from the source[1,2].	The duration of plumes associated with sand extraction is short (hours for cessation) and mostly confined to period of extraction activities[1,2,3].	Unknown at this time, but unlikely during dredging of offshore sand ridge and swale complexes; more likely at rehandling sites in shallow water with higher wave energy and turbulence.	Reduction in water clarity can have temporary impacts on the feeding efficiency of predatory fish species that rely on visual cues[4,5]. The impacts on fishery resources from sediment plumes are likely short-lived [6,7] and reversible.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Studies are lacking on the impacts to fish species in the open water, which are possibly less tolerant to increased turbidity than estuarine and nearshore species.	None specific to fish; however, measures to limit the spatial and temporal extent of turbidity plumes would reduce potential impacts to fishes and large motile epifauna; screening strategies can be implemented to reduce suspend sediment loading.[8] Monitoring the borrow and placement sites during operations, with cessation of dredging when water-quality criteria are exceeded[9,10].		Unknown at this time.	

[1]Hitchcock et al. 1999; [2]Newell et al. 2004a; [3]see Continental Shelf Associates Inc. (CSA) et al. 2010; [4]ICES 1992; [5]USDOI MMS 2009; [6]Saunders and Roberts 2010; [7]Tillin et al. 2011; [8]Office of the Deputy Prime Minister 2005; [9]U.S. Army Corps of Engineers (USACE) 2011a; [10]U.S. Army Corps of Engineers (USACE) 2011c

balances the needs for shoreline protection with the needs to minimize impacts to fish communities in this region.

Given the paucity of information regarding the effects of OCS sand mining on fish communities and indirect effects on predatory species, research and monitoring efforts at the borrow area could consider the following (this synthesis; La Salle et al. 1991; DEFRA 2001; Greene 2002; Rogers and Carlin 2002; Johnson et al. 2008; Austen et al. 2009):

- Long-term, before-after, control-impact studies (BACI) including pre-, during- and post-sand dredging may be useful to substantially contribute to an increased understanding of scale and nature of impacts, including the duration of impacts and recovery of both the benthic community and its associated fish community. These studies would also generate information on the response of fish populations to disturbance, and on their ability to locate comparable suitable habitat and alternate prey. In order to clearly distinguish impacts associated with sand dredging and the impacting mechanisms evaluated here, it is imperative to generate robust baseline information that captures natural seasonal and spatial variability of key indicator species or communities of concern.
- There are insufficient data on which to determine if there are spatial and temporal thresholds in local or cumulative habitat disturbances (including partial or complete removal of a sand shoal) within a large marine ecosystem that fish species populations and fishing levels are chronically impacted. BOEM has scheduled a "Workshop and Research Planning to Improve Understanding of the Habitat Value and Function of Shoal/Ridge/Trough Complexes to Fish and Fisheries on the Outer Continental Shelf" with the goal of bringing together a group of people with broad knowledge base to characterize the current scientific understanding of the function of shoal habitats. This workshop and the workshop final white paper will be essential to address critical gaps in understanding the habitat uniqueness, functions, and values of ridge/swale and shoals (individually and within a region) and to identify studies to fill those gaps. It is hoped that this current literature synthesis will provide a good basis for the workshop participants.
- Studies are needed to determine and document whether or not species displaced by sand mining activities are able to compensate elsewhere the lost energy and/or prey base, and if displacement increases crowding in nearby areas leading to density-dependent competition for prey resources. Similarly, these studies should evaluate if the displaced individuals represent a significant proportion of the population.
- Additional information is needed to identify the proximity of the borrow area to reefs and similar physical features that serve as important habitat for migratory and resident fishes, and to determine if site-specific buffers are needed to minimize impacts of siltation injury to hard-bottom habitats.
- Studies should focus on determining the ecological linkages between predator population metrics (growth and reproductive success) and food resource availability and quality at the borrow area, and collect information to help quantify potential changes in the food web via carbon budget analyses or food-web modeling.
- Other recommendations specific to certain impacting mechanisms are as follows:
  - Document the spatial and temporal scale of increased turbidity and suspended-solid levels, and characterize the potential effects to fishes and large motile

invertebrates based on comparison with existing empirical data, relative to background levels, or relative to clear-water species.

- Identify the fishes and mobile macrofauna species potentially impacted by dredge entrainment (cutterhead and hopper dredges) and those capable of escaping entrainment, along with information on entrainment rates and subsequent survival post outwash return.

- Assess the effect of sound from dredging operations on the feeding, reproduction, and migratory behavior of finfish, and exceedances of the proposed sound injury criteria (120 dB re 1 µPa for behavioral disturbance/harassment from a continuous sound source (e.g., dredging), Section 4.7; no criteria has been proposed for fish).

There is an increasing need to enable regulators and managers to better assess impacts to the seabed and their consequences to sensitive biological resources. However, there are no robust or reliable tools sensitive enough to quantify/predict these impacts and their short- and/or long-term consequences. Furthermore, because long-term and ecological interaction changes are generally not well documented, the lack of information makes identifying cumulative effects challenging.. Virtually all impacting mechanisms discussed in this section have significant data gaps on our understanding of their effects on fishes and large motile invertebrates. Efforts aimed at collecting such information, will improve the overall understanding of the impacts associated with offshore sand mining on fishery resources, as well as the potential impacts on the overall ecosystem health. Specific recommendations concerning major data gaps related to fish and large motile invertebrates are provided in Section 5.

4.4 FORAGING SEABIRDS

4.4.1 Key Species of Concern, Their Status, and Regulatory Protection Requirements

Seabirds are species that spend most of their time on open ocean waters and come to shore only to breed. Seabirds can also be categorized by the marine zones in which they tend to forage as: 1) pelagic seabirds that forage over open oceans during both the breeding and non-breeding seasons (e.g., shearwaters, petrels, fulmars, gannets); and 2) nearshore seabirds that forage in coastal waters and winter in coastal zones relatively close to shore (e.g., sea ducks, loons, grebes, terns, most gulls). Seabirds are protected under the Migratory Bird Treaty Act and many are considered Birds of Conservation Concern by the U.S. Fish and Wildlife Service (USFWS). Shorebirds are not included because few shorebirds have been documented in the Atlantic OCS (with the exception of phalaropes), though some species may pass over the OCS during migration flights (O' Connell et al. 2011b).

Pelagic seabirds feed mostly on fish, by surface feeding, plunge diving, pursuit diving, and stealing from other birds. Foraging locations are dictated by a combination of habitat features that affect prey availability, including attributes such as ocean and wind circulation patterns, the extent of upwelling and productivity, and water turbidity. Although there are limited data, the abundance of pelagic seabirds in areas of potential sand mining along the Gulf of Mexico and Atlantic coasts does not appear to be very high. However, species that are rare globally can be uncommon in these regions, such that few will be at risk but the risk to the population could be high. In the Central and North Atlantic region of the U.S., the highest densities of pelagic seabirds occur in the spring on the OCS near the shelf break (Kaplan 2011). In the South Atlantic, most pelagic seabirds occur far offshore; they use these areas both for feeding and migration (Jodice et al. 2012). The main exception among pelagic seabirds is the northern gannet, which is also often seen in small numbers from the shoreline. In the northern Gulf of Mexico, pelagic seabirds are found mostly far offshore and in relatively low numbers (Davis et al. 2000). Based on the available data and their feeding patterns, pelagic seabirds are less likely to be present in high concentrations in proximity to potential sand mining locations, they feed over large areas, and they are less likely to be at risk of impacts during any of the phases of OCS sand mining operations. The U.S. Geological Survey and BOEM have compiled all available datasets for seabirds on the Atlantic OCS (O' Connell et al. 2011a) and shorebirds (O' Connell et al. 2011b).

Along the U.S. Atlantic and Gulf of Mexico OCS, the species of foraging nearshore seabirds of greatest concern because of their high overwintering densities in offshore waters where sand dredging is likely to occur are discussed below. Much of this discussion is based on work done by the Sea Duck Joint Venture research program, consisting of aerial surveys consisting of 253 unique transects from the U.S.-Canadian border (44°46’ N) to Palm Beach, FL (26°56’ N) between January and March perpendicular to shore at intervals of 9.26 km and extending to the greater of 14.8 km or water depths of 16 m; the average transect length was 29.6 km (Silverman et al. 2011). Every other transect was replicated about one week later. Additional information was obtained from the two-year study (2007-2009) of avifauna off New Jersey as part of the Ocean/Wind Power Ecological Baseline Studies (Geo-Marine Inc. 2010).

Scoters (black, surf, and white-winged): These species overwinter in large number in U.S. offshore waters from Nantucket Shoals to Florida and along the Gulf of Mexico, with highest concentrations between Long Island Sound and Virginia including Delaware Bay and the mouth of Chesapeake Bay. They feed on bivalves, gastropods, crustaceans, polychaetes, and annelids, diving to depth up to 15-20 m. Scoters appear to exhibit strong site fidelity to overwintering and staging areas. According to studies by the Sea Duck Joint Venture research program (SDJV 2012) and the Atlantic Coast Wintering Sea Duck Survey (Silverman et al. 2011), particularly important offshore areas for black scoters during fall and spring migration along the U.S. Atlantic coast include the area around Cape Cod and Nantucket Shoals, though they can also occur in larger numbers off Delaware, Maryland, South Carolina, and Georgia. Surf scoters concentrate during winter offshore the mid-Atlantic coast, particularly along the Maryland, Delaware, and Virginia coasts, but also around Cape Cod and Nantucket Shoals. White-winged scoters winter along the Atlantic coast in smaller numbers and are most concentrated off the south coast of Long Island. All scoters had similar distributions in terms of the distance that flocks were observed offshore, with the mean distance ranging from 2.2-4.1 nm offshore and the maximum distance ranging from 9.3-12.8 nm offshore (Table 4.16). See Figures 4.7 and 4.8 for density distribution maps for scoters. Off New Jersey, the distribution of scoters peaked in water depths of 10 m and 6 km (3.8 mi) from shore, then increased again at 30 km (19 mi) offshore, and they were highly associated with shoals, most often occurring within 3-6 km (1.9-3.8 mi) from shoals, possibly indicating scoters feed at and between shoals (Geo-Marine Inc. 2010).

Long-tailed ducks: The Cape Cod and Nantucket Shoals areas are the most important offshore wintering areas for along the Atlantic Coast for long-tailed ducks, reaching densities of 38 and 69 birds per nm^2 respectively (Figure 4.8; Silverman et al. 2011). The distribution of long-tailed ducks in the Nantucket overwintering area, which may constitute a separate overwintering population, is driven by the high densities of amphipods (SDJV 2012). In New Jersey, long-tailed ducks were found to be concentrated around the edges of nearshore shoals (Geo-Marine Inc. 2010).

Bufflehead, Goldeneyes, and Mergansers: These sea ducks were grouped in the study by Silverman et al. (2011) because they are difficult to differentiate during aerial surveys, less common than scoters, and seen in similar habitats and locations. They were observed in low densities nearshore from Delaware Bay northward, and even lower and more variable off the southern coast of South Carolina and Georgia (Figure 4.9). They also tend to be more common closer to shore, including residence inside sounds (Table 4.16).

Figure 4.9 shows the distribution of all sea ducks along the Atlantic region. Table 4.16 shows the mean and standard deviation of the distance of sea duck flocks along the Atlantic coast to the nearest land in nautical miles, by species and year; the maximum distance is for the three years combined. These calculations were based on the east-west transects, excluding transects within the major bays and shoals, i.e., without including Pamlico Sound, Delaware Bay, Chesapeake Bay, Long Island Sound, Nantucket Shoals, and Cape Cod/Nantucket Bay (but not excluding the lines along the eastern edge of the Cape from 41°16'N to 42°06'N). Table 4.17 shows the estimated abundances by region for the period 2009-2011.

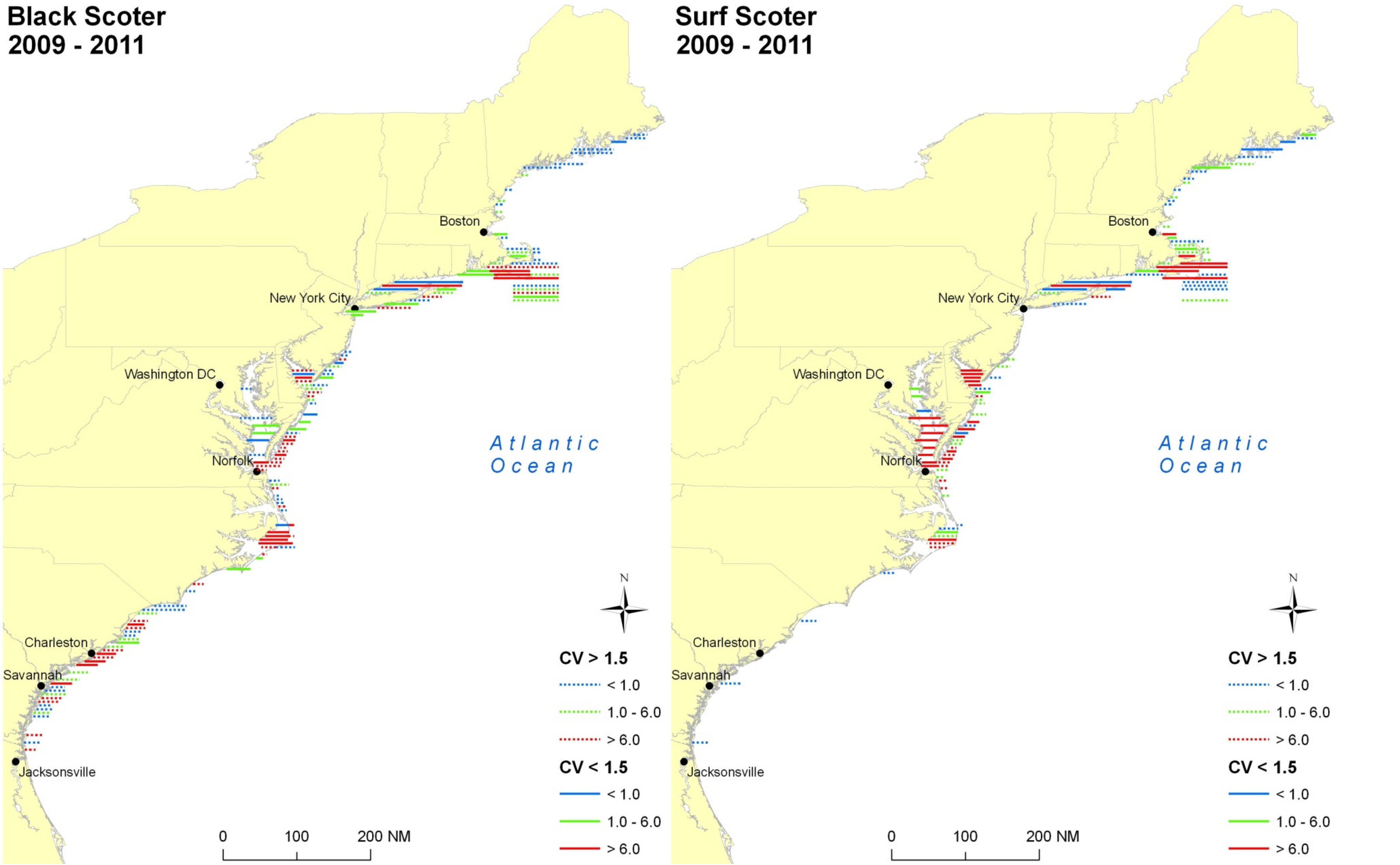
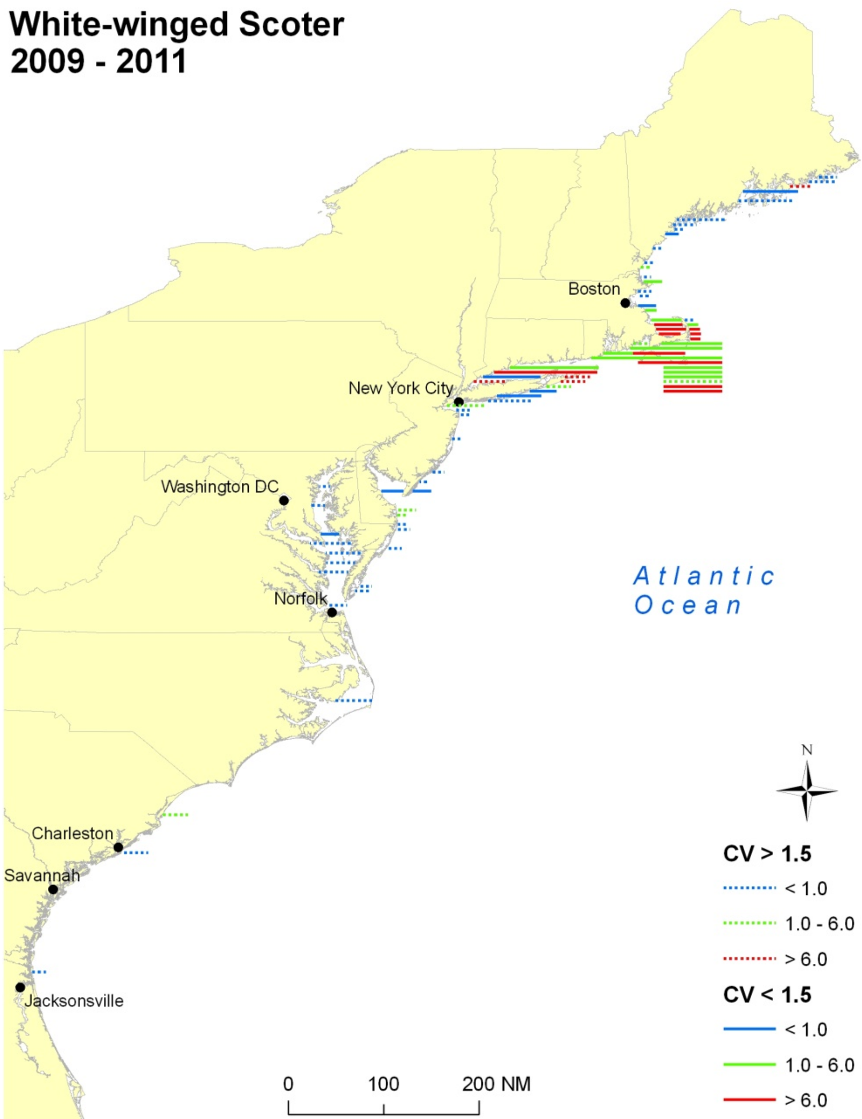


Figure 4.7 Average sea duck densities for black scoter and surf scoter for the period 2009-2011. Red > 6 birds/nm^2, Green = 1-6 birds/nm^2, Blue = < 1 birds/nm^2. CV = coefficient of variation, with dashes indicating higher variability in distributions. From Silverman et al. (2011).



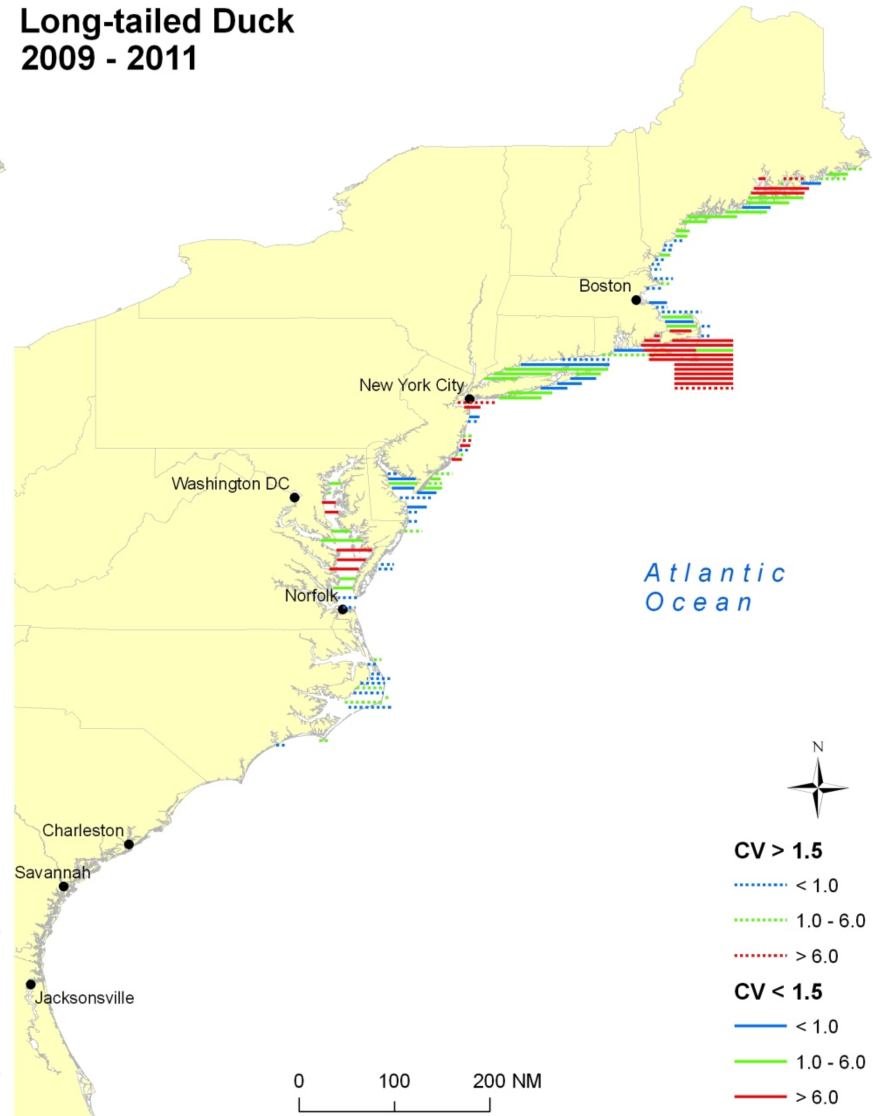


Figure 4.8 Average sea duck densities for white-winged scoter and long-tailed duck for the period 2009-2011. Red > 6 birds/nm^2, Green = 1-6 birds/nm^2, Blue = < 1 birds/nm^2. CV = coefficient of variation, with dashes indicating higher variability in distributions. From Silverman et al. (2011).

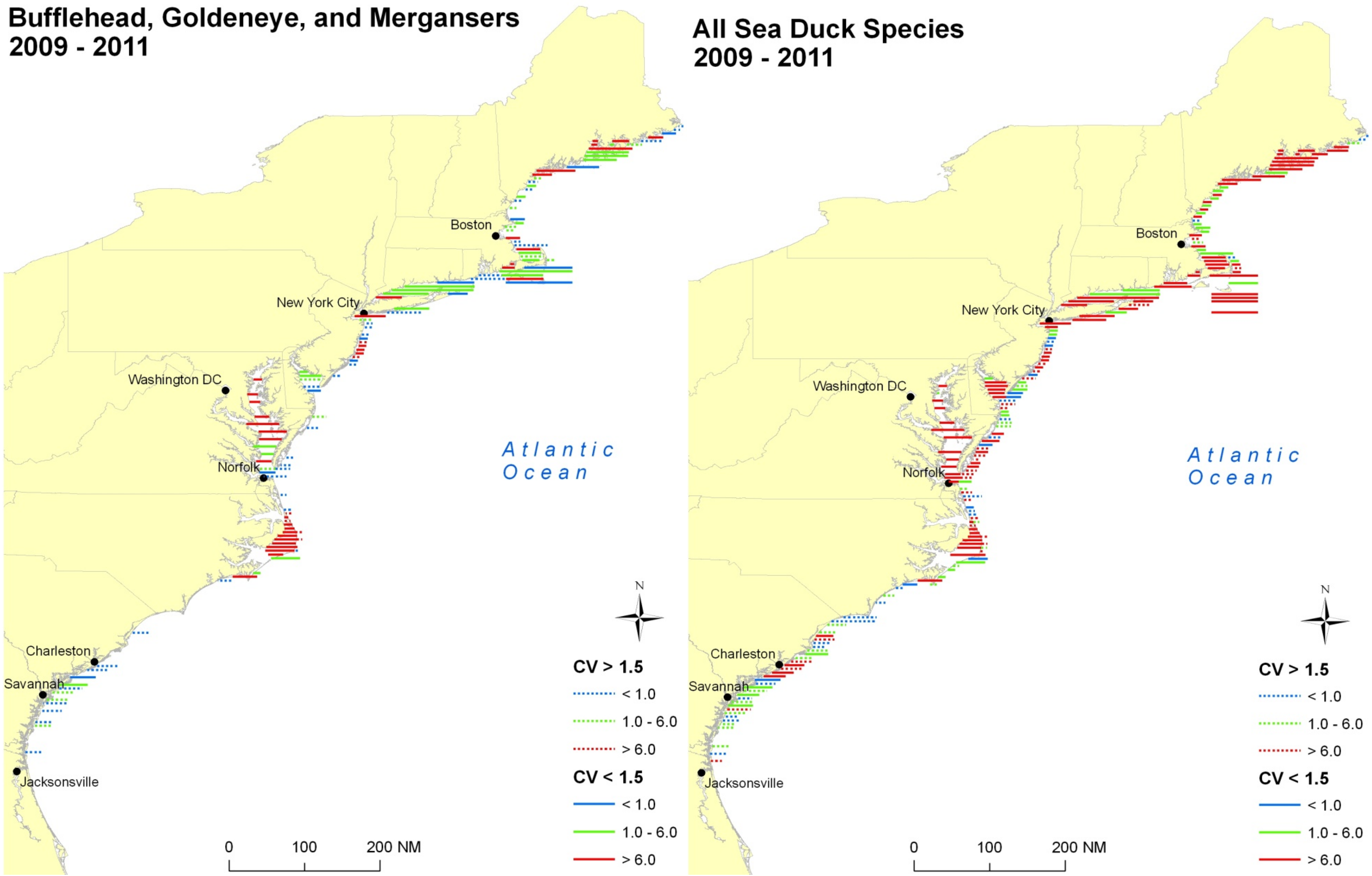


Figure 4.9 Average sea duck densities for bufflehead, goldeneye, and mergansers, and all sea duck species for the period 2009-2011. Red > 6 birds/nm^2, Green = 1-6 birds/nm^2, Blue = < 1 birds/nm^2. CV = coefficient of variation, with dashes indicating higher variability in distributions. From Silverman et al. (2011).

Table 4.16
Mean (Standard Deviation) distance of wintering flocks of sea ducks along the Atlantic coast to nearest land in nautical miles (nm), by species and year. From Silverman et al. (2011).

	Mean distance (nm)			Max distance (nm)
Species	2009	2010	2011	
Black scoter	4.1 (2.2) n = 383	3.2 (2.3) n = 152	3.4 (2.9) n = 120	12.8
Surf scoter	3.5 (2.1) n = 213	3.1 (1.9) n = 172	4.1 (2.2) n = 204	9.9
White-winged scoter	3.4 (2.3) n = 48	2.2 (1.8) n = 48	3.8 (2.1) n = 127	9.3
Long-tailed duck	3.4 (4.4) n = 423	2.8 (2.6) n = 373	3.2 (2.8) n = 592	24.9
Bufflehead, goldeneye, and merganser spp.	0.5 (1.1) n = 303	0.4 (1.0) n = 203	0.4 (0.8) n = 195	14.4

Table 4.17
Estimated three-year mean abundance (estimated SE) in thousands, by survey region and species for three yearly surveys conducted in 2009-2011. 0.00 values indicate estimates in the single digits. From Silverman et al. (2012).

		Common eider	Long-tailed duck	White-winged scoter	Surf scoter	Black scoter	Scoter spp.
	All Regions	251.4 (43.4)	236.6 (25.7)	58.6 (11.1)	149.2 (17.1)	211.3 (63.8)	429.4 (70.9)
1	Maine & New England	45.5 (6.3)	11.9 (2.9)	1.7 (1.0)	2.2 (0.5)	0.5 (0.2)	4.7 (1.3)
2	Cape Cod & Long Island Sound	205.8 (43.3)	194.6 (25.2)	55.2 (11.0)	66.0 (11.8)	32.6 (8.3)	155.4 (25.7)
3	New Jersey coast		2.7 (0.9)	0.08 (0.04)	0.1 (0.1)	0.3 (0.1)	1.8 (1.5)
4	Delaware Bay & DE/MD coast		0.8 (0.2)	0.7 (0.3)	47.7 (9.0)	34.0 (6.2)	84.6 (14.8)
5	Chesapeake Bay		25.4 (5.8)	0.8 (0.4)	30.3 (8.9)	4.0 (2.4)	37.9 (11.3)
6	Virginia coast & Pamlico Sound		1.2 (0.5)	0.01 (0.02)	2.8 (1.0)	19.5 (6.5)	23.6 (7.0)
7	Southern North Carolina coast				0.01 (0.01)	1.8 (0.7)	1.9 (0.8)
8	Central South Caroline coast			0.1 (0.2)	0.00 (0.01)	113.6 (62.5)*	112.2 (62.0)
9	S. SC coast & N. Georgia coast				0.00 (0.00)	3.2 (1.6)	3.3 (1.6)
10	southern Georgia coast			0.01 (0.01)	0.01 (0.01)	2.0 (2.4)	3.9 (4.2)

* Black scoter estimate excluding the large replicate value at 32°41′ in 2010 is 32.8 (12.5).

Silverman et al. (2012) noted that total scoter flock counts were consistent between years and more skewed than eider or long-tailed duck counts, indicating many transects with few flocks and a few transects with many flocks. Over 75% of sea ducks were observed in less than 20 m of water, within 7.4 km of the shoreline, and over seafloors with slope shallower than 1°.

Loring (2012) used satellite telemetry to study the overwintering distribution and behavior of black scoters in southern New England to estimate migratory timing and length of stay, quantify winter home range size and site fidelity between winters, characterize the habitat associated with core-use areas, and map relative probabilities of use in areas proposed for offshore wind energy development. Key results of these studies included:

- Black scoters spent about five months in wintering areas, arriving late October and departing early April
- About 50% returned to the area during the second winter
- They mostly occurred in shallow water close to shore (the mean and standard deviation for sample size of 140 was 4.0 ± 0.3 km from shore and water depth of 15.5 ± 0.7 m)
- Core-use areas (where birds spent >50% of their time) ranged from 3 to 2,500 km^2
- Half of the tracked birds occupied multiple core-use areas that were separated by an average distance of over 100 km
- Black scoters most often utilized benthic habitats composed of coarse-grained sand, where preferred prey items were both abundant and accessible, over medium-grained sand

Other seabirds that use offshore habitats for overwintering, migration, and feeding (mostly on fish) include northern gannet, red-throated loon, common loon, gulls, terns, pelicans, and cormorants. There are studies underway to better understand the distribution and use of offshore waters along the Atlantic coast, mostly triggered by the growing offshore energy developments in the region. BOEM has funded a study where surf scoter, northern gannet, and red-throated loon will be tracked using satellite transmitters for three years (2013-2015). This study will permit delineation of specific fall and spring migration corridors used by these species and will help to identify winter concentration areas for each species. These species are of particular concern because of their population trends and paucity of information on their use of offshore areas along the Atlantic coast from south of New England to the Outer Banks. The Biodiversity Research Center is also conducting a Department of Energy-sponsored baseline study along the Mid-Atlantic region that will focus on northern gannets, surf scoters, and red-throated loons as well as all seabirds, marine mammals, and sea turtles. Based on the available analyses, it is not possible to determine if sea ducks prefer to feed or occur in high concentrations over OCS sand shoals. It may be possible to analyze the existing transect data from 2009-2011 with the specific objective of testing hypotheses about sea duck distributions relative to sand shoals. Furthermore, in the coming years, there will be many advances in our knowledge about how foraging seabirds use offshore areas along the Atlantic coast.

4.4.2 Potential Environmental Effects and Mitigation Methods on Foraging Seabirds from OCS Sand Dredging by Impacting Mechanism

Existing information on each of the impacting mechanisms discussed in Section 3 and mitigation methods current in use for seabirds are summarized in the following sections. No direct studies of the potential environmental effects of dredging OCS borrow areas on seabirds were identified. Cook and Burton (2010), in their review of potential impacts to seabirds from aggregate dredging in the U.K., also did not find any such studies. Therefore, many of the potential effects discussed in this section are derived from studies of similar types of activities.

Cook and Burton (2010) classified potential effects of offshore dredging operations into two categories:

- Direct effects

  - Disturbance associated with dredging operations;
  - Increased turbidity associated with dredging operations; and
  - Shipping including disturbance, oil pollution, and collisions.

- Indirect effects through impacts on food supply
  - Impacts on benthic and fish communities;
  - Deposition of resuspended sediment, which may impact fish communities through alterations to habitat and the smothering of eggs and larvae; and
  - Potential release of naturally occurring toxins held within the sediment.

In their analysis of the vulnerability by species to these impacts, Cook and Burton (2010) concluded that divers (loons), grebes, and seaducks (eiders, scoters, and long-tailed ducks) were likely to be the most vulnerable, whereas storm petrels, gannets, and gulls were likely to be among the least vulnerable.

The types of impacting mechanisms described in Section 2 that could potentially affect foraging seabirds and are discussed below except for entrainment near the seafloor, and UXO, shipwrecks, other hard structures temporarily exposed during dredging, which are not likely to affect foraging seabirds.

4.4.2.1 Alteration of benthic habitat at the borrow area

Prey availability and quality for overwintering seabirds are important because of the high-energy demands of thermoregulation, migration, and breeding. Mass mortalities and decreased reproductive success the next breeding season for seabirds have been linked to winter food shortages (Camphuysen et al. 2002; Oosterhuis and van Dijk 2002). The preferred prey items and general habitats for wintering seabirds are generally known, but little is known about the relative importance of different types of habitats, and offshore ridge and swale complexes in particular, as foraging habitat for wintering seabirds.

At the 2006 Regional Workshop on Dredging, Beach Nourishment, and Birds on the South Atlantic Coast, Forsell and Watson (2006) reported on the results of USFWS offshore surveys in the region, noting that gulls, loons, northern gannets, and scoters were frequently observed over offshore shoals during winter. They suggested that dredging on shallow shoals and those closest to the shoreline should be avoided until more is known about the importance of these shoals as foraging habitat for migratory and overwintering seabirds. Overwintering birds concentrate in areas that provide suitable foraging habitat with preferred prey and water depths. Silverman et al. (2012) and Loring (2012) reported wintering seaducks were most common with 4-7 km from the shoreline and in shallow water. The avifauna studies off New Jersey (Geo-Marine Inc. 2010) found bird distributions highest closer to shore, particularly in winter. Loring (2012) also found that black scoters had large core-use areas and moved often among them. Long-term impacts to seabirds from repeated dredging, fragmentation, or complete removal of offshore shoals, with consequent potential reductions in prey availability, are unknown.

4.4.2.2 Increased turbidity in the water column

No literature was found that assessed the potential impacts of increased turbidity and suspended solids in the water column from OCS sand dredging and conveyance operations on foraging seabirds. No mention was made of this potential effect in any of the Biological Opinions, EISs, or EAs that were reviewed. There have been studies that show increases in turbidity can differentially affect foraging conditions for plunge-diving seabirds such as pelicans or pursuit divers such as loons (Haney 1986; Henkel 2006; Hao 2008). Henkel (2006), in a study in nearshore waters about 500 m off Monterey Bay, California, found that Forster's tern foraged more frequently over turbid waters, likely because small fishes were feeding in these areas of higher plankton abundance. Brandt's cormorants were more common in clear waters, likely because they rely on sight to catch fish near the bottom; turbid water may reduce their ability to see prey. Brown pelicans, western grebes, Clark's grebes, and marbled murrelets did not show any influence of water turbidity on their habitat usage.

Essink (1999) suggested that one cause of the long-term reduced breeding success of sandwich terns in the Wadden Sea, The Netherlands, was the increased turbidity in the coastal waters, forcing adults to forage out farther for prey to feed their young. Depending on the amount of fines in the dredged sand, there could be increased turbidity when the dredged materials are placed in nearshore rehandling areas. However, Bodge (2002) reported very little turbidity for homogenous coarse sand at the rehandling site for a project in Brevard County, Florida.

Cook and Burton (2010) conducted a desktop risk assessment to evaluate the potential impacts of marine aggregate extraction on seabirds in the U.K. Gravel is the targeted material and the overflow can contain very high levels of suspended sediments in a plume that can extend for up to 10 km. Thus, under these conditions, they expected that sediment plumes were likely to impact seabirds by reducing the ability of species to forage visually, However, the limited data from OCS borrow areas (e.g., Bodge 2002) indicated that turbidity during dredging and rehandling phases of the operations is low–only a few NTUs above background for homogeneous coarse sand. Thus, any effects from increased turbidity are likely to be very short-term, localized, and of minimal impact to foraging seabirds.

4.4.2.3 Increased sedimentation/deposition of fines on the seafloor

No literature was found that assessed the potential impacts of increased sedimentation or deposition of fines from OCS sand dredging and conveyance operations on foraging seabirds. No mention was made of this potential effect in any of the Biological Opinions or EISs/EAs that were reviewed. In their desk-top risk assessment, Cook and Burton (2010) expected that the elevated sediment plumes from extraction of aggregate would impact seabirds indirectly by smothering shellfish and the eggs and larvae of key forage species such as herring and sand eel. However, as discussed in Section 4.3.2.3, OCS borrow areas are usually composed of sand with few fines, and data on fish egg survival and hatching success relative to the sedimentation levels typically observed following sand mining are very limited.

Seaducks that feed on prey that live in the sediments have to dig several centimeters to access buried animals. Increased sedimentation could possibly reduce prey accessibility by the added sediment layer thickness. However, no measurements on increased sedimentation during dredging of offshore sand shoals where seabirds would be foraging were identified. Furthermore, such habitats are coarse-grained and exposed to both waves and currents; any increase in sedimentation on the seafloor in areas important to seabird foraging would likely be minimal and of short duration.

4.4.2.5 Sound

No literature was found that assessed the potential impacts of sound from OCS sand dredging and conveyance operations on foraging seabirds. No mention was made of this potential effect in any of the Biological Opinions or EISs/EAs that were reviewed. There are no measurements of underwater hearing of any diving bird. Consequently, Therrien et al. (2011) are training captive scaup to respond to sound underwater so behavioral audiograms can be compared with other measurements. Seabirds that dive deeply (e.g., cormorants, shearwaters, petrels, scoters) would have a higher risk of exposure to underwater sound compared to those species that feed or the surface or plunge-dive.

In the workshop summary for a panel convened by the U.S. Navy on underwater noise injury thresholds for marbled murrelets (from piling driving), SAIC (2011) reported that most sensitive frequency range of birds was 1,000-5,000 Hz. They also noted that "A literature review focusing on the effects of underwater sound on birds revealed very little information, most of which involved observations of behavioral responses." Because there were no relevant data on seabirds on which to determine injury thresholds, the panel decided to use data from fishes because of their similarity in body mass to the small marbled murrelets. The consensus from the workshop was that, for a 150 g marbled murrelet, the threshold for auditory injury (defined as hair cell damage due to implusive acoustic overexposure) was estimated to be a sound exposure level of 202 dB re 1μPa^2-s, and the non-auditory injury was estimated to be a sound exposure level of 208 dB re 1μPa^2-s. Larger seabirds (body mass of scoters is around 1,200 g; common eiders is about 1,500 g; Lovvorn and Jones 1991) would have higher injury thresholds. The panel noted that implusive sounds will have lower thresholds than continuous sounds.

Dooling and Popper (2007) conducted a literature synthesis of the effects of highway and construction sound on terrestrial birds. Based on data for 49 bird species (all terrestrial except for mallard duck), they generated a bird audiogram that showed birds hear (in air) best at frequencies from 1 to 5 kHz, the most sensitive frequencies were 2-3 kHz, and the low frequency cut-off of hearing was about 300 Hz. They concluded that birds are more resistant to both temporary and permanent hearing loss or to hearing damage from acoustic overexposure than are humans and other mammals that have been tested.

During sand and gravel extraction, source levels less than 180 dB re 1 μPa at 1 m can be anticipated, with the majority of the energy occurring continuously in the low frequency region (i.e., <1,000 Hz) (OSPAR 2009; Thomsen et al. 2009). Furthermore, as discussed in the next section, studies have shown that seabird flocks flush at distances of 500-1,000 m from an approaching vessel. Reine et al. (In prep) calculated SPLs of 120-132 dB re 1 μPa at distances of 500-1,000 m from dredges in transit. Based on the available data (which are very minimal), impacts to foraging seabirds from sounds during dredging operations (such as loss of foraging habitat and general stress) are not likely to be of concern because the sounds generated by dredges are generally lower in frequency than the frequencies birds hear (based on hearing of terrestrial birds in air).

4.4.2.6 Water quality

Oil spills are likely to be the only water quality concern for foraging seabirds. Even small amounts of spilled oil can affect seabirds under certain conditions. Offshore spills can become concentrated in convergence zones, where seabirds also tend to concentrate, to feed or when resting on the water surface. The two major pathways of oil exposure for birds are fouling of the feathers and ingestion (NRC 2003). Oiled feathers on birds lose their water-repellency, which leads to loss of buoyancy and insulating characteristics (Fry and Lowenstine 1985; Wiens 1995). When oiled, birds may lose their ability to dive and fly, have difficulty feeding, and increase their energy demands. The results include death by starvation, drowning, and hypothermia (Wiens 1995).

Birds can ingest oil during preening or ingestion of oil adhered to food items. Potential effects of ingestion include Heinz-body hemolytic anemia, immunosuppression, pneumonia; intestinal irritation, kidney damage, altered blood chemistry, impaired osmoregulation, decreased growth, decreased production and viability of eggs, and abnormal conditions in the lungs, adrenals, liver, nasal salt gland, and fat and muscle tissue (Fry and Addiego 1987; NRC 2003). Oil ingestion can result in three categories of population-level effects: 1) reduction in reproduction; 2) destruction of red blood cells leading to anemia; and 3) increased stress resulting in an increased susceptibility to disease, all of which reduce the health, survival, and abundance of oiled birds.

The effects of oil on birds vary by behavior, ecology, and life history. Tuck (1961) reported that only a small spot of oil on the belly was sufficient to kill murres. Fry and Lowenstine (1985) reported two of three Cassin's auklets died from application of 3-5 milliliters of oil to the breast feathers. However, Birkhead et al. (1973) reported observations of visibly oiled gulls successfully cleaning themselves after several weeks. Overwintering and migrating seabirds would likely to more susceptible to hypothermia because they would be present in offshore waters during cold periods when they have very high energy demands, and they only come ashore when very ill.

Though no oil spills have been reported during OCS dredging projects, oil spills from dredges have occurred, mostly close to shore and along inland waterways. Spills of lighter fuels will be less persistent than heavy fuels, particularly in open, offshore waters where natural dispersion rates are higher, reducing the potential for impacts to seabirds. Spills are low frequency but potentially high-consequence events. Dredging companies have oil spill contingency plans, but offshore spill response operations can be more difficult, with low recovery rates of spilled oil expected.

4.4.2.7 Vessel operations and interactions

There are some data on seabird collisions with other types of vessels, but not dredging operations. Black (2005) reported a very large event where, in one night, 899 seabirds collided with a fisheries research vessel transiting at speeds of 4-7 knots during fog and rain in the Southern Ocean, with 15 km of land. Of these, 215 died and 684 were collected, dried, and released alive. The use of ice-lights during night with poor visibility near seabird nesting sites was the likely cause of such high numbers of bird collisions with the vessel.

As part of their wind farm sensitivity index for birds in the North Sea, Garthe and Hüppop (Garthe and Hüppop 2004) classified common and velvet scoters as the most sensitive of 26 species of seaducks and seabirds to disturbance by ship and helicopter traffic (using their own personal experience during extensive at-sea surveys because of the paucity of data).

Kaiser (2002) and Kaiser et al. (2006) reported of a study in Liverpool Bay, England on seabird avoidance of high-use shipping routes and disturbance by vessel traffic. Their results showed that 82% of the common scoters were observed in areas that had no shipping activity for vessels >300 tonnes, and 12% of the birds occurred in areas that had light shipping activity (Figure 4.10). They indicated that these results suggested that common scoter avoid areas with activity associated with large vessels. They also reported that large flocks of common scoter were put to flight at a distance of 1-2 km from a 35-m long vessel, while smaller flocks were put to flight at a distance of <1 km, and suggested that common scoters are sensitive to disturbance by moving vessels. Kaiser (2002) inferred that vessels larger than that used in the study (U.S. dredging vessels can be longer than 100 m) would be expected to have a larger flushing distance.

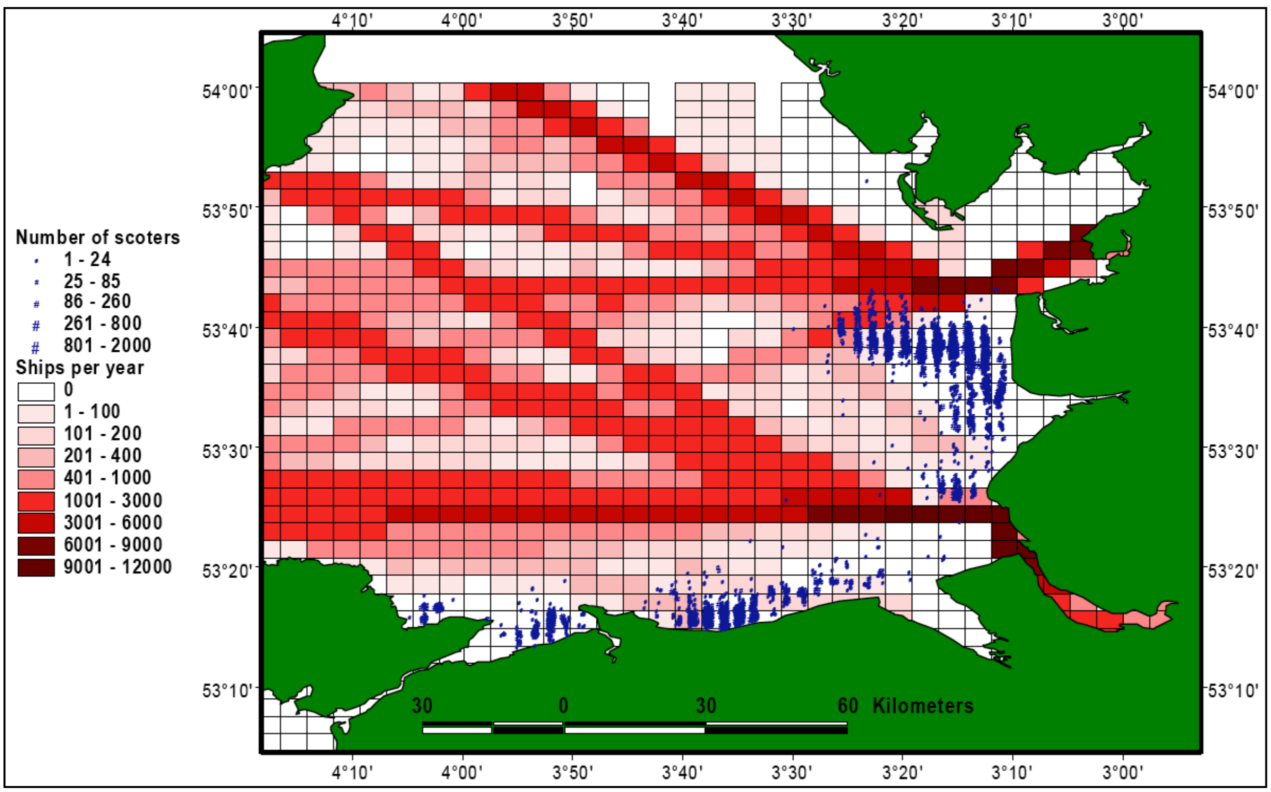


Figure 4.10 Map of the number of ships larger than 300 tonnes that passed through each 3.7 x 3.35 km cell for the period September 2003 to July 2004 and the number of common scoter sighted during eight overflights for the period 2002/2004, indicating a clear avoidance of the most intense shipping routes by scoters. From Kaiser (2002).

Schwemmer et al. (2011) studied avoidance of high-intensity shipping routes and disturbance by vessel traffic in the German parts of the North Sea and Baltic Sea. Loons clearly avoided the shipping routes. Flush distances differed significantly by species, with the longest distances reported for common scoters (median, 804 m), followed by white-winged scoters (404 m), long-tailed ducks (293 m), and common eiders (208 m). Flush distance was positively related to flock size for common scoter, long-tailed duck, and common eider, but not for white-winged scoter. Schwemmer et al. (2011) found indications of sea duck habituation within areas of channelized traffic; however, they questioned if habituation would occur in open waters outside of high-intensity vessel routes.

There have been several studies of the disturbance effects of high-speed ferries on overwintering seaducks. Larsen and Laubek (2005) conducted two types of surveys in the southern Kattegat Sea, Denmark: 1) four days of aerial surveys before, during, and after the passage of ferries in the ferry corridor through overwintering flocks of common eiders and common scoters; and 2) three days of ferry-based surveys of the proportion of flocks with escape response (either flight or diving) in 100-m intervals (Figure 4.11). Their results indicated major disturbances of birds within 500-1,000 m of the ferry route, and a possible increase of effects with flock size. Skov-og Naturstyrelsen (1997) reported similar distances for disturbance of overwintering common eiders by high-speed ferries in an earlier Danish study. They also reported that common eiders took about 10 minutes to return to feeding after being displaced.

In summary, it does appear seabirds avoid areas of high-intensity vessel traffic, and that flocks will be flushed by the passage of vessels traveling within 500-1,000 m of the flock. Larger flocks may be more likely to flush. Where and when seabirds concentrate around offshore sand shoals, they could be disturbed by dredging operations for the weeks and months of nearly continuous operations at the borrow area. Flocks would likely move to areas outside the dredging operations. Dredges and associated vessels traffic to/from the site could also temporarily disturb flocks along the transit routes, though studies have shown that they return quickly after vessel passage. Possible effects would be loss of foraging habitat and a negative effect of energetics.

4.4.3 Summary of Known Impacts on Foraging Seabirds due to OCS Dredging and Data Gaps

Because so little is known about the importance of offshore shoals as foraging habitat for migratory and overwintering seabirds, it is difficult to draw any conclusions on OCS dredging impacts from the available data. The greatest concerns expressed during biological assessments are the indirect, long-term impacts to foraging seabirds from repeated dredging, fragmentation, or removal of sand shoals. However, it is not even known if seabirds prefer to forage over OCS sand shoals, or if specific OCS shoals are more important than others. The available data indicates that distributions decrease with distance offshore. Assessing impacts to seabirds depends on understanding what they feed on and how these prey items may be affected by sand mining. There may be different responses among species by feeding guilds, that is, those species that feed on benthic invertebrates versus those that feed primarily on fish, with further differences based on demersal versus pelagic prey fishes. Hopefully, ongoing research on the use of OCS habitats, driven in part by offshore energy development along the Atlantic coast, will fill some of these key knowledge gaps.

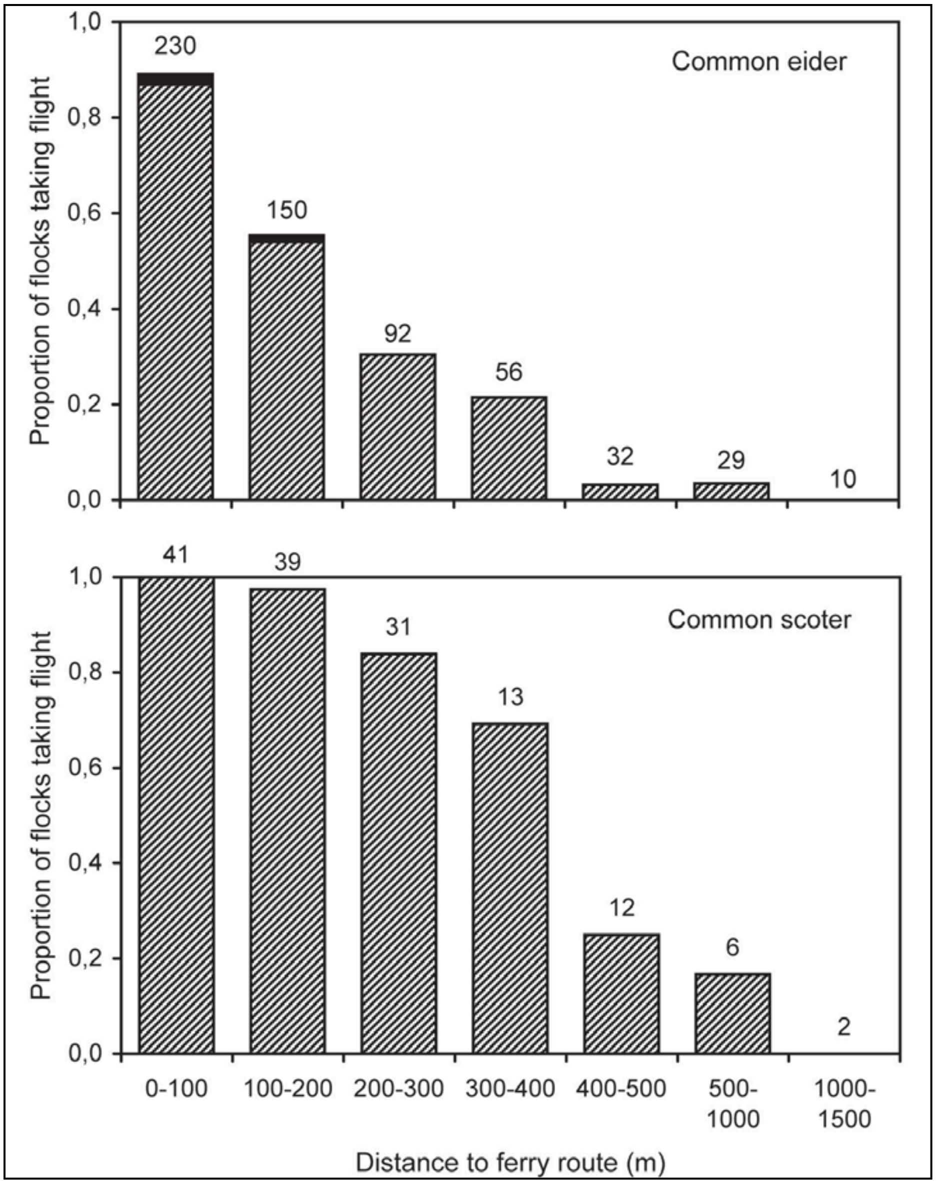


Figure 4.11 Proportion of common eider and common scoter flocks in the southern Kattegat Sea, Denmark, taking flight (hatched) or diving (solid) in response to an approaching ferry at varying distances from the ferry route. Sample size for each distance interval is given above columns. From Larsen and Laubek (2005).

Using mostly a basic understanding of seabird foraging requirements, Table 4.18 summarizes the likely impacts of sediment and biota removal during sand mining on foraging seabirds. Indirect and cumulative impacts could occur from an imposed alteration or loss of foraging habitats during the highly stressful wintering and migration periods. Offshore sand shoals are not only sites targeted for sand mining; they are also sites being considered for offshore wind energy development. In the long term, cumulative impacts could become important.

Cook and Burton (2010), in their review of potential impacts to seabirds from aggregate dredging in the U.K., came to the following conclusions:

- The relative importance of dredging zones as foraging locations for seabirds has not been directly assessed;
- There have been no direct studies of the use of dredging areas by birds before, during and after dredging activities; and
- There have been no direct studies of the interactions between seabirds and dredging vessels.

Forsell and Watson (2006) made the following recommendations to fill some of the key data gaps in our ability to assess the impacts of dredging offshore sand shoals:

- Identify bird use of shoals (three years minimum), including the seasonal and annual patterns, and the magnitude of bird use;
- Determine if and why birds are attracted to shoals;
- Determine the prey items selected by birds on the shoals, and the impact of shoal removal on these organisms;
- Develop models to predict the expected results of sand mining on shoals; and
- Test this model and develop dredging plans that have minimal impacts on bird populations.

Some of these data gaps may be addressed by ongoing studies that are looking at seabird use patterns in offshore areas. However, more-detailed studies will be needed to more quantitatively assess the potential impacts from repeated dredging that would result in partial or complete removal of individual shoals. These are very difficult questions to answer, involving multiple trophic levels and linkages, highly variable patterns in prey and bird distributions that are affected by short-term and long-term factors other than dredging (e.g., climatic and oceanographic conditions and trends), and inherent limitations in the available methods to conduct studies offshore. Marine bird researchers are working together at many different levels and coordinating with Federal agencies such as BOEM and the Department of Energy who are responsible for managing offshore development projects. Such coordination and cooperation among groups are essential, particularly now with shrinking research budgets.

Another potential impact on seabirds could be from disturbance and/or displacement during dredging operations at the borrow area and repeated vessel transits through areas with dense flocks (Table 4.19). Dredging operations can involve multiple vessels transiting several times per day (and night) between the borrow and placement areas for periods of months. Such ship traffic during the periods of high densities of overwintering seaducks could disturb them. However, as noted by Larsen and Laubek (2005), "The fact that disturbance may be observed cannot by itself be taken as evidence that the birds' ability to exploit the available food resources is negatively affected." They cite the study by Skov-og Naturstyrelsen (1997) that reported common eiders resumed feeding in about 10 minutes after being displaced by a passing ferry. Based on what is known from studies by Loring (2012) that showed core-use areas for wintering scoters in southern New England ranged from 590-759 km^2, it is not likely that temporary disturbances from dredge ship transits to/from offshore borrow areas would have impacts on the ability of seaducks to access preferred feeding habitats. There could be increased energy demands as a result of being flushed by a vessel, depending on the frequency of transits.

Table 4.18

Impacting mechanism for OCS dredging on foraging seabirds: *Alteration of benthic habitat at the borrow area.*

Impact Pathway	*Potential Effects on Foraging Seabirds*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☐ Indirect: ☒ Cumulative: ☒	Direct: None known Indirect: Altered forage base for species that feed on fishes and benthos in borrow areas Abandonment of foraging areas Cumulative: Reduced fitness due to reduced foraging or travel to alternative foraging areas, particularly if other offshore development activities reduce access to alternative areas (e.g., offshore wind energy development areas)	One-time removal of prey would be confined to the footprint of activities. Repeated sand removals could eventually affect the entire shoal, depending on allowed dredging patterns.	Sand-associated fauna are expected to recover within 1-3 years after a dredging event; repeated dredging could slow recovery, or cumulatively remove the sand habitat[1]. However, little is known about what are preferred prey species.	Unknown for specific shoals, but highly variable. There are plans for repeat dredging of shoals at five-year intervals for specific borrow areas.	Potential for long-term impacts if sand shoals are determined to be important foraging habitats for seabirds.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Good studies on recovery of benthos post-dredging[1]; but limited studies on longer-term impacts from repeated dredging or impacts to fishes[2]. No information on importance of borrow areas for prey species or seabird foraging.	None specific to seabirds; however, measures to speed recovery of fishes and benthic communities and habitats would also reduce potential impacts to seabirds. Could include time of year restrictions if it were a very important habitat for at-risk species and if the time of year of highest use was known and consistent.		Unknown.	

[1]See Table 4.2; [2]See Table 4.10

Table 4.19
Impacting mechanism for OCS dredging on foraging seabirds: *Vessel operations and interactions*.

Impact Pathway	*Potential Effects on Foraging Seabirds*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☐	Direct Effects Disturbance of flocks during dredging at the borrow area and during each vessel passage between the borrow area and placement site Indirect Effects Reduced fitness because of repeated disturbances or displacement from high-quality foraging habitats for the duration of vessel transits throughout the project Cumulative Effects Not likely because each dredging operation is expected to have a different transit pattern	Vessel operations at the borrow area; during transits from ports to the borrow area, and transits between the borrow area and placement site. Would include dredgers, support vessels, and survey vessels.	Vessel operations can be conducted 24 h per day, over periods of months.	Unknown. There are no reports of disturbances to flocks during dredging operations or transits.	Unknown. It has been shown that seabirds will avoid high-use shipping lanes[1,2,3,4]; birds will return to foraging shortly after a single displacement[5].
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Studies have documented flush distances of 500-1,000 m for similar species from passage of large and high-speed vessels [1,2,3,4,5]. Unknown if dredgers would have similar flushing distances when dredging at slow speeds, or during transits to/from the borrow area, particularly because the transits would likely be outside of normal shipping lanes	None currently proposed. Vessel transit routes could avoid known concentrations areas, particularly when there are dense wintering flocks.		Unknown.	

[1]Skov-og Naturstyrelsen 1997, [2]Kaiser 2002, [3] Larsen and Laubek 2005, [4]Kaiser et al. 2006, [5]Schwemmer et al. 2011

There are no data to assess the potential impacts of dredging vessel operations on seabirds or what might be appropriate mitigation measures if there were negative impacts. For dredging operations that take place during periods of high seabird densities, it may be possible to have the NMFS observers record information on seabird behavior during transits to/from the borrow area. USFWS seabird specialists could advise on the types of data to be collected, such as flock size within specific distances from the vessel, flush distance by species, percent of flock that flushes, and type of flush response (dive or taking flight).

Though there are very limited data, sounds generated during dredging operations are not likely to affect seabirds while diving underwater (Table 4.20). The frequencies generated during dredging operations are mostly lower than those heard by seabirds, and the sound levels are below thresholds thought to cause injury, though there are very limited data.

Oil spills from vessels are always a risk (Table 4.21). Dredgers can carry large volumes of fuel (up to 2.7 million liters). Spill response offshore can be difficult. Birds and oil slicks tend to concentrate offshore in convergence zones, and even small spills can affect large numbers of birds. The September 1999 spill of 7,950 liters of Intermediate Fuel Oil 180 from the dredge M/V *Stuyvesant* near the mouth of Humboldt Bay, near Eureka, California, killed an estimated 2,405 seabirds (California Department of Fish and Game (CDF&G) et al. 2007). The high costs of spill response and damage assessments are good incentives for dredging companies to make sure that they operate safely. BOEM requires preparation of a marine pollution control plan. Because fuel is stored in multiple compartments on dredges, the likelihood of a large release from a collision is low.

The potential for impacts to seabirds from increased turbidity in the water column (Table 4.22) and increased sedimentation on the seafloor (Table 4.23) is considered to be low. Increases in water turbidity are short term in duration and limited in areal extent, and no studies were identified that indicated that seabirds were particularly sensitive to such conditions. They would be affected by increased sedimentation only in the event that this led to reduce prey items, or perhaps their ability to access prey under a temporary layer of sediment. Again, any increases in sedimentation are likely to be short term and limited in areal extent considering the wave and current energies at most OCS borrow areas.

Based on the literature reviewed in this section, listed below are key data gaps in our understanding of the potential impacts to foraging seabirds from OCS dredging operations and recommendations for studies or syntheses to address these gaps.

- Determine how sea ducks and other foraging seabirds use different regions along the Atlantic inner shelf using current and on-going studies. Transect data along the mid-Atlantic region for 2009-2011 could be overlain on fine-resolution bathymetry and grain-size data to first determine if there are spatial relationships between sand shoals and flock numbers and flock sizes by species. The SDJV group flew a set of four additional replicates over the area during the 2010 surveys, so there may be sufficient data for this more detailed analysis.

Table 4.20
Impacting mechanism for OCS dredging on foraging seabirds: *Sound*.

Impact Pathway	*Potential Effects on Foraging Seabirds*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☐ Cumulative: ☐	Direct Effects None known Indirect Effects None known Cumulative Effects None known	Unknown. Seabirds flush at distances of 500-1,000 m from approaching vessels, removing them from the highest sound exposures.	Dredges generate broadband sounds at 160-180 dB re 1µPa at the source during dredging and transiting, and at 120-132 dB re 1µPa at 500-1,000 m from the dredge during transiting [3]. Dredging can be conducted 24 h per day, over periods of months. However, exposure would be short term, as the vessel transits past seabird flock.	Unknown.	Likely to be low, though there are very little data.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Very little data. Birds can hear frequencies of 1,000-5,000 Hz, with the most sensitive range being 2,000-3,000 Hz[1,2]. Auditory injury (defined as hair cell damage due to impulsive acoustic overexposure) for marbled murrelets was estimated to be a sound exposure level of 202 dB re $1\mu Pa^2$-sec, and the non-auditory injury was estimated to be a sound exposure level of 208 dB re $1\mu Pa^2$-sec[1].	None known at this time.		No mitigation measures proposed at this time.	

[1]SAIC 2011; [2]Dooling and Popper 2007; [3]Reine et al. In prep

Table 4.21
Impacting mechanism for OCS dredging on foraging seabirds: *Water quality*.

Impact Pathway	*Potential Effects on Foraging Seabirds*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☐ Cumulative: ☐	Direct Effects Direct mortality, reduced health and survival of oiled birds Indirect Effects None known Cumulative Effects Not likely	Would depend on the spill size, oil type, weather and oceanic conditions at the time of the release.	Would depend on the spill conditions.	Unknown; no spills have been reported during operations at the borrow area; one spill reported off CA.	Would depend on the number of birds oiled; the CA spill of 2,100 gallons resulted in 2,405 dead birds[1].
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
		A marine pollution control plan is required.		No spills from OCS dredging operations have been reported.	

[1]California Department of Fish &Game et al. 2007

Table 4.22
Impacting mechanism for OCS dredging on foraging seabirds: *Increased sedimentation/deposition of fines*.

Impact Pathway	*Potential Effects on Foraging Seabirds*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☐ Indirect: ☒ Cumulative: ☐	Direct Effects None known Indirect Effects Temporary reduction in access to benthic prey due to burial Cumulative Effects None known	Unknown, but likely very limited in scale.	Unknown, but likely very short duration because of sediment remobilization by currents and wave action in shallow areas used by seabirds for foraging.	Unknown, but likely uncommon occurrence.	Likely to be minor because of the limited spatial and temporal extent, and seabirds are known to regularly travel up to 100 km between foraging habitats.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	No data for offshore sand shoals where seabirds concentrate.	None specific to foraging seabirds.		None specific to foraging seabirds,	

Table 4.23

Impacting mechanism for OCS dredging on foraging seabirds: *Increased turbidity in the water column*.

Impact Pathway	*Potential Effects on Foraging Seabirds*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☐ Indirect: ☒ Cumulative: ☐	Direct: None known Indirect: Avoidance of foraging habitat in areas of high turbidity for species that rely on visual clues for prey location and capture Cumulative: None known.	At most OCS borrow areas, turbidity plumes are likely to be very limited; plumes could be higher at a rehandling area, but still are low[1].	The duration of plumes is short (hours for cessation) and mostly confined to period of extraction activities.[2,3,4].	Unknown.	Likely to be minor because of the limited spatial and temporal extent, and seabirds are known to regularly travel up to 100 km between foraging habitats.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Limited nearshore studies show variable responses by species and foraging methods[5].	None specific to seabirds; measures to limit the spatial and temporal extent of turbidity plumes would reduce potential impacts to fishes and large motile epifauna.		Unknown.	

[1]Bodge 2002; [2]Hitchcock et al. 1999; [3]Newell et al. 2004a; [4]CSA 2010; [5]Henkel 2006

- Determine the overwintering distribution and behavior of key species to estimate migratory timing and length of stay, quantify winter home range size and site fidelity between winters, characterize the habitat associated with core-use areas, and map relative probabilities of use in areas over OCS borrow areas, particularly shoals. Based on the available data, it appears that there is very little within-winter movement (small home ranges), but not much winter site fidelity. There is much uncertainty about fall/spring movements, distributions, what areas are important for spring staging, and why the birds select certain sites during the winter (E. Silverman, USFWS, pers. comm. 2012).
- Conduct finer-scale surveys aimed at more precise determination of flock locations and sampled more comprehensively over time, tides, weather conditions, different sea states, etc. to better understand the existing data on seabird distributions and short-term behavior.
- Better understand the prey consumed by key species and how dredging may affect these species.
- Better understand the underwater hearing capabilities of seabirds.

4.5 MARINE MAMMALS

4.5.1 Key Species of Concern, Their Status, and Regulatory Protection Requirements

At least 30 species of cetaceans (whales and dolphins), four pinnipeds, and one sirenian (Florida manatee, a sub-species of the West Indian manatee) are known or believed to occur within the U.S. Gulf of Mexico and Atlantic waters. All of these species are covered under the Marine Mammal Protection Act (MMPA), which prohibits the "taking" (harassing, hunting, capturing, or killing) of marine mammals. Six whales and the Florida manatee are listed under the ESA, all as endangered (Table 4.24). Short species summaries are provided for these endangered species.

Table 4.24

Cetacean species listed under the Endangered Species Act (ESA). All estimates of abundance for whales are taken from Waring et al. (2012). CV = coefficient of variance.

Scientific Name	Common Name	ESA Status	Abundance (CV)
Eubalaena glacialis	Right whale	Endangered	396 (North Atlantic)
Megaptera novaeangliae	Humpback whale	Endangered	847 (0.55)
Balaenoptera borealis	Sei whale	Endangered	386 (0.85)
Balaenoptera physalus	Fin whale	Endangered	3,985 (0.24)
Balaenoptera musculus	Blue whale	Endangered	unknown
Physeter macrocephalus	Sperm whale	Endangered	4,804 (0.38) (North Atlantic 1,665 (0.20) Gulf of Mexico
Trichechus Manatus latirostris	Florida manatee	Endangered	4,843[1]

[1]FWC (2012)

The North Atlantic right whale is one of the world's most endangered marine mammals (Clapham et al. 1999; IWC 2001). Total population size is unknown; however, 361 individually recognized whales were alive in 2009 (Waring et al. 2012). NMFS has designated critical habitats in Great South Channel, Cape Cod Bay, and Stellwagon Bank off Massachusetts, between 31°15N (approximately the mouth of the Altamaha River, Georgia) and 30°15N (approximately Jacksonville, Florida) from the coast out to 15 nm offshore, and within coastal waters out to 5 nm between 30°15N and 28°00N (approximately Sebastian Inlet, Florida). North Atlantic right whales are threatened by entanglement in fishing gear and collisions with large vessels. For the period 2005-2009, Waring et al. (2012) reported that the minimum rate of annual human-caused mortality and serious injury to right whales from ship strike records averaged 1.2 per year in U.S. waters.

Read (2012) provided this summary of the North Atlantic right whale:

> Coastal waters off the coasts of northeastern Florida and Georgia are the only known calving grounds for this critically endangered species (Kraus et al. 1986; Winn et al. 1986; Kraus et al. 1993; Garrison 2007a, Garrison 2007b; Good 2008). Approximately three-quarters of all known right whale births are believed to occur in this area (Kraus et al. 1993). Adult female whales take advantage of relatively calm nearshore waters to

give birth and nurse their young calves (Knowlton et al. 1994; Keller et al. 2006; Good 2008). A few juveniles and adult males also occur in this area each winter. This area was designated as critical habitat under the ESA in 1994 (NMFS 1994, 1995). A few right whale births have been observed outside this area, including one in 2010 that occurred further offshore, adjacent to the proposed Undersea Warfare Testing Range (USWTR) site off Jacksonville (Foley et al. 2011). NMFS is currently considering whether or not to expand this area of critical habitat.

Right whales arrive in the southeastern U.S. in November and typically remain through March (Kraus et al. 1993; Kraus et al. 1986; Keller et al. 2006). During this period, the whales fast, relying on stored energy deposited during the feeding season (Winn et al. 1986). In spring the whales return north to feeding grounds in the Gulf of Maine, the Bay of Fundy, and further north (Garrison 2007a). The migratory corridors linking the calving and feeding grounds have not been well documented (Firestone et al. 2008), but limited observations suggest that migrating whales stay relatively close to shore as they travel north in the spring (Schick et al. 2009). Today, right whales are vulnerable to ship strikes during this migratory period (Ward-Geiger et al. 2005; Knowlton and Brown 2007). As a result, NMFS has implemented a series of regulations, including restricted vessel speeds near ports, to reduce the risk of such collisions (NMFS 1994, 1995; see also Schick et al. 2009).

The few published records of right whales in Gulf of Mexico waters represent either distributional anomalies, normal wanderings of occasional animals, or a more extensive historic range beyond the sole known calving and wintering ground in the waters of the southeastern U.S. (Waring et al. 2012).

Humpback whales feed during spring, summer, and fall in the Gulf of Maine. Prey items are small schooling fish, crustaceans, and plankton. They use bubbles to trap or corral prey and, like other baleen whales, they filter large volumes of water. In winter, breeding whales migrate to calving grounds in the West Indies (Waring et al. 2012), though the migratory routes between tropical breeding and temperate feeding grounds are poorly understood. Since the 1990s, research has shown that the Mid-Atlantic region has becoming an increasingly important habitat for juvenile humpback whales. For the period 2005-2009, Waring et al. (2012) reported that the minimum rate of annual vessel collisions involving humpback whales averaged 1.4 per year in U.S. waters.

Fin whales are common from Cape Hatteras northward, and New England waters represent a major feeding ground (Waring et al. 2012). Like other baleen whales, they filter large volumes of water to feed on invertebrates and small schooling fishes. Calving is thought to take place October to January in the U.S. mid-Atlantic region. For the period 2005-2009, Waring et al. (2012) reported that the minimum rate of annual vessel collisions involving fin whales averaged 1.4 per year in U.S. waters.

Three of the ESA-listed species of whales in Table 4.24 are not likely to be at risk from OCS sand dredging activities: sei and sperm whales generally occur far offshore; and blue whales are only occasional visitors in U.S. waters (Waring et al. 2012).

Manatees occur throughout the southeastern U.S., regularly as far north as North Carolina in summer, but mostly in Florida and southeastern Georgia, and rarely west of the Suwannee River, Florida in the Gulf of Mexico (USFWS 2001). In 1976, critical habitat was designated for the Florida manatee including all of the known range at that time; all critical habitats are in Florida State waters. Manatees are herbivores that feed opportunistically on a wide variety of submerged, floating, and emergent vegetation. During winter, they concentrate at warm-water refuges, such as springs and power-plant cooling-water outfalls. Each winter, a population assessment is made using statewide aerial surveys and ground counts at these winter concentration areas; the latest results indicate a population of 4,843 animals, mostly in state waters (FWC 2012).

All marine mammals are protected under the MMPA, which prohibits harassment of marine mammals, which is defined as any act of pursuit, torment, or annoyance that: 1) has the potential to injure a marine mammal or marine mammal stock in the wild (Level A); and has the potential to disturb a marine mammal or marine mammal stock in the wild by causing disruption of behavioral patterns, including, but not limited to, migration, breathing, nursing, breeding, feeding, or sheltering (Level B).

4.5.2 Potential Environmental Effects and Mitigation Methods for Marine Mammals from OCS Sand Dredging by Impacting Mechanism

The types of impacting mechanisms described in Section 2 that could potentially affect marine mammals are discussed below except for UXO, shipwrecks, other hard structures temporarily exposed during dredging, which are not likely to affect marine mammals.

4.5.2.1 Alteration of benthic habitat at the borrow area

No literature was found that assessed the impacts to marine mammals from removal of biota from the borrow area. Most cetaceans in the Atlantic and Gulf of Mexico feed primarily on pelagic prey in the water column, and feeding areas are mostly in the north Atlantic or far offshore in the Gulf of Mexico. Juvenile humpback whales are thought to be increasing their use of the mid-Atlantic region for feeding during winter months, though there is no information on where they feed and if sand shoals are important feeding areas. Manatees feed in seagrass and algal habitats, which are not sand borrow areas.

4.5.2.2 Increased turbidity in the water column

No literature was found that assessed the impacts of increased turbidity in the water column on marine mammals. The short duration of suspended sediment plumes are not likely to affect highly mobile cetaceans. In the Biological Opinion for the Wallops Island project, NMFS 2008 stated: "… whales are not very easily affected by changes in water quality or increased suspended sediments, but if these alterations make habitat less suitable for prey species of turtles and/or whales, foraging capabilities may be hindered resulting in whales and/or sea turtles eventually leaving or avoiding these less desirable areas." Most concerns about the effects of turbidity on manatees were associated with impacts to seagrass habitats. Standard best management practices during dredging and rehandling operations would prevent impacts to seagrass through avoidance, buffers, and silt barriers, as appropriate.

4.5.2.3 Increased sedimentation/deposition of fines on the seafloor

No literature was found that assessed the direct impacts of increased sedimentation or deposition of fines on the seafloor on marine mammals. Any increases would be temporary and not likely in areas important for feeding for marine mammals in the region.

4.5.2.4 Sound

Anthropogenic sounds can affect marine mammals in a number of ways, including: a) disruption of behavior (e.g., feeding, breeding, resting, displacement from habitat, migration); b) masking of important sounds (which would reduce the ability to detect communications between animals due to elevated levels of background noise); c) temporary or permanent hearing loss; d) physiological stress or physical injury; and e) changes to the ecosystems that result in a reduction of prey availability (Richardson et al. 1995; Moore et al. 2012). Southall et al. (2007) grouped cetaceans into the following hearing groups based on estimated auditory bandwidths:

Low-frequency cetaceans	7-22,000 Hz (all are baleen whales)
Mid-frequency cetaceans	150-160,000 Hz
High-frequency cetaceans	200-180,000 Hz

Thus, cetaceans can hear the sounds generated during dredging operations, which are considered as continuous (versus pulsed with a rapid rise time, as for airguns and pile driving) and low in frequency (dredges generate sound in the range of 30-20,000 Hz, but generally below 1,000 Hz).

NOAA is in the process of developing a comprehensive acoustic policy that will provide guidance on assessing the impacts of anthropogenic sound on marine mammals. The acoustic guidelines will reflect qualitative considerations (e.g., masking, stress, cumulative impacts, and population consequences of sound exposure), as well as numerical criteria for temporary threshold shift (TTS: the hearing threshold rises and a sound must be stronger in order to be heard for a period usually minutes to hours, sometimes days) and permanent threshold shift (PTS: an irreversible elevation in the hearing threshold) onset (Southall et al. 2007). In the interim, NOAA’s current (2012) in-water thresholds for determining impacts to cetaceans are:

- <u>Level A</u>: 180 dB rms @ 1 µPa for potential PTS
- <u>Level B</u>: 160 dB rms @ 1 µPa for behavioral disturbance/harassment from an impulsive sound source (e.g., chirp seismic, side scan sonar, bathymetric surveys)
- <u>Level B</u>: 120 dB rms @ 1 µPa for behavioral disturbance/harassment from a continuous sound source (e.g., dredging)

To determine the distance from the dredging operations at which these thresholds would be exceeded requires estimation of the source level and calculation of the transmission loss. Furthermore, the actual responses to sound vary among species, individuals, the behavior at the time of exposure, past exposure to the sound that may have resulted in either increased or decreased sensitivity, and demographic factors (NRC 2005).

Richardson et al. (1985) studied the response of bowhead whales to dredging operations in the Beaufort Sea (suction dredging to create islands and undersea berms, and hopper barges dumping dredge materials). They also conducted experiments by playing back recorded dredging sounds (with a peak source level of 161 dB @ 1 μPa) for 40 minutes without the physical presence of dredges. They observed bowhead whales exhibiting disturbance behaviors (i.e., avoidance of the area, reduced number of blows/surfacing, and reduced duration of surfacing) at levels above 120 dB re 1 μPa in the 20 Hz to 1,000 Hz band, but not for all tests. Since then, as summarized in Richardson et al. (1995), there have been a few studies documenting how whales respond to dredging operations. Bryant et al. (1984) reported that gray whales temporarily abandoned a breeding lagoon in Baja, Mexico for several years due to intense shipping and dredging activity. They suggested that the constant dredging operation necessary to keep the channel open could have been the main source of disturbance, though it was not determined if the whales abandoned the area because of the shipping activity, increased sound from the dredging, or other reasons. According to OSPAR (2009), there are no recent studies (since the studies by Richardson et al. 1995) on the effects of dredging noise on marine mammals.

Southall et al. (2007) compiled information on whales and dolphin responses to vessels, from seven studies in which there were sufficient data on sound exposure received levels. They developed a severity scale for ranking observed behavioral responses, from 0 (no response) to 9 (outright panic, flight, attack of conspecifics, or stranding events; avoidance behavior related to predator detection). As shown in Table 4.25, for four of the studies, the exposure-received levels were 110-120 dB SPL, two studies were 110-130 dB SPL, and one study was 110-140 dB SPL. Severity ranks ranged from 0 to 6, with differences within a species and among species. These results suggest some avoidance at received levels of 110-120 dB @ 1 μPa, with increased negative behavioral responses at higher levels. It is interesting to compare these levels with the "background" sound levels of 116-118 dB @ 1 μPa measured by Reine et al. (In prep) off Wallops Island, Virginia.

For the Wallops Island dredging project, NASA 2010a) and NMFS (2007) both and separately calculated the distance over which sound would exceed the Level B continuous source threshold during dredging operations as:

- NASA (2010a): with a source level of 140 dB and transmission loss of 15 log R, sound would reach 120 dB within 862 m from the source
- NMFS (2007): with a source level of 164 dB rms @ 1 μPa and transmission loss of 15 log R, sound would reach 120 dB rms @ 1 μPa within 794 m from the source during loading, the noisiest part of the operations

Both organizations determined that underwater sound from the hopper dredge would not reach the Level A threshold; thus dredge-generated sounds would not result in any injury or mortality to marine mammals. In the Biological Opinion for the Wallops Island project, NMFS (2007) used these model results in their exposure analysis on potential behavioral responses to TSHD dredging operations over the 50-year lifetime of the project (dredging is proposed to occur every five years). They determined that whales would not be exposed to underwater noise levels greater than or equal to 120 dB based on the literature and the following factors: use of the area (for migration mostly); low likelihood that there would be high concentrations of animals present; required mitigation measures that include shut down of dredge pumps when a whale is observed within 1 km of the dredge; and the 500-yard restriction of vessel approach to right whales.

Table 4.25
Summary of behavioral responses of cetaceans exposed to nonpulses by type of sound source, available acoustic metrics, description of behavioral response (by individual and/or group), and a summary of corresponding severity score(s). Extracted from Southall et al. (2007). RL = response level.

Subject Species	Sound Source	Type of Acoustic Measurements	Type of Individual and/or Group Behavioral Responses	Summary of Severity Scale Analysis
Humpback whales	Vessel noise and presence	Individual RLs not reported but vessels identical to previous measurements	Vessel-based observations of individual movement and behavioral patterns around vessels	Exposure RLs 100-140 dB SPL; severity scores: 0 and 6
Humpback whales	Vessel noise and presence	RLs measured *in situ* near individuals observed	Visual observations of individual movement and behavioral patterns during vessel approaches	Exposure RLs 110-130 dB SPL; severity score: 6
Minke whales	Vessel noise and presence	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns during vessel approaches	Exposure RLs 110-120 dB SPL; severity score: 3
Sperm whales	Vessel noise and presence	Calibrated RL measurements made *in situ* near areas of exposure	Vessel-based observations and passive acoustic monitoring of individuals; movement patterns and behavioral responses	Exposure RLs 110-120 dB SPL; severity score: 3
White-sided and white-beaked dolphins	Vessel noise and presence	RL estimates based on source and environmental characteristics	Visual observations of individual and group movements and behavioral patterns during vessel approaches	Exposure RLs 110-120 dB SPL; severity score: 3
Bottlenose dolphins	Vessel noise and presence (approaches)	Calibrated RL measurements made *in situ* near areas of exposure	Passive acoustic monitoring of individual vocal output during vessel approaches	Exposure RLs 110-120 dB SPL; severity score: 2
Indo-Pacific dolphins	Vessel noise and presence	Calibrated RL measurements made *in situ* near areas of exposure	Passive acoustic monitoring of individual vocal output during vessel approaches	Exposure RLs 120-130 dB SPL; severity score: 5

With the actual field measurements of the sounds generated during various activities by three dredges used during the Wallops Island project (Table 3.1 in Section 3.3.5), the highest calculated source level was 178.7 dB re 1 μPa at 1 m and the lowest source level was 161.3 dB re 1 μPa at 1 m; at 1 km, the average sound was 125 dB re 1 μPa (Reine et al. In prep). Calculations can now be run with these new data for dredging operations and transits in the OCS and the nearshore pump-out operations, taking into consideration the level of propulsion.

Based on detailed study of two manatees, Gertein (2002) determined that manatees have a functional hearing range from 400 to 46,000 Hz, with peak sensitivities between 16,000 and 18,000 Hz. Thus manatees have difficulty distinguishing low-frequency sounds. In fact, research by Gerstein (2002) shows that manatees have difficulty hearing sound from vessels above the background noise, making them more susceptible to vessel strikes because they do not hear the vessels in time to move away. Research by Gerstein (2002) showed that manatees prefer seagrass habitats with less low-frequency sound; however, it could not be determined whether the manatees were avoiding higher sound levels from snapping shrimp or boat traffic. Miksis-Olds et al. (2007) showed that ambient sound levels do have a detectable effect on manatee communication; that is, under conditions of elevated sound levels, call rates decreased during feeding and social behaviors, and the duration of each call type was differently influenced by the presence of calves. These studies suggest that manatees have some ability to detect the sounds generated during dredging, but are not likely to be affected by them. Further, manatees are not likely to inhabit an OCS borrow area where the largest sounds are generated.

4.5.2.5 Vessel operations and interactions

Vessel strikes are a major source of marine mammal injuries and mortalities. Jensen and Silber (2003) analyzed the records for the 292 large whale ship strikes worldwide for the period 1975-2002 and found that the average ship speed was 18.1 knots. Vanderlaan and Taggart (2007) used these data in a regression model to show that the probability of lethal injury to large whales is 21% at 8.6 knots and increases to 79% at 15 knots. Manatees are also at a high risk of impact from vessel strikes; Lightsey et al. (2006) reported that, over the period 1993-2003, 24% of the 2,940 dead manatees that were examined in Florida were killed by watercraft-induced trauma. While dredging, the vessels travel at 2-3 knots; when in transit, they can travel at peak speeds of 17 knots.

In 2009, NMFS issued new regulations to reduce the threat of collisions between ships and North Atlantic right whales (50 CFR Part 224). They established three seasonal management areas (Northeast, Mid-Atlantic, and Southeast) and seasons (based on use of the area by right whales for feeding, migration, or calving and nursing) where vessels greater than 65 feet in length are under mandatory speed restrictions of 10 knots or less. These restrictions apply to most offshore dredges; however, they do not apply to vessels owned or under contract to the Federal Government, which actually excludes all USACE Civil Works projects from the requirements of this law.

Other mitigation measures often used to reduce the potential for vessel strikes or disturbance to whales during dredging projects include:

- Observers on board the dredge during specified periods (depending on when whales are likely to be present) to alert the captain when a listed whale is spotted within 1 km of the dredge.
- If a whale is observed within 1 km of the dredge, all pumps are to be turned off until the whale leaves the area (i.e., is farther than 1 km from the dredge).
- Dredge operators to monitor the right whale sighting reports (i.e., sighting advisory system, dynamic management areas, seasonal management areas) to remain informed on the whereabouts of right whales in the vicinity of the action area.
- Dredge operators to conform to the regulations prohibiting the approach of right whales closer than 500 yards (50 CFR 224.103 (c)). If a dredge vessel comes within the 500-yd buffer zone created by a surfacing whale, it would depart from the area immediately at a safe, slow speed.
- For dredging operations at night, the work area to be lit well enough to ensure that the observer/lookout can perform his/her work safely and effectively and that all mitigation measures can be performed to the extent practicable.

To protect manatees in Florida, the operator is usually required to comply with the Florida Fish and Wildlife Conservation Commission Standard Manatee Conditions for In-water Work:

a. All personnel associated with the project shall be instructed about the presence of manatees and manatee speed zones, and the need to avoid collisions with and injury to manatees. The permittee shall advise all construction personnel that there are civil and criminal penalties for harming, harassing, or killing manatees, which are protected under the Marine Mammal Protection Act, the Endangered Species Act, and the Florida Manatee Sanctuary Act.
b. All vessels associated with the construction project shall operate at "Idle Speed/No Wake" at all times while in the immediate area and while in water where the draft of the vessel provides less than a four-foot clearance from the bottom. All vessels will follow routes of deep water whenever possible.
c. Siltation or turbidity barriers shall be made of material in which manatees cannot become entangled, shall be properly secured, and shall be regularly monitored to avoid manatee entanglement or entrapment. Barriers must not impede manatee movement.
d. All on-site project personnel are responsible for observing water-related activities for the presence of manatee(s). All in-water operations, including vessels, must be shutdown if a manatee(s) comes within 50 feet of the operation. Activities will not resume until the manatee(s) has moved beyond the 50-foot radius of the project operation, or until 30 minutes elapses if the manatee(s) has not reappeared within 50 feet of the operation. Animals must not be herded away or harassed into leaving.
e. Any collision with or injury to a manatee shall be reported immediately to the FWC Hotline. Collision and/or injury should also be reported to the U.S. Fish and Wildlife Service in Jacksonville for north Florida or Vero Beach for south Florida, and to FWC at ImperiledSpecies@myFWC.com.
f. Temporary signs concerning manatees shall be posted prior to and during all in-water project activities. All signs are to be removed by the permittee upon completion of the project. Temporary signs that have already been approved for this use by the Florida Fish and Wildlife Conservation Commission must be used (see MyFWC.com/manatee). One sign which reads *Caution: Manatee Area* must be posted. A second sign measuring at least 81/2" by 11" explaining the requirements for “Idle Speed/No Wake” and the shut down of in-water operations must be posted in a location prominently visible to all personnel engaged in water-related activities.

No studies have been conducted to determine the effectiveness of these mitigation measures; however, no marine mammal strikes by dredging vessels have been reported to date. However, there is the potential for dolphins to be captured during sea turtle relocation trawling.

4.5.2.6 Water quality

The primary water quality concern to marine mammals during OCS dredging would be an accidental oil spill, though as noted in Section 3.3.7, no oil spills from offshore dredging operations have been reported. Most dredges burn diesel, which spreads quickly to a thin sheen and readily disperses in offshore areas. Based on studies with bottlenose dolphins, Geraci (1990) found that they could visually detect darker oil slicks, but would not be able to easily detect the thin sheens that cover large areas. In other studies, conducted in ocean pens using booms to contain a very thin, clear oil sheen in part of the pen, some dolphins showed avoidance of the oiled area after having surfaced in it, while others showed no avoidance. Geraci (1990) attributed this erratic behavior to discontinuities of thin sheens or limited ability to detect the oil on their skin. Smultea and Würsig (1995) tracked nine bottlenose dolphin groups for 5.6 hours during the 1990 T/V *Mega Borg* oil spill of 17.4 million liters of light grade, Angolan crude oil off Galveston, Texas. They reported that bottlenose dolphins could detect slick and mousse oils but did not react to lighter sheen oil.

Tests with gasoline-soaked sponges in contact with the skin of four species of whales showed no reaction (Geraci 1990). Based on these results, short-term exposures to thin sheens of diesel are not likely to affect the behavior or irritate the skin of whales and dolphins.

Geraci (1990) summarized what was known about potential impacts to cetaceans from historical spills up to the mid-1980s. He included one spill of light diesel fuel somewhere along the Alaskan shore where two killer whales (one sick and one dead) were reported, with no additional information provided.

Inhalation is likely the main pathway of exposure for marine mammals in open water. Whales and dolphins breathe air at the water surface. Inhalation of oil or oil vapors can result in mild irritation, respiratory stress, and death, depending on the degree and duration of exposure (Geraci 1990). Shortly after the *Exxon Valdez* oil spill in 1989 in Alaska, fourteen killer whales were missing from a very stable pod; though no link to oil exposure could be documented, inhalation was thought to be a likely pathway of exposure and death (Matkin et al. 2008).

Because fuel is stored in multiple compartments on dredges, the likelihood of a large release from a collision is low. Dredges are subject to U.S. Coast Guard rules and inspections, including having oil spill contingency plans in place. However, response options for offshore spills of light refined products such as diesel are few; most of the spilled oil will evaporate and disperse.

4.5.3 Summary of Known Impacts on Marine Mammals due to OCS Dredging and Data Gaps

Given the available information, some general conclusions can be drawn regarding the potential impacts of mining of OCS sand resources on marine mammals by impacting mechanism, discussed in order of greatest to least potential impact: vessel operations and interactions; sound; and water quality. The other potential impact mechanisms are considered to have minimal to inconsequential impacts or have no available data to present.

Vessel operations and interactions have the potential for impacting marine mammals during dredging operations (Table 4.26). Often, vessel strikes of large whales at speeds greater than 13 knots result in mortality (NMFS 2004), and it is the leading cause of death for manatees. Vessel strikes are one of the leading causes in the lack of recovery of the highly endangered North Atlantic right whale (Waring et al. 2012). However, there have been no reports of marine mammal strikes by dredges or support vessels during dredging operations, indicating the success of the mitigation measures taken to avoid interactions between dredges, associated vessels, and marine mammals.

Table 4.26
Impacting mechanism for OCS dredging on marine mammals: *Vessel operations and interactions*.

Impact Pathway	*Potential Effects on Marine Mammals*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☒	Direct Effects Increased risk of vessel strikes, thus injury or mortality Indirect Effects Avoidance, displacement from preferred feeding habitats Cumulative Effects Reduction in populations of listed species or distinct populations	Vessel operations extend from ports to the borrow area, and transits between the borrow area and placement site. Will include dredgers, support vessels, and survey vessels.	Vessel operations can be conducted 24 hours per day, over periods of months.	There are no reports of strikes of marine mammals by dredges or associated vessels[1,2]	During actual dredging operations, speeds are 2-3 knots, reducing the risk of injury from a strike. During transits, vessel speeds of up to 18 knots could result in mortality from a strike.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	NMFS maintains a large whale ship strike database, which includes information on vessel type when known[1]. The majority of vessel collisions with whales occurred at speeds between 13-15 knots[3]. Compliance with required notifications is high because of the presence of observers or lookouts on each dredge.	1. NMFS-approved observers to note presence of listed whales within 1 km 2. Shut down of pumps until the whale leaves the area 3. Monitor right whale sighting reports and comply with vessel speed restrictions 4. Move away from right whales that come within 500 yards 5. Follow the Standard Manatee Conditions for In-water Work		Likely to be highly effective, as there are no reports of strikes of marine mammals by dredges or associated vessels[1,2]	

[1] Jensen and Silber 2003; [2]USACE 2008a

Concerns about the potential impacts of anthropogenic sound in the ocean have grown in the last decade. Studies indicate that CSD operations are relatively quiet compared to other sound sources in aquatic environments, while hopper dredges produce sounds similar to those generated by vessels of comparable size (Clarke et al. 2002). The measurements of underwater sounds from three dredges working the Wallops Island dredging project (Reine et al. In prep) were used to calculate SPLs at a distance of 1 km to be 127.73 dB re 1 μPa during sediment removal, and 25 dB re 1 μPa during transit to the pump-out location (fully loaded). The SPLs for most dredging activities attenuated to below 120 dB re 1 μPa within 1-1.5 km. The data from multiple studies show that, while in transit, dredges generate sound at levels similar to other large vessels, which are dominated by cavitation noise from propellers and bow thrusters (de Jong et al. 2010). At these source levels, NOAA's in-water threshold for Level A (180 dB rms @ 1 μPa) would not be exceeded and no injury to marine mammals is likely. However, dredges will likely sounds above NOAA's in-water threshold for Level B (120 dB rms @ 1 μPa) at which there could be behavioral disturbance and/or harassment for animals within 1-2 km of the dredge during operations or transit, depending on the site-specific characteristics and dredge(s). With these new measurements, resource managers and project sponsors will be able to better estimate the potential impacts to marine mammals from sounds generated during dredging operations. Table 4.27 summarizes what is known about the potential impacts of dredge-generated sound on marine mammals.

Because migrating whales transit along corridors between feeding and nursery areas, they are likely to encounter dredges only once. Right whales in their calving and nursery areas off Georgia to northern Florida could encounter dredges operating in these areas. Also, juvenile humpback whales have increased their use of the Mid-Atlantic region as a winter-feeding range; there is the potential for multiple encounters in this region, though little is known of where they feed.

Oil spills from dredges are low-probability events but can have high consequences. The likelihood of a large release is low; there has been only one incident of an oil spill from a dredge not in inland waters (see Section 3.3.7). Most dredges use diesel fuel, which is expected to spread quickly into thin sheens. Cetaceans are not likely to avoid these kinds of sheens. Inhalation of oil or oil vapors within the first few hours to days after a large release is thought to be the exposure pathway of greatest impact. The overall risk from oil spilled in offshore areas is low and difficult to estimate the frequency or degree of impacts (Table 4.28).

Based on the literature reviewed in this section, listed below are key data gaps in our understanding of the potential impacts to marine mammals from OCS dredging operations and recommendations for studies or syntheses to address these gaps.

- The audibility and behavioral response of marine mammals is dependent on many factors, such as the physical environment (e.g., water depth, substrate type), existing ambient sound, hearing ability of the animal, behavioral context of the animal (e.g., feeding, migrating, resting), and acoustic characteristics of the sound. Therefore, this is one area in which additional in-water field measurements are needed for the source levels, frequency content, and radiated fields typical of OCS dredging operations.

Table 4.27

Impacting mechanism for OCS dredging on marine mammals: *Sound*.

Impact Pathway	*Potential Effects on Marine Mammals*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☒ Cumulative: ☒	Direct Effects Temporary or permanent hearing loss, physiological stress, physical injury Indirect Effects Behavior disruption, displacement from habitat and migration routes, masking of important sounds, changes to the ecosystems that result in a reduction of prey availability Cumulative Effects Reduction in populations of listed species or distinct populations	Increase sound above background can be detected a few kilometers from the source[1,2,3]. Modeling of dredge sound by NOAA and NASA for the Wallops Island project estimated exceedance of Level B: 120 dB rms @ 1 µPa at about 800 m from the dredge[4,5]. Actual measurements during this dredging project were used to calculate source levels and attenuation rates, and show decreases to 120 dB@ 1 µPa within 1.2 km and maintenance of that level for ~2.1 km from the source[6]	Dredges generate broadband sounds at 160-180 dB during dredging and transiting. Dredging can be conducted 24 h per day, over periods of months. However, impacts would be short term, as the vessel transits past individual animals.	Depends on species behavior. Migrating whales move quickly, so individual animals likely to experience one encounter. Right whales in their calving areas from Georgia to northern Florida could have multiple encounters. Juvenile humpbacks in the Mid-Atlantic winter feeding range could have multiple encounters with dredges. Manatees are not likely to be in Federal waters.	Dredges do not generate sound at levels expected to cause injury to marine mammals. Most likely impact is behavior disruption of individual animals in right whale calving areas or humpback whale feeding ranges if dredging occurs in concentration areas.

Availability of Empirical Information	*Mitigation Measures*	*Mitigation Effectiveness*
Only one field study where dredge sound levels were measured and related to bowhead whale behavior responses at 120 dB re 1 µPa in the 20 Hz to 1,000 Hz band[7]. There are several studies of vessel noise and presence with in-water measurements, showing minor behavioral response at 110-120 dB SPL and more intense behavioral responses at up to 140 dB SPL[8].	None specific to sound. However, mitigation measures for vessel operations (Table 4-46) likely effectively reduce the risks of exceeding sound thresholds.	No mitigation measures proposed at this time.

[1]Greene 1987; [2]Clarke et al. 2002, [3]USACE 2012; [4]NMFS 2007; [5]NASA 2010b; [6]Reine et al. In prep; [7]Richardson et al. 1985; [8]Southall et al. 2007

Table 4.28

Impacting mechanism for OCS dredging on marine mammals: *Water quality (mainly oil spills)*.

Impact Pathway	*Potential Effects on Marine Mammals*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☐ Cumulative: ☒	Direct Effects Mild irritation, respiratory stress, and death from inhalation Indirect Effects None known Cumulative Effects Reduction in populations of listed species or distinct populations if animals are killed	Spills of light refined products will spread quickly into thin sheens that can cover large areas, depending on the spill size.	Spills of light refined products will evaporate and naturally disperse within hours to days.	Very low probability of spills from dredges and associated vessels. There has been only one reported spill from a dredge in coastal waters, and no reported spill in the OCS.	Will depend on the spill volume and conditions. Large spills are unlikely; small spills quickly dissipate.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Little known about impacts from light refined product spills (only one poorly documented report[1]). Both field studies and actual spills shown that bottlenose dolphins likely do not detect or avoid thin sheens[1,2].	None specific to dredging operations, however, all vessels have to be inspected by and follow the rules of the U.S. Coast Guard and have oil spill contingency plans.		There have not been any reported spills in the OCS.	

[1]Geraci 1990; [2]Smulte and Wursing 1995

- More data on the level of background, or ambient, sounds in borrow areas, along transit routes, and at pump-out locations would allow a better understanding of the potential increased risks to marine mammals from sounds generated during dredging operations. Reine et al. (In prep) measured background levels at the Wallops Island, Virginia project areas that averaged 117 dB or just 3 dB below the current NOAA threshold for Level B behavioral disturbance from a continuous sound source.
- NOAA is expected to release their new acoustic guidelines for assessing the effects of anthropogenic sound on marine mammals for public comment soon. Once the new numerical criteria are finalized, the U.S. Navy Acoustic Effects Model may be a useful screening tool for BOEM to evaluate potential acoustic impacts to marine mammals during dredging operations.

4.6 SEA TURTLES

4.6.1 Key Species of Concern, Their Status, and Regulatory Protection Requirements

Five species of sea turtles occur in U.S. waters along the Atlantic Ocean and Gulf of Mexico, and all are listed as threatened (T) or endangered (E) under the ESA: green (T, except the Florida breeding population, which is listed as E), hawksbill (E), Kemp's ridley (E), leatherback (E), and loggerhead (T). All sea turtles lay eggs on tropical or semi-tropical sand beaches. Juveniles are pelagic for years to decades and can migrate over very large areas, foraging in convergence zones and *Sargassum* mats. As they approach adulthood, they head to coastal waters and feed on vegetation (green), sponges (hawksbill), jellyfish and other soft-bodied invertebrates (leatherback), or mollusks and crustaceans (loggerhead and Kemp's ridley). All sea turtles are listed under the ESA and thus require consultation for dredging operations with NMFS (for at-sea life histories) and USFWS (for nesting on beaches) under Section 7. The following summaries of the life history of sea turtles are only for the Atlantic and Gulf of Mexico OCS sand mining areas, focusing on nesting, nearshore, and pelagic habitats in U.S. waters.

Green sea turtles (*Chelonia mydas*) nest along the east coast of Florida, particularly in Brevard, Indian River, St. Lucie, Martin, Palm Beach and Broward Counties. Hatchlings migrate to the oceanic zone and concentrate in convergence zones. When juveniles reach a certain size, they move to the nearshore zone to forage, occurring as far north as Long Island Sound. The bays, sounds, and estuaries of North Carolina are particularly important neritic habitat for large juvenile green turtle. Green sea turtles mostly forage in seagrasses and macroalgae habitats, but small green turtles can also be found over coral reefs, worm reefs, and rocky bottoms. Adult green turtles are only occasionally found north of Florida (summary based on Smultea and Würsig 1995).

Hawksbill sea turtles (*Eretmochelys imbricata*) occur throughout the Gulf of Mexico (especially Texas) and offshore southern Florida (most often near the Florida Keys and off Palm Beach County). Nesting is restricted to the southeastern coast of Florida (Volusia through Dade counties) and the Florida Keys. Hawksbill sea turtles are rare north of Florida, but strandings and sightings have been recorded as far north as Massachusetts. Sponges are the principal diet, thus they forage on coral reefs, hard bottoms, and mangroves along Florida, and stone jetties in Texas (summary based on NMFS and USFWS 1993).

Most Kemp's ridley sea turtles (*Lepidochelys kempii*) nest at a single aggregation area, Rancho Nuevo on the eastern coast of Mexico, though some nesting also occurs in Texas, Alabama, Florida, Georgia, South Carolina, and North Carolina. Hatchlings occur offshore of the shelf break in floating mats of *Sargassum*; most Kemp's ridley post-hatchlings remain in the Gulf of Mexico, though some are transported out of the Atlantic Ocean. Juveniles occur in shallow coastal water, feeding mostly on crabs and shrimp in a range of benthic habitats. They are most frequently found in bays from south Texas to southwestern Florida in the Gulf of Mexico, although they also are found in large estuarine systems from Florida to New England along the Atlantic coast. Known foraging areas along the Atlantic coast during the warmer months include Pamlico Sound, North Carolina, Chesapeake Bay, Virginia, and Long Island Sound, New York. Other foraging areas likely include Charleston Harbor, South Carolina, and

Delaware Bay, New Jersey. Kemp's ridleys migrate south in winter, and large clusters of Kemp's ridleys have been reported off North Carolina, representing a confluence of turtles migrating south from various summer grounds. The offshore hard bottom and live bottom habitats south of Cape Canaveral have been identified as an important overwintering area for seasonal migrants along the U.S. Atlantic coast. In spring, there is northward migration back to coastal foraging grounds (summary based on NMFS and USFWS 1993).

The leatherback sea turtle (*Dermochelys coreacea*) nests in Florida, with records of 540-1,747 nests per year between 2006 and 2010; most nests are along the southeast Atlantic coast in Brevard through Broward Counties. Little is known about the habitats utilized by young juvenile leatherbacks because these age classes are entirely oceanic. However, it is known that they do not associate with *Sargassum* or other flotsam, as is the case for juveniles of the other sea turtle species. In the North Atlantic, leatherbacks are found from the Caribbean to Newfoundland with extensive seasonal migrations between temperate and tropical waters. They migrate farther and into colder water than any other species, thus they are the most pelagic sea turtle. They feed mainly on jellyfish, and their distribution is related to jellyfish aggradations, such as off the mouth of the Chesapeake Bay (summary based on NMFS and USFWS 1992)

The loggerhead sea turtle (*Caretta caretta*) is the most abundant sea turtle occurring in U.S. waters. NMFS and USFWS have identified five nesting subpopulations for loggerheads in the northwestern Atlantic: the Northern U.S. (Florida/Georgia border to southern Virginia); Peninsular Florida; Dry Tortugas; Northern Gulf of Mexico; and Greater Caribbean. However, loggerhead sea turtles nesting is most concentrated on beaches from North Carolina through Florida. Hatchlings migrate to the oceanic zone and become associated with *Sargassum*, drift lines, and other convergences. Juveniles inhabit continental shelf waters from Cape Cod Bay, Massachusetts, south through Florida and the Gulf of Mexico. Juveniles prefer relatively enclosed, shallow water estuarine areas (such as Pamlico Sound and Indian River Lagoon), whereas non-nesting adults prefer more open ocean access (such as Chesapeake Bay and Florida Bay). Essentially all shelf waters along the Atlantic and Gulf of Mexico shorelines are inhabited by juvenile and adult loggerheads (summary based on NMFS and USFWS 2007b).

4.6.2 Potential Environmental Effects and Mitigation Methods for Sea Turtles from OCS Sand Dredging by Impacting Mechanism

The types of impacting mechanisms described in Section 2 that could potentially affect sea turtles and are discussed below include: sediment and biota removal within the borrow area, increased turbidity in the water column, increased sedimentation/deposition of fines on the seafloor, sound, vessel operations and interactions, and water quality. Impacting mechanisms that are not likely to affect foraging sea turtles and are not discussed further are UXO, shipwrecks, and other hard structures temporarily exposed during dredging.

4.6.2.1 Alteration of the benthic habitat within the borrow area

Loggerhead and Kemp's ridley sea turtles are the most likely to be directly affected by disturbance or loss of foraging habitat from removal of sand and biota from sandy substrates because they feed on invertebrates that live in these habitats. However, little is known about how important offshore sand deposits are for foraging. Large juvenile loggerhead sea turtles prefer to forage in semi-enclosed, shallow water estuarine habitats, such as Pamlico Sound and the Indian

River Lagoon; Adults tend to utilize estuarine areas with more open ocean access, such as Chesapeake Bay (Dodd 1988). However, juvenile loggerheads readily move between coastal and pelagic habitats (McClellan and Read 2007), and females are known to aggregate offshore nesting beaches before coming ashore to nest and during inter-nesting periods. Thus, loggerheads may be present over and forage in sand borrow areas.

Kemp's ridleys feed predominantly on portunid crabs (NMFS 2011), thus sand borrow areas, particularly those closer to shore and in shallow water (<35 m) in the Gulf of Mexico, could be important foraging habitats. In a Biological Opinion for the USCAE New York District, NMFS (1995) stated that the preferred prey for Kemp's ridleys in that region was the spider crab, which would not be found in high abundances on large, sandy bottom habitats. In the biological opinion for the Wallops Island, Virginia project, NMFS (2007) stated that: "…there is no information to indicate that the borrow areas proposed for dredging have more abundant turtle prey or better foraging habitat than other surrounding areas. The assumption can be made that sea turtles are not likely to be more attracted to the borrow areas than to other foraging areas and should be able to find sufficient prey in alternate areas." In the Gulf of Mexico, Condrey and Gelpi (2010) found that Ship and Tiger Shoals off Louisiana (which are sources of OCS sand for coastal restoration projects in Louisiana) were important offshore spawning/hatching/foraging grounds for a large segment of the Gulf of Mexico blue crab fishery from at least April-October. However, no information is available on the importance of these shoals as foraging habitat for Kemp's ridley sea turtles.

Because benthic resources are expected to recover in a relatively short period (see Section 4.6.2) and dredging affects mostly a small area of the foraging habitat, most Biological Opinions indicate that dredging of OCS sand borrow areas will temporarily disrupt normal feeding behaviors for sea turtles, but will not likely affect the forage base. In the long term, it is not known if complete removal of offshore sand within specific coastal regions would potentially impact distinct populations that would have to spend additional energy resources to travel to other foraging areas.

4.6.2.2 Increased turbidity in the water column

No literature was found that assessed the direct impacts of increased turbidity and suspended solids in the water column from OCS sand dredging and conveyance operations on sea turtles. The Dam Neck Biological Opinion (NMFS 2012) stated: "Excessive turbidity due to coastal development and/or construction sites could influence Atlantic sturgeon, sea turtle, and whale foraging ability; however, based on the best available information, whales, Atlantic sturgeon, and turtles are not very easily affected by changes in water quality or increased suspended sediments unless these alterations make habitat less suitable for listed species and hinder their capability to forage and/or for their foraging items to exist. If the latter occurs, eventually these species will tend to leave or avoid these less desirable areas (Ruben and Morreale 1999)." However, the reference cited was a draft, unpublished Biological Assessment that could not be located. Studies by Ruben and Morreale (1999) in New York waters found that foraging turtles mainly occurred in areas where the water depth was between approximately 16 and 49 feet, interpreted to be a natural limiting depth where light and food are most suitable for foraging turtles. Sea turtles normally forage in areas with a wide range of turbidity, and no studies were identified indicating that temporary increases in turbidity interfered with sea turtle ability to detect prey items.

Potential indirect impacts could be related to turbidity impacts to prey items or habitats. However, the mobility of sea turtles would likely minimize such impacts (see discussion in Section 4.6.2.4 on increased sedimentation/deposition of fines on the seafloor, which could affect benthic prey items).

4.6.2.3 Increased sedimentation/deposition of fines on the seafloor

No literature was found that assessed the direct impacts of increased sedimentation or deposition of fines from OCS sand dredging and conveyance operations on sea turtles. No mention was made of this potential effect on sea turtles in any of the Biological Opinions or EISs/EAs that were reviewed.

Biological Opinions for dredging in areas adjacent to hard-bottom habitats mention the potential impact of increased sedimentation on hard-bottom and reef communities and seagrass beds, which are important foraging areas for certain sea turtle species, particularly off Florida. Sea turtles that have specific foraging requirements in hard-bottom habitats would be at greatest risk of impact from loss of their forage base, such as green sea turtles that feed on seagrass and macroalgae and hawksbills that feed on sponges. Live bottom has been documented as a preferred habitat of neritic juvenile Kemp's ridleys in the coastal waters of western Florida (NMFS et al. 2011).

Prekel et al. (2008) documented a statistically significant reduction in the percent cover of macroalgae adjacent to nourished beaches in Brevard County, Florida (in State waters) that persisted for over 18 months (pre-nourishment had about 3% cover; at mid-construction it was less than 1% cover; and at 18 months it was about 2% cover). It is important to note that the nourishment project occurred during the very active 2005 hurricane season, and there was extensive beach erosion during the eight major storm events that year. In a parallel study of this project, Makowski and Kruempel (2007) reported a decline of 25% in observations of juvenile green sea turtles (which have strict home ranges) in the area offshore the nourished segment versus a 12% decline in a nearby un-nourished control segment. However, Makowski and Fisher (2008) indicated that juvenile green sea turtles continued to utilize adjacent habitats to the nourished beach; that is, 89% of the turtles sighted (39 of 44) during post-construction surveys were recorded along reefs adjacent to the nourished beach. Makowski and Kruempel (2007) found no statistically significant change in population abundance two years after construction. Although these studies were in shallow State waters adjacent to nourished beaches, the results indicate short-term changes in green sea turtle abundance during and post-dredging, possibly due to reductions in their preferred prey (macroalgae).

There have been numerous studies that have shown that increased sedimentation and water column turbidity from dredging operations and beach nourishment in close proximity to coral reefs can affect corals (see synthesis by Erftemeijer et al. 2012). Many of the case studies presented in this synthesis were dredging and land reclamation operations for coastal construction of ports, resorts, industrial facilities, thus included release of extensive fine-grained sediment plumes and burial of coral habitats. Goldberg (1988) summarized studies in South Florida of dredging activities that damaged stony corals in adjacent patch reef habitat. Therefore, for borrow areas with adjacent sensitive communities such hard-bottom habitats, the approach is to require a buffer around the habitat to reduce the risk of impact. The state-of-the-practice has been to specify a 122 m (400 ft) buffer based on the report "Best Management Practices (BMPs) for Construction, Dredge and Fill and Other Activities Adjacent to Coral Reefs" (PBS&J 2008) which includes this statement:

> "According to Goldberg (1988), the accepted standard distance between a borrow area and hard bottom community in the SEFCRI region is 400 feet. This is a minimum buffer zone between hard bottom and the borrow area, which may be adjusted according to the specific situation and environmental conditions."

This grey literature report has become the standard guidance for setting buffers around sensitive habitats in Florida.

4.6.2.4 Entrainment near the seafloor

Entrainment and mortality are the most serious of the likely impacts for sea turtles during OCS sand dredging projects conducted by TSHDs; they are not at risk of entrainment by CSDs. Based on their foraging habitat preferences, loggerhead, green, and Kemp's ridley sea turtles are mostly at risk of entrainment during dredging operations along the Atlantic and Gulf of Mexico coasts. Loggerheads are also the most abundant sea turtle in this region, making them more at risk. Of the 401 sea turtle takes by USACE dredging projects using hopper dredges in navigational channels from Texas to New York conducted between 1995 and 2008, 70% were loggerheads, 16% were Kemp's ridleys, and 13% were greens (Dickerson 2009). It is important to note the difference between USACE dredging projects, which occur primarily in navigational channels close to shore, and OCS sand mining which occurs at least 4.8 km (3 miles) offshore in areas with less dense turtle abundances.

Table 4.29 shows the historical turtle takes from hopper dredging of OCS sand in the south Atlantic region for the period 1995-2012; there have been no sea turtle takes in the Gulf of Mexico or mid-Atlantic regions during OCS dredging as of February 2013. Nineteen loggerhead sea turtles were taken during 21 projects over the eighteen-year period. For these projects listed in Table 4.29, the sand volumes total 32,423989 yd^3; as of 2013, BOEM has conveyed rights to 73 million yd^3 of OCS sand for 38 coastal restoration projects in six states. Thus, the known sea turtle take rate is one sea turtle per 3.8 million yd^3.

Figure 4.12 shows the annual sea turtle take per USACE project in all U.S. waters for the period 1990-2008, showing a large decrease in the number of turtle takes since 1992, when the listed protection methods were implemented. Navigational dredging poses greater risks of entrainment of sea turtles because of their tendency to concentrate in channels in the southeastern U.S. and bury in mud, presumably to slow their metabolic processes, which is called brumation or reptilian hibernation (Carr et al. 1980). Sea turtles are less able to escape under these conditions because they are partially buried in the sediment and slower to move of the way due to their torpor or reduced metabolic rate. Cape Canaveral Harbor, Florida and Kings Bay, Georgia are two channels where large numbers of turtles have been killed or injured during navigational dredging. During field studies in Cape Canaveral Harbor, turtles resting on or in the sediment were more vulnerable to entrainment by a trailing suction draghead than turtles swimming in the water column above the draghead (USACE 1997).

Table 4.29
Historic turtle take from hopper dredging on the OCS in the South Atlantic for 1995-2013. From USACE Sea Turtle Data Warehouse from projects completed as of March 2013.

Dates	Location	Borrow Area	Sand Volume (yd^3)	Turtle Takes
6/14/95 - 11/9/95	Duval County, FL	Jacksonville	1,240,000	No take
9/11/96 - 11/18/96 12/8/96 - 2/24/97; 3/30/97 - 5/13/97	North Myrtle, SC (Reach 1)	Surfside	2,925,649	5 Loggerheads
10/6/97 - 12/31/97;	Myrtle Beach, SC (Reach 2)	Surfside	1,081,527	2 Loggerheads
8/14/98 - 8/22/98 9/9/98 - 11/24/98	Surfside/Garden City, SC (Reach 3)	Surfside	1,774,279	2 Loggerheads
5/16/98 - 7/21/98	Sandbridge Beach, VA	Sandbridge Shoal	1,100,000	No take
12/10/00 - 2/4/01; 4/4/01 - 4/16/01	Patrick Air Force Base, FL	Canaveral Shoals	577,598	No take
10/1/00 - 2/23/01; 3/8/01 - 4/5/01	Brevard County, FL (North Reach)	Canaveral Shoals	2,800,000	1 Loggerhead
1/13/02 - 2/22/02; 2/24/02 - 4/4/02; 3/28/03 - 4/26/03	Brevard County, FL (South Reach)	Canaveral Shoals	2,800,000	1 Loggerhead
8/13/02 - 12/29/02	Assateague Island State Park, MD	Great Gull Bank	100,000	No take
2/03	Assateague Island State Park, MD	Great Gull Bank	1,800,000	No take
4/02-12/02	Sandbridge Beach, VA	Sandbridge Shoal	2,000,000	No take
12/26/03 - 3/14/04	Dam Neck Naval Facility, VA	Sandbridge Shoal	700,000	No take
3/19/05 - 5/14/05	Brevard County, FL	Canaveral Shoals	2,000,000	3 Loggerheads
6/10/05 - 8/7/05	Duval County, FL	Federal Site	1,500,000	1 Loggerhead
3/7/05 - 3/19/05	Patrick Air Force Base, FL	Canaveral Shoals	400,000	No take
1/10/07 - 3/29/07	Hurricane Ophelia FEMA Sand Replacement, NC	Morehead City ODMDS	1,229,836	No take
11/14/07 - 12/7/07; 12/31/07 - 2/18/08; 3/7/08 - 3/23/08	Myrtle Beach, SC- Part 1	Surfside Little River	778,600	No take
8/14/08 - 1/8/09	Myrtle Beach, SC- Part 2	Cane South	1,442,500	2 Loggerheads
2/13/10 - 3/1/10; 3/22/10 - 4/17/10	Brevard County, FL (South Reach)	Canaveral Shoals	1,300,000	No take
7/8/11 – 8/16/11	Duval County, FL	Duval Borrow Area A	1,200,000	2 Loggerheads
3/13/12 – 4/4/12	Dade County, FL	Southern Government Cut Extension	474,000	No take
4/8/12 – 4/4/12	Wallops Island, VA	Shoal A	3,200,000	No take

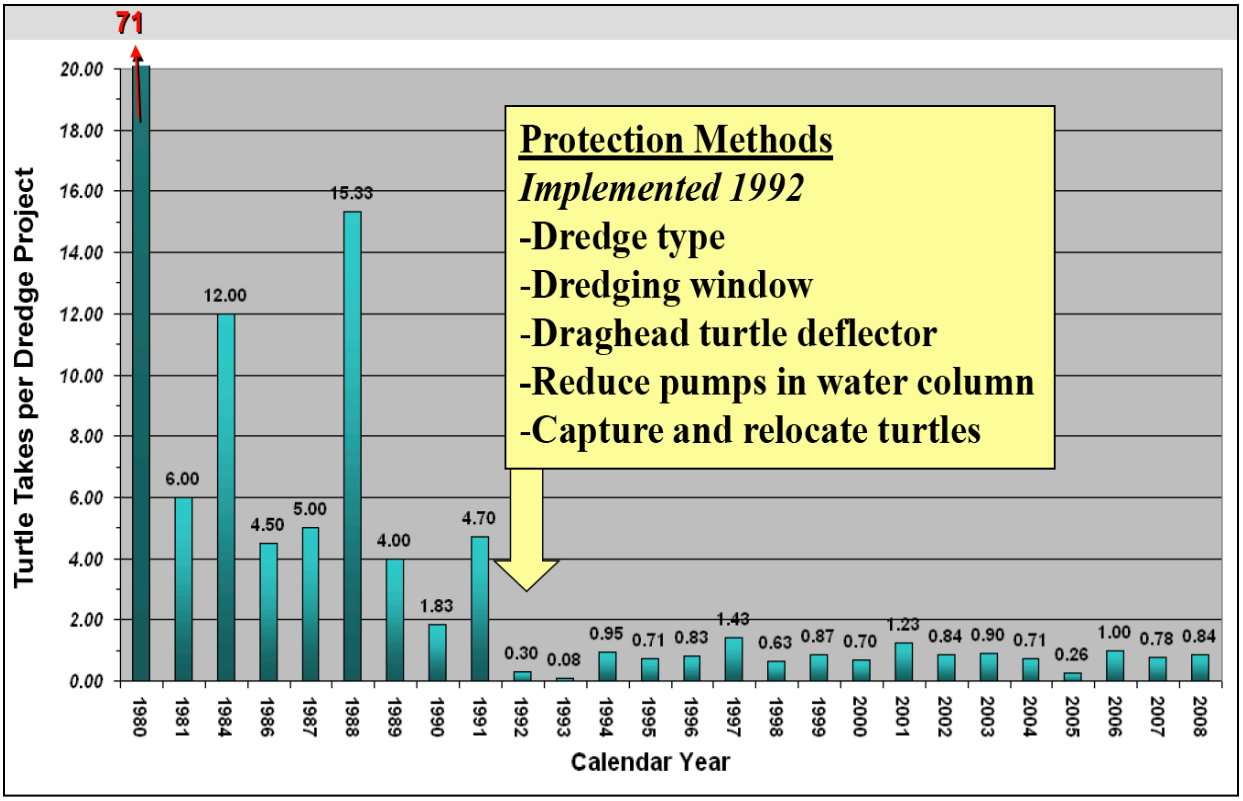


Figure 4.12 Annual turtle takes per USACE project in the Atlantic and Gulf of Mexico regions, showing the effectiveness of protection methods implemented in 1992. From Dickerson (2009).

BOEM-negotiated agreements for OCS sand dredging include measures to protect sea turtles, such as:

- Instruction of all personnel associated with the project of the potential presence of threatened and endangered species, such as sea turtles, and the need to avoid collisions with these animals or harming them in any way.
- During dredging operations, a NMFS-approved protective species observer shall be aboard the dredge to monitor for the presence of sea turtles.
- Where appropriate, relocation trawling may be required.
- Immediate reporting of any take concerning a sea turtle or sighting of any injured or incapacitated sea turtle.
- Hopper dredge dragheads equipped with rigid sea turtle deflectors which are rigidly attached must be used at all times.
- 100% of the hopper inflow must be screened and an observer will monitor the screens.
- When initiating dredging, suction through the draghead shall be allowed just long enough to prime the pumps, and then the draghead must be placed firmly on the bottom. When lifting the draghead from the bottom, suction through the draghead shall be allowed just long enough to clear the lines, and then must cease. Pumping water through the draghead shall cease while maneuvering or during travel to/from the disposal area. Compliance with these requirements is documented using the Dredging Quality Management System.

- The draghead must be buried a minimum of 6 inches in the sediment at all time. Raising the draghead off the bottom to increase suction velocities is not acceptable.
- During turning operations the pumps must either be shut off or reduced in speed to the point where no suction velocity or vacuum exists.

Relocation trawling, where a modified shrimp net is used to sweep the bottom to remove turtles in front of the dredger, has been used as a mitigation strategy to reduce turtle takes. Relocation trawling can affect sea turtles due to the rigors and stress of trawling and on-deck handling. Effects range from raised levels of stressor hormones, to mortality (particularly of cold-stressed or unhealthy sea turtles). Dickerson et al. (2007) reported that, of the 1,239 sea turtles relocated using trawlers for USACE projects over the period 1980 to 2006, four sea turtles died. It is unknown what long-term effects may occur to sea turtles when they are relocated.

There has been very little documentation of the effectiveness of relocation trawling in reducing sea turtle takes. Bargo et al. (2005) presented the results of thirteen USACE dredging projects where relocation trawling was used to catch and release 349 turtles, whereas thirteen turtles were taken by dredges working while the trawling was being conducted. They concluded that, if turtles are abundant in the dredged areas, relocation trawling can be an effective tool in significantly reducing dredge takes during the project.

Dickerson et al. (2007) conducted a detailed analysis of reports on sea turtle takes, relocation trawling, and dredging activities for USACE projects between 1995 and 2006 to evaluate the effectiveness of relocation trawling (the data they used was based on 319 hopper dredging projects where sea turtle observers were used to monitor take, with 70 projects using relocation trawling and 249 projects where relocation trawling was not required). They presented the results using many graphs and tables and various metrics over time and by sub-region. Figure 4.13 shows the turtle takes and relocated turtles and the CPUE using various metrics for USACE projects over the period 1995-2000 where relocation trawling was used. There is a clear relationship between the number of turtles relocated and the number of turtle takes when using various CPUE metrics. Thus, high relocation rates may be a predictor of high sea turtle abundance and potential for takes, but these data do not allow evaluation of the effectiveness of relocation trawling in reducing turtle takes. When evaluating the data by region, Dickerson et al. (2007) noted that relocation trawling did reduce sea turtle entrainment rates in the west Gulf, northwest Gulf, and south Atlantic sub-regions but was least effective for the north Atlantic sub-region (Figure 4.14). They suggested two factors to the increased effectiveness of relocation trawling: 1) Relocation trawling initiated at the onset of dredging resulted in the lowest take rates; and 2) the level of trawling effort used for each project. Dickerson et al. (2007) concluded:

> The effectiveness of relocation trawling in terms of reducing incidental take of sea turtles may not be determined as much by the number of turtles relocated as by the amount of time the trawl nets are able to sweep the bottom as well as when trawling is initiated relative to the onset of dredging reducing incidental take of sea turtles… This study concludes that relocation trawling is an effective management option for reducing incidental take of sea turtles during hopper dredging in some locations provided aggressive trawling effort is initiated either at the onset of dredging or early in the project….. Additional analyses are

critically needed to evaluate the effectiveness of relocation trawling in reducing incidental take rates among varying site-specific circumstances. These more refined evaluations may provide a clearer understanding of the effectiveness of relocation trawling in reducing incidental takes of sea turtles under different conditions of dredging operations. The merits of using relocation trawling as a mitigation tool must also be weighed against human safety and potential trawling-related impacts to the sea turtles or other species captured as bycatch.

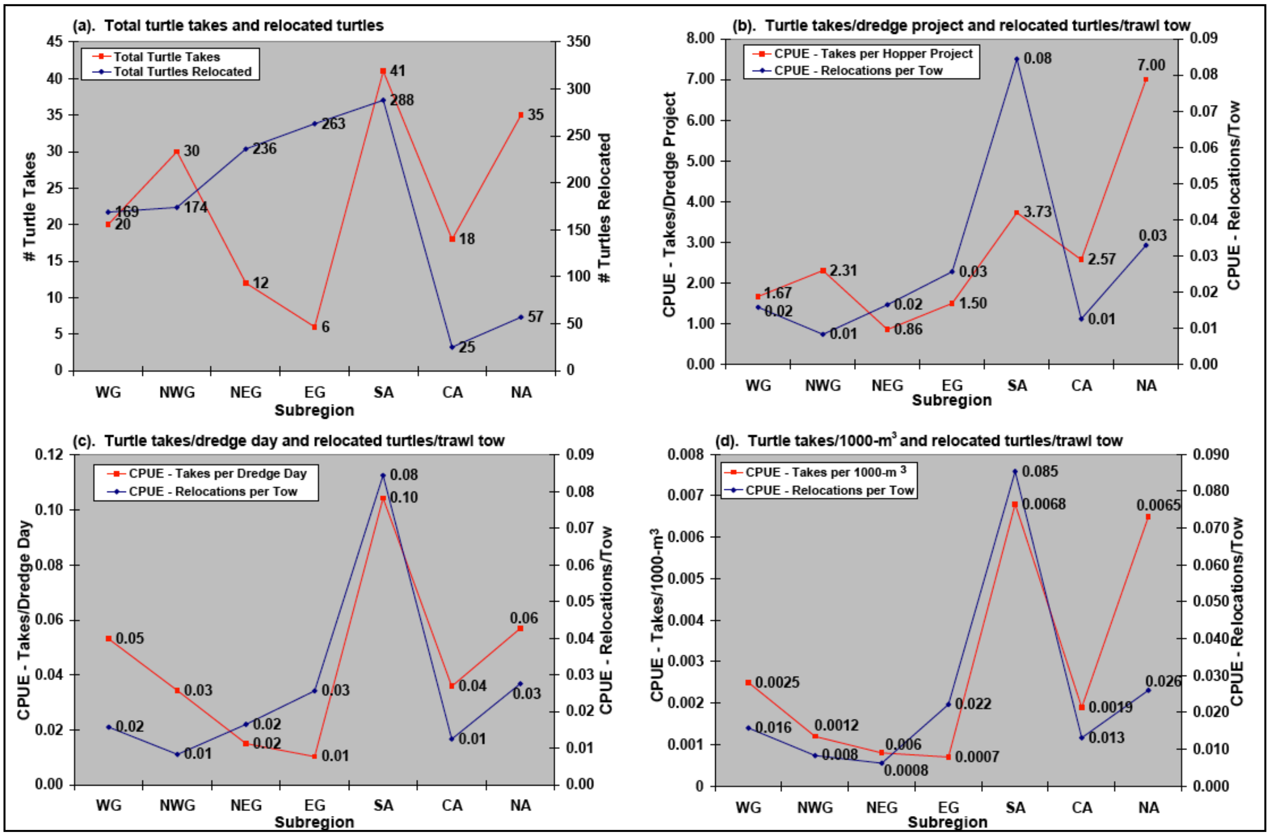
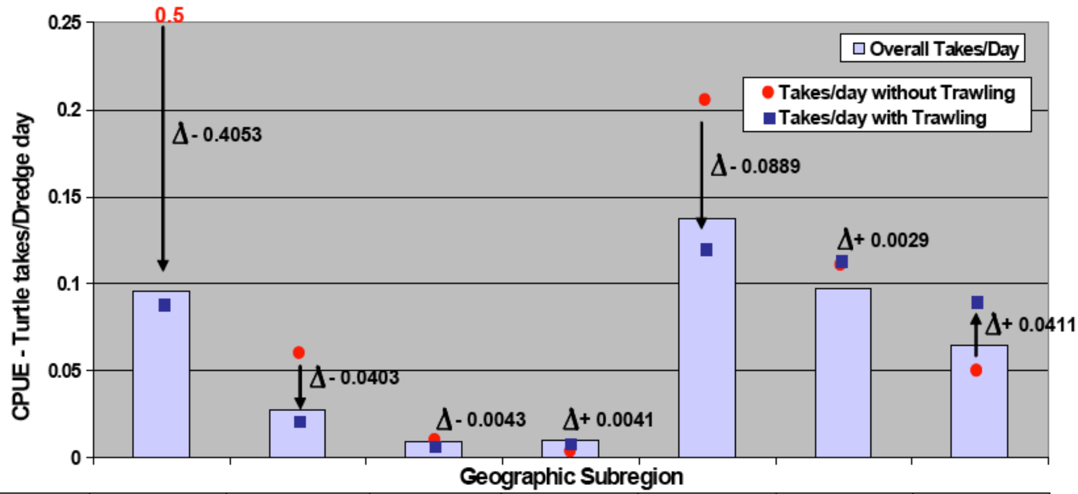


Figure 4.13 Various comparisons of the number of turtle takes and the number of relocated turtles for USACE projects from 1995-2000 using different metrics: (a) total numbers by sub-region; (b) numbers per project by region; (c) numbers by dredge day and trawl tow; and (d) numbers per 1,000 m^3 and trawl tow. WG=west Gulf; NWG=northwest Gulf; NEG=northeast Gulf; EG=east Gulf; SA=south Atlantic; CA=central Atlantic; NA=north Atlantic. From Dickerson et al. (2007).



CPUE Take Rate	West Gulf	Northwest Gulf	Northeast Gulf	East Gulf	South Atlantic	Central Atlantic	North Atlantic
Overall Takes/Day	n = 12 0.0956	n = 13 0.0275	n = 14 0.0095	n = 4 0.0107	n = 13 0.138	n = 8 0.0972	n = 5 0.0652
● Takes/Day w/out Trawling	n = 1 0.5	n = 4 0.0648	n = 4 0.0132	n = 1 0.0064	n = 10 0.2164	n = 6 0.1175	n = 5 0.0547
■ Takes/Day w/ Trawling	n = 12 0.0947	n = 13 0.0245	n = 14 0.0089	n = 4 0.0105	n = 13 0.1275	n = 8 0.1204	n = 5 0.0958
■ - ● = Δ	- 0.4053	- 0.0403	- 0.0043	+ 0.0041	- 0.0889	+ 0.0029	+ 0.0411

Figure 4.14 Effect of relocation trawling on sea turtle CPUE takes per dredge day by sub-region. From Dickerson et al. 2007).

Based on the available information, relocation trawling appears to be an appropriate mitigation tool to reduce incidental sea turtle take in areas with abundant sea turtles present at the time of dredging. However, relocation trawling does have its negative consequences on sea turtles directly, as well as potential impacts to other species captured as bycatch, increase in the dredging duration when sea conditions do not allow trawling, and vessel safety risks from snags and collisions. There was one report in the Sea Turtle Data Warehouse of a capture of a bottlenose dolphin during relocation trawling for the 2006 Collier County, Florida project. Therefore, relocation trawling should be considered where it will be most effective.

Another mitigation measure to reduce sea turtle takes is the use of environmental windows; sea turtle abundance is reduced at water temperatures below 13°C, and they typically are absent during temperatures below 11°C (NMFS 2007). In the Gulf of Mexico Regional Biological Opinion, NMFS and USFWS (2007a) recommended that hopper dredging activities for USACE projects be completed, whenever possible, between 1 December and 15 March for the waters from the Mexico-Texas border to Key West, Florida. The USACE South Atlantic Division Regional Biological Opinions (NMFS 1995, 1997) set the restrictions for hopper dredging activities to the following dates by geography: North Carolina to Pawley's Island, South Carolina – no restrictions; Pawley's Island, South Carolina to Tybee Island, Georgia – 1 November to 31 May; Tybee Island, Georgia to Titusville, Florida – 1 December to 15 April; Titusville to the Florida Keys, Florida – no restrictions. It should be noted that these windows for nesting sea turtles in northeast Florida constrain dredging activities into the time period of North Atlantic right whale critical habitat, which could incrementally increase the vessel strike risk for them.

USACE and BOEM have worked jointly on consultation with NMFS on the re-initiation of the 1997 South Atlantic Coast Regional Biological Opinions for listed species, including sea turtles, potentially affected by dredging projects in the South Atlantic region. USACE has also been working with industry on various approaches to improve the effectiveness of the turtle deflecting draghead, including 1) an adjustable (versus rigid) design that would better maintain contact with the sediment surface, and 2) automatic addition of mixing water to reduce plugging, so the operator would not have to lift the draghead to do so (Dickerson et al. 2004).

According to Dickerson (pers. comm. July, 2012), recent efforts to redesign turtle deflectors such as putting slots on the deflector have been put on hold because there was concern that this design would increase the rate of turtle takes. Turtle deflectors need to have a solid face so they pile up a sand ridge or wave in front of the deflector that pushes the turtle out of the way before it comes in contact with the draghead. The deflector should maintain contact with the bottom and bury up to 25 cm. In 2013, the USACE proposes to conduct computer modeling of different turtle deflector designs prior to field testing (Dickerson, pers. comm.).

Other sea turtle entrainment issues that the USACE are working on include (Dickerson, pers. comm. July, 2012): 1) optimization of dredging windows, which is of particular concern because sea turtle populations are increasing, which can result in increased sea turtle takes during dredging operations; 2) use and design of bed-leveling devices, because most takes occur at the end of dredging operations when the dredged surface is uneven and deflectors are less effectively operated; and 3) mandates for screening on the draghead when dredging in areas of potential UXO, which eliminates the use of screening prior to filling of the hopper to detect turtle takes. The UXO screens are generally 2.5-5 cm, so they would screen out turtle parts as well. Thus, when UXO screening is required, it eliminates the requirement for turtle observers on dredgers.

4.6.2.5 Sound

Sea turtles do not have external ears, ear canals, or a specialized eardrum; sound is conducted through the shell and bone to the inner ear (Lenhardt et al. 1983). Very limited data on the hearing ranges and hearing thresholds for sea turtles are available (Table 4.30). BOEM funded a study in 2010 to measure the hearing sensitivity (develop auditory sensitivity curves) of leatherback sea turtles in air and under water, which can be used to determine if leatherbacks are able to hear, and therefore respond to, sounds produced by marine anthropogenic sources. Based on the current data, the sensitive hearing ranges for sea turtles vary by species and life stage, but they hear low frequencies ranging from 100 Hz up to 1,000 Hz. Older turtles appear to have a narrower hearing range compared to juveniles. There are no data on how sea turtles use their hearing to identify prey, for communication, or detect predators. Lenhardt et al. (1983) suggested that low-frequency sounds emanating from natal beaches might be one cue used by females returning to nest.

Sea turtles can hear frequencies and source levels generated by dredgers during dredging, transit, and other operations (see Section 3.3.5) as these overlap with their sensitive hearing range. However, there are even more limited data on what sound levels cause adverse responses or impacts to sea turtles physiology or behavior. In their recent draft EIS for the Atlantic Fleet Training and Testing, the U.S. Navy (U.S. Navy 2012) stated that no known data are available on which to determine the sound levels causing potential hearing impairments in sea turtles.

Table 4.30
Studies on hearing in sea turtles based on auditory-provoked potential testing except as noted.

Species/Life Stage	Hearing Range (Hz)	Sensitive Hearing Range (Hz)	Hearing Threshold (dB re 1 μPa)	Source/In or Out of Water
Green juvenile	200-700	300-500	-	Ridgway et al. 1969; out of water
Green juvenile	100-800	600-700	94-120	Ketten and Bartol 2005; in water
Green juveniles (2)	100-800	600-700	98-122	Bartol and Ketten 2006 in water
Green juvenile	-	300	-	Yudhana et al. 2010 out of water
Green sub-adult	100-500	600	96-106	Ketten and Bartol 2005; in water
Green sub-adults (6)	100-500	200-400	83-108	Bartol and Ketten 2006; in water
Green 60 year old adult	100-500[1]	-	107-119 @ 220 Hz 121-131 @ 400 Hz	ONR 2012
Kemp's ridley juvenile	100-500	100-200	110-117	Ketten and Bartol 2005; in water
Kemp's ridley juveniles (2)	100-500	100-200	103-117	Bartol and Ketten 2006; in water
Loggerhead juveniles	250-1,000	-	-	Moein et al. 1993; in water
Loggerhead juveniles (36)	250-1,000	250	-	Bartol et al. 1999; out of water
Loggerhead 1 year old	100-900	500	82-97	Ketten and Bartol 2005; in water
Loggerhead 2 year old	100-700	500	86-92	Ketten and Bartol 2005; in water
Loggerhead 3 year old	100-400	300	94-102	Ketten and Bartol 2005; in water
Loggerhead adult	100-1,131[1] 50-800[2]	200-400 100	110 98	Martin 2011; in water

[1] Behavioral testing

Lenhardt et al. (1983) reported startle responses to vibrational stimuli in loggerhead and Kemp's ridley sea turtles, though these responses diminished with habitation to the stimuli. O' Hara and Wilcox (1990) observed changes in swimming patterns and orientation in free-swimming loggerheads in a canal when exposed to pulses from a high-pressure air gun. Mccauley et al. (2000) estimated that the received level at which turtles avoided sound in the O'Hara and Wilcox (1990) experiment was 175-176 dB re 1 μPa rms.

The USACE sponsored several studies to determine the effectiveness of using seismic sources on hopper dredges as a means to repel sea turtles, and thus reduce entrainment, during dredging operations. Initial studies showed some promise; Lenhardt et al. (1994) observed turtles swimming toward the surface of the water when exposed to low-frequency, high-intensity sounds (20 to 80 Hz, 175 to 180 dB). In subsequent studies, Lenhardt et al. (1994) tested five loggerhead sea turtles each, swimming in a net in the York River, Virginia, and in a tank. They noted whether a sea turtle in the net would swim towards or away from a sound projector generating tone bursts of 250, 500, and 750 Hz; for the tank study, frequencies of 100-2,000 Hz were tested. In each test, the sea turtle showed no significant approach or avoidance behavior in response to the sound; that is, each turtle continued in the direction it was headed when the source projector was activated. The USACE concluded that acoustic repellents on hopper dredgers would not be effective at reducing sea turtle mortality. Sea turtles either did not respond to them or became acclimated over a short time (Moein et al. 1993).

McCauley et al. (2000) conducted studies of behavioral responses to sound from an approaching-departing single air gun for 1 or 2 hours at a repetition rate of 10 seconds using caged green and loggerhead sea turtles. The turtles noticeably increased their swimming activity above a received level of 166 dB re 1 μPa rms; swimming time increasing as the air gun levels increased. Above 175 dB re 1 μPa rms, their behavior became more erratic, possibly indicating the turtles were in an agitated state (McCauley et al. 2000).

Bartol and Ketten (2006) conducted studies of sea turtle and tuna hearing to evaluate the feasibility of using sound stimuli to deter sea turtles from approaching tuna longlines and reduce bycatch-related turtle mortalities. They generated much of the current data on in-water sea turtle hearing. However, they found that both sea turtles and yellowfin tuna were low-frequency specialists, and that tuna would hear any deterrent sounds used for sea turtles.

Because of the lack of data on which to assess potential impacts of sound on sea turtles, the U.S. Navy (2012) had to derive acoustic thresholds and criteria for their Draft EIS. Based on the analysis by Finneran and Jenkins (2012), the temporary threshold shift (TTS: temporary reduction of hearing sensitivity) for non-impulsive sounds was estimated to be a sound exposure level of 178 dB re 1 μPa^2-s based on data for mid-frequency cetaceans. They estimated that the permanent threshold shift (PTS: resulting in tissue damage that does not recover and permanent reduced sensitivity to sounds over specific frequency ranges) for non-impulsive sounds to be a sound exposure level that would be 20 dB re 1 μPa^2-s higher than the TTS, for a sound exposure level of 198 dB re 1 μPa^2-s. Based mostly on the studies by McCauley et al. (2000), Finneran and Jenkins (2012) proposed a behavioral disturbance threshold as a weighted sound pressure level of 175 dB re 1 μPa.

The U.S. Navy developed a model, called the Navy Acoustic Effects Model, to estimate the potential acoustic effects of proposed Navy training and testing activities on marine mammals and sea turtles in which they use these three levels of impact criteria: TTS, PTS, and behavioral disturbance (Ciminello et al. 2012). As more field data are compiled for the sound source levels generated by dredging operations, this model may be a useful tool for initial screening of potential effects. The recent measurements of three dredges operating during the Wallops Island, Virginia project were used to calculate maximum source levels of 178.7 dB re 1 μPa at 1 m and an average of about 145 dB re 1 μPa at 50 m (Reine et al. In prep).

In summary, there are limited data on the hearing of sea turtles. However, as of February 2013, there are no data specifically for sea turtles on which to determine the levels of sound that will cause adverse impacts, either temporary or permanent. The results BOEM study on leatherback sea turtles will be very valuable.

4.6.2.6 Vessel operations and interactions

Vessel strikes are a major source of sea turtle injuries and mortalities: 30% of all sea turtle strandings in Florida had propeller wounds (Singel et al. 2007). From 1997 to 2005, 14.9% of all stranded loggerheads in the U.S. Atlantic and Gulf of Mexico regions were reported to have sustained some type of propeller or collision injuries, although it is not known what proportion of these injuries were post- or ante-mortem (NMFS and USFWS 2008). The incidence of loggerhead propeller wounds has risen from approximately 10% in the late 1980s to a record high of 20.5% in 2004.

Although sea turtles spend 90% of their time underwater, they do come to the water surface to breath, warm themselves during the day, and to recover from anaerobic metabolism after deep dives. Hoschscheid et al. (2010) used satellite relay data loggers to track the diving depth, water temperature, and time spent on the surface in ten free-ranging loggerhead turtles in the Mediterranean Sea for up to 450 days. The average "extended surface time" was 90 minutes but such extended periods at the water surface were mostly infrequent and irregular. Most of these periods (82%) occurred during daylight, around noon, suggesting that the turtles came to the surface to warm, likely to enhance digestive processes after feeding at depth. Night-time periods followed deep dives when they would have accumulated lactic acid after anaerobic activity. Thus, sea turtles are at risk of vessel strikes both day and night, though mostly during the day.

Hazel et al. (2007) found that green sea turtles moved away from encounters with a 20-m research vessel 60% of the time at vessel speed of 2.2 knots, 22% of the time at vessel speed of 6 knots, and 4% of the time at vessel speeds >10 knots. Based on this study, sea turtles would not likely move away from dredgers and associated vessels that travel at speeds of up to 17 knots while transiting to the borrow area and up to 15 knots when fully loaded (though vessel speed is restricted to 10 knots at specific areas/times to minimize marine mammal strikes). Thus, the requirement for turtle observers during transits through areas with high sea turtle densities may be an important mitigation method.

4.6.2.7 Water quality

Water quality concerns focus mainly on potential impacts from oil spilled from vessels involved in dredging operations. There are limited data on which to evaluate the effects of exposure to light refined products such as diesel, the type of fuel used on most dredging vessels. Most laboratory experiments have been conducted with crude oils, and most spills that have affected sea turtles were of more persistent crude oils and heavy refined products. Laboratory studies by Lutcavage et al. (1995) using exposure to a 0.05 cm thick layer of South Louisiana crude oil for 96 hours showed the following effects on sea turtles: no avoidance of the oil; disturbance by the fumes; sloughing off of the skin which took 21 days to recover; changes in blood chemistry; and no increase in enzymes in the liver that are involved in chemical detoxification. It is likely that light fuel oil spills in open water will spread quickly into thin sheens, be naturally dispersed into the water column, and result in short-term impacts to sea turtles, mostly in the form of skin, eye, and nasal irritation.

4.6.3 Summary of Known Impacts on Sea Turtles due to OCS Dredging and Data Gaps

Given the available information, some general conclusions can be drawn regarding the potential impacts of mining of OCS sand resources on sea turtles by impacting mechanism, discussed in order of greatest to least impact: entrainment, sediment and biota removal, increased sedimentation/deposition of fines, vessel operations and interactions, and sound. The other potential impact mechanisms are considered to have minimal to inconsequential impacts or have no available data to present.

The greatest concerns are associated with entrainment during dredging operations by TSHDs at the borrow area and the rehandling area (Table 4.31), which usually results in mortality. Mitigation measures to reduce sea turtle mortality during OCS dredging operations include observers monitoring for sea turtles in the area, relocation trawling when necessary, operation of the draghead in a manner which reduces the potential for entrainment of sea turtles, use of turtle deflectors, and 100% screening of the inflow with monitoring to document any sea turtle takes. To date, environmental windows have not been required for dredging at OCS borrow areas, presumably because these areas are not known to have high abundances of sea turtles associated with increased water temperatures. That these mitigation measures have been successful for OCS sand dredging is demonstrated by the facts that there have been 19 loggerhead sea turtles for 38 projects conducted over eighteen years of dredging operations on the OCS with removal of 73 million yd^3 of sand, all in the south Atlantic region. The introduction of these mitigation measures by the USACE in 1992 resulted in a large drop in sea turtle takes during dredging of navigation channels. However, it is difficult to determine the relative effectiveness of the different mitigation measures. The USACE conducted an analysis of the effectiveness of relocation trawling on their dredging projects, indicating that there were fewer incidental sea turtle takes when the effort was aggressive, started early during the project, and continued to the end. It is not known whether this approach would be effective for OCS borrow areas where sea turtle densities and behaviors differ significantly, compared to dredging in navigational channels. As Dickerson et al. (2007) note, there are human safety issues (particularly for OCS projects) and potential trawling-related impacts to the sea turtles or other species captured as bycatch that must be evaluated on a project-by-project basis.

In borrow areas with potential UXO and other munitions, the key issue is the tradeoff between screening at the draghead, to reduce the potential to place these items on the nourished beach, versus screening at the inflow to detect incidental turtle takes. To date, the NMFS has been reluctant to drop the requirement for inflow screening because of the reduced ability to detect takes. The USACE is working on better designs for turtle deflectors, which could be a solution in areas with less-dense populations, such as OCS sand borrow areas.

For all other impacting mechanisms, there are very limited data on which to evaluate the potential degree of impacts from OCS sand dredging operations. Most of the impacts must be inferred from an understanding of the life history, prey and habitat preferences, and behavior of sea turtles.

With little information on the importance of OCS sand borrow areas as sea turtle foraging habitat, it is difficult to assess the impacts of alteration of benthic habitat (Table 4.32).

Table 4.31
Impacting mechanism for OCS dredging on sea turtles: *Entrainment*.

Impact Pathway	*Potential Effects on Sea Turtles*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☐ Cumulative: ☒	Direct Effects Mortality/injury of entrained animals Indirect Effects None known Cumulative Effects Reduction in populations of listed species or distinct populations	Confined to the dredged area footprint, but only for hopper dredges.	Entrainment usually results in death of the sea turtle; sea turtles are long-lived, thus impacts can be long-term.	In practice, entrainment during dredging of OCS sand occurs very infrequently (19 takes in 17 years; see Table T1).	In practice, impacts are minor because of effectiveness of mitigation measures that are employed.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Sea turtle takes are reported and tracked.	1. During months that turtles are present, hopper dredges are outfitted with state-of-the-art sea turtle deflectors on the draghead and operated in a manner that will reduce the risk of direct interactions with sea turtles present in the dredging area. 2. NMFS-approved observers detect interactions with turtles and handle, collect, and resuscitate turtles injured during dredging operations. 3. Relocation trawling is conducted in areas deemed appropriate due to high concentrations of sea turtles in the borrow area.		No studies of the effectiveness of these measures during OCS dredging operations. However, studies by the USACE of dredging in navigation channels have shown significant reductions in sea turtle takes using the suite of mitigation measures since 1992[1,2]. One evaluation of USACE projects where relocation trawling was conducted suggested that this practice was effective in areas of high turtle densities[3].	

[1]Dickerson et al. 2007; [2]Dickerson 2009; [3]Bargo et al. 2005

Table 4.32

Impacting mechanism for OCS dredging on sea turtles: *Alteration of benthic habitat at the borrow area.*

Impact Pathway	*Potential Effects on Sea Turtles*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☐ Indirect: ☒ Cumulative: ☒	Direct: None known Indirect: Altered forage base for species that feed on benthos in borrow areas Temporary disruption of normal feeding behavior Cumulative: Reduced fitness due to reduced foraging or travel to alternative foraging areas if foraging habitat is removed or greatly changed	Removal of prey would be confined to the footprint of activities.	Sand-associated fauna are expected to recover within 1-3 years after a dredging event; repeated dredging could slow recovery, or cumulatively remove the sand habitat[1]	Unknown for specific shoals. There are plans for repeat dredging of shoals at five-year intervals.	Likely to be minor; the importance of OCS sand borrow areas as foraging habitat for sea turtles is not known, but generally they are not considered to be particularly important foraging habitats.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Numerous studies on short-term recovery of benthos post-dredging; but insufficient long-term monitoring to infer temporal duration of benthic community change. Refer to Table 4.2. No information on importance of sand borrow areas for sea turtle foraging[1]	None specific to sea turtles; however, measures to speed recovery of benthic communities and habitats would also reduce potential impacts to sea turtles. Refer to Table 4.2.		Refer to Table 4-2.	

[1]See Table 4.2

Juvenile sea turtles spend years at sea, foraging over large areas. Subadult loggerheads feed in estuaries, lagoons, and the mouths of rivers and bays (Dodd 1988). Kemp's ridleys highly prefer portunid crabs; loggerheads prefer mollusks and crustaceans; greens prefer seagrass and algae; leatherbacks are primarily pelagic and feed on jellyfish; and hawksbills prefer hard-bottom habitats because they feed primarily on sponges. Therefore, Kemp's ridleys and loggerheads would be the two species that could be affected by the alteration of the benthic habitat

Most Biological Opinions conclude that alterative of the benthic habitat at the borrow area is not likely to significantly affect sea turtles because: studies have shown relatively rapid recovery of benthic communities; most dredging projects affect a relatively small portion of the available foraging habitat; and borrow areas are not known to be particularly important foraging habitats for sea turtles. If any of these conditions become important for a specific borrow area, then the conclusion might differ. Any mitigation measures to speed the recovery rates of benthic communities would also benefit sea turtle foraging in these habitats.

For species that have more selective diets, such as green and leatherback sea turtles, there has been concern about reductions in prey items in hard-bottom habitats as a result of increased sedimentation and deposition of fines adjacent to borrow areas (Table 4.33), particularly for green sea turtles in Florida. Green sea turtles feed in nearshore areas with pastures of seagrasses and/or macroalgae, but small green sea turtles can also be found over coral reefs, worm reefs, and rocky bottoms (NMFS and USFWS 1991). Studies of the Broward County Shore Protection Project in 2005/2006 (Makowski and Kruempel 2007; Makowski and Fisher 2008) found that juvenile green sea turtles used adjacent, unaffected habitats and returned to normal abundances within two years post-dredging and beach fill. In Florida, where the risk of impacts to hard-bottom habitats from sedimentation resulting from dredging activities is greatest, there is a requirement for a 122 m (400 ft) buffer around these areas, to prevent damage from direct contact and high sedimentation. This requirement would also reduce potential impacts to sea turtle foraging habitat. It also reduces the risk of entrainment of juvenile greens, which use these habitats for foraging and resting.

Vessel strikes during dredging and vessel transits could result in sea turtle mortalities (Table 4.34). Dredging operations can include multiple support and survey vessels, most of which travel at fast speeds between the borrow area, the placement site, and port. One study showed most sea turtles do not move out of the way of vessels traveling at speeds greater than 10 knots. Dredges travel at speeds up to 17 knots, though this speed varies by dredge size and propulsion system, and they travel ~2 knots slower when loaded. Although there are no reported sea turtle strikes by vessels associated with dredging activities, it would be difficult to use sea turtle stranding data to identify the source of sea turtle strikes by activity. The full suite of mitigation measures for reducing sea turtle takes during dredging has been shown to be effective; however, it is not possible to determine the effectiveness in reducing vessel strikes because there are no reports of any sea turtle interactions or strikes during any phase of OCS dredging operations.

The increase in the level of sound in marine environments has been a growing concern, not only for marine mammals, but also for sea turtles. Being low-frequency specialists, sea turtles will hear broadband sounds generated by dredging operations (Table 4.35). Thus, the main uncertainty in assessing the potential effects of increased sound during dredging operations is the

Table 4.33

Impacting mechanism for of OCS dredging on sea turtles: *Increased sedimentation/deposition of fines*.

Impact Pathway	*Potential Effects on Sea Turtles*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☐ Indirect: ☒ Cumulative: ☒	Direct Effects None known Indirect Effects Altered forage base for species that feed in adjacent habitats, esp. hard bottom and seagrass Cumulative Effects Reduced fitness due to reduced foraging travel to alternative foraging areas	Greatest risk of impact is to adjacent hard-bottom habitats that are important for green sea turtle foraging.	Will depend on the recovery of any affected prey items.	Unknown, but not likely for most OCS borrow areas; exceptions may be for sites adjacent to hard bottom.	Likely to be minor because sea turtles can move to adjacent habitats.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Studies in Florida State waters showed reduced numbers of juvenile green sea turtles in nearshore habitats shortly after dredging (they moved to adjacent habitats) but returned to normal in 2 years, which may have been related to reductions then recovery of macro-algae cover [1,2,3,4]	None specific to sea turtles; however, buffers to reduce damage from increased sedimentation in hard-bottom communities would also reduce potential impacts to foraging sea turtles.		Uncertain until completion of further compelling studies or necessary field tests. Monitoring during and after the South Government Cut dredging project showed increased sedimentation and coral stress at locations within 400 ft and buffers had to be expanded[5]	

[1]Makowski and Kruempel 2007; [2]Makowski et al. 2008; [3]Makowski and Fisher 2008; [4]Prekel et al. 2008; [5]MDCDERM 2010

Table 4.34
Impacting mechanism for OCS dredging on sea turtles: *Vessel operations and interactions*.

Impact Pathway	*Potential Effects on Sea Turtles*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☒ Indirect: ☐ Cumulative: ☒	Direct Effects Increased risk of vessel strikes, thus injury or mortality Indirect Effects None known Cumulative Effects Reduction in populations of listed species or distinct populations	Vessel operations extend from ports to the borrow area, and transits between the borrow area and placement site. Will include dredgers, support vessels, and survey vessels.	Vessel operations can be conducted 24 h per day, over periods of months.	Not known.	There are no reported sea turtle strikes during actual dredging operations, when speeds are slow enough for sea turtles to move away or avoid serious injury. During transits, vessel speed could result in strikes.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Limited data on movement of sea turtles away from vessels based on vessel speed, showing that sea turtles do not move away from vessels at speeds >10 knots[8].	1. NMFS-approved observers to note presence of sea turtles in the vessel path 2. Training of vessel operators on actions to take to avoid vessel strikes of sea turtles (e.g., attempt to maintain a distance of 50 yards or greater between the animal and the vessel whenever possible)		USACE studies have shown large reductions in sea turtle takes from the full suite of mitigation measures[2,3]; there are no studies that evaluated the effectiveness of observers or operator training in reducing vessel strikes.	

[1]Hazel et al. 2007; [2]Dickerson et al. 2007; [3]Dickerson 2009

Table 4.35
Impacting mechanism for OCS dredging on sea turtles: *Sound*.

Impact Pathway	*Potential Effects on Sea Turtles*	*Spatial Extent of Impact*	*Duration of Impact*	*Frequency of Impact*	*Severity of Impact*
Direct: ☐ Indirect: ☒ Cumulative: ☐	Direct Effects None known Indirect Effects Behavior changes, displacement Cumulative Effects None known	Increase sound above background can be detected a few kilometers from the source, but the greatest sound levels are in close proximity to the source[1,2].	Dredges generate broadband sounds at the source up to 160-180 dB during dredging and transiting[3]. At 50 m from the source, SPLs ranged from 128.9-144.9[3]. Dredging can be conducted 24 hours per day, over periods of months. However, impacts would be short term, as the vessel transits past individual sea turtles.	Depends on habitat; could be a one-time event during transits or several times per day for species with small foraging ranges, such as green sea turtles	Sea turtles will hear sounds during dredging and transiting, however, potential impacts are unknown.
	Availability of Empirical Information	*Mitigation Measures*		*Mitigation Effectiveness*	
	Sea turtles hear low-frequency sounds (100-1,000 Hz, with highest sensitivity at 100-700 Hz) with hearing thresholds of 82-131 dB, depending on species and life stage[4]. However, there are no sea turtle-specific data on which to determine at what levels adverse effects might occur.	None known at this time.		No mitigation measures proposed at this time.	

[1]Saunders and Roberts 2010; [2]Tillin et al. 2011; [3] Reine et al. In prep; [4]See Table 4.30.

lack of data on the levels causing adverse impacts to sea turtles. Another highly variable factor would be the frequency of exposure to elevated sound during dredging operations. One could imagine that exposure could be a short (minutes) one-time event for a sea turtle in the open ocean from the passage of a dredger in transit from the borrow area to the placement site. It could also be a 3-4 times a day exposure as the dredged materials are off-loaded at the placement site or a rehandling area, for a sea turtle with a small foraging area, such as juvenile sea turtles off Florida.

Based on the literature reviewed in this section, listed below are key data gaps in our understanding of the potential impacts to sea turtles from OCS dredging operations and recommendations for studies or syntheses to address these gaps.

- Entrainment poses the greatest risks to sea turtles; however, there are no studies to determine the need for and effectiveness of the different mitigation measures during OCS dredging operations. An analysis, similar to that conducted by Dickerson et al. (2007) for USACE dredging projects in navigational channels, would provide more data on which BOEM and NMFS could evaluate the effectiveness of these mitigation measures.
- Relocation trawling is one mitigation measure that needs further evaluation. Relocation trawling is expensive, involves complex logistics during implementation, and poses additional safety hazards for small vessels towing nets in open water and over long periods. Also, there is no information available on the bycatch mortality associated with relocation trawling during OCS dredging operations or impacts to sea turtles during trawling and handling on deck. When relocation trawling is conducted, it is suggested that the observer reports include information on bycatch mortalities, particularly for key species of concern. The existing data on number of turtles captured during relocation trawling during OCS dredging operations should be evaluated and compared with turtle take rates for different regions. With such information, BOEM and NMFS could better evaluate the tradeoffs of relocation trawling during OCS sand dredging operations by region.
- Improvements are needed in the design and operation of turtle deflectors, to further reduce sea turtle takes. BOEM should continue to coordinate with USACE on their Sea Turtle Research Program.
- There are few data available on how sounds at the frequencies and levels generated by dredging operations might affect sea turtle behavior in OCS borrow areas. Available data for dredging operations in the U.S. and other countries indicate that the sound levels are not much different than other large vessels. As our knowledge of the sounds generated during OCS dredging operations increases, these results can be evaluated as more data on how sea turtles respond to sounds are available.
- The U.S. Navy Acoustic Effects Model may be a useful screening tool for BOEM to evaluate potential acoustic impacts to sea turtles during dredging operations. However, the model results will be limited because there are no sea turtle-specific data on the sound levels that cause behavioral changes, avoidance, or injury.

4.7 IMPACTS OF OCS DREDGING ON ECOLOGICAL INTERACTIONS AMONG BIOLOGICAL RESOURCES

4.7.1 Introduction

The mining and relocation of sandy sediments from the OCS to the shorezone not only physically alters the immediate seabed, it also transmits a perturbation that may modify species interactions throughout the ecosystem on local and perhaps broader spatial scales. Because sand habitats in the OCS are highly dynamic environments that experience frequent disturbances from storms, biotic disturbance to the bottom, and changing climatic conditions, the organisms that occupy these OCS ecosystems have adapted to disturbance on the scales of these natural processes. Dredging and or trawling the seafloor for the purposes of mining and fishing are among the anthropogenic disturbances that commonly impact the coastal oceans of almost every continent. The disturbance of storms to the biota is modest compared to the disturbance from dredging for sand, which simultaneously extracts and kills such a large fraction of the benthic animals where the dredge passes. Trawling for fish is less damaging than dredging for shellfish because the depth of penetration into the sediments is greater for shellfish dredges. In this chapter we begin to identify and address the direct and indirect impacts to the U.S. continental shelf ecosystems that result from the harvest of sand resources, to enhance management of our Nation's public trust resources in a deliberate and sustainable manner.

To discuss ecosystems with clarity, spatio-temporal scales must be defined. Ecosystems can be modeled across a continuum of spatial scales. The largest ecosystem scale in the ocean typically examined is that of the Large Marine Ecosystem (LME) (Sherman 2001). LMEs are biogeographic regions of the ocean that contain relatively homogeneous groupings of species within environmental zones that differ from one another. One LME is identified as the entire South Atlantic Basin, for example. The boundaries between adjacent LMEs are not absolute without connectivity across them, but each LME possesses some level of intrinsic uniqueness containing organisms that largely share the same biogeographic distribution and interact broadly within the LME. Temporally, the boundaries to LMEs can be and are broached by seasonally migrating species, such as many seabirds and whales, thereby having potential to propagate impacts among LMEs. When ecosystem studies are conducted, the "ecosystem" is practically defined, often around a spatial scale associated with a particular habitat, such as the coral reef ecosystem in a particular location, such as the Florida Keys. The spatial scale is chosen such that data can be gathered that meaningfully include the major interactors and quantify energy flows among them (Figure 4.14). Choosing such scales does not imply that the impacts extend across the entire LME: they almost always do not, unless the impact process being studied has spatio-temporal cumulative effects on the entire LME or else some especially mobile organism that ranges over the entire LME migrates to and makes critical use of resources and habitat on a very small spatial scale. This may include the breeding and calving grounds of some great whales, such as North Atlantic right whales using the coastal ocean from about Jacksonville, Florida through Georgia; during warm months, this protected species migrates to feeding grounds as far north as Nova Scotia. The longevity of affected organisms influences the temporal scales of impacts. For benthic invertebrates that can recover in months to a few years, temporally cumulative impacts would be highly unlikely, whereas for the North Atlantic right whale with a clutch size of at most one calf annually, temporal impacts could accumulate over years to affect the actual population viability. Even storm impacts would be unlikely to affect the composition of the entire LME because they would be experienced on smaller scales, except perhaps for some hurricanes. This discussion of spatio-temporal scales of ecosystems relates to OCS sand mining through defining scales at which impacts may occur. All known scales of ecosystem impact from sand mining studies are small-scale, extending spatially only to the extent that mobile consumers, such as demersal fishes displaced from feeding on benthos in the borrow area, range. It is possible, but highly unlikely on the basis of what is now known, that sand mining on certain habitat features over a large geographic area of the OCS, could induce an impact that is detectable at the LME scale.

Because the most immediate impacts of dredging for sand fall upon the benthic invertebrates on the seabed, and the greatest mortality and biomass losses are probably those of the benthos (see Section 4.2), consideration of mechanistic pathways from sand mining to ecological interactions among various biological resources at OCS mining sites should include assessing both direct and indirect consequences of modifications of benthic resources, communities, and habitats. Although benthic substrates function as refuge for juveniles and spawning habitat for adults of numerous marine organisms, perhaps the most important impact from dredging is the perturbation of the food web. This perturbation is best characterized as inducing potential impacts of unknown scale and persistence that depend on disturbance intensity and duration, the spatial scale of the affected food web, natural temporal dynamics, roles of keystone species, intrinsic system resilience, capability of predators to exhibit adaptive switching of prey in their diets, and other factors. The persistence of the effects caused by extraction of sediments and its benthos are best understood within the context of disturbance theory and ecological succession to produce a synthetic general model of benthic habitat impacts, which must then be associated with consequences to higher trophic levels (Oliver et al. 1977; Rhoads et al. 1978; Bolam and Rees 2003; Hill et al. 2011). It is the energy or biomass flow from the benthos to fishes, crabs, and shrimps and from them to higher trophic level animals that environmental policy should aim to protect. Dredging impacts to fish and invertebrate species are described in Sections 4.2 and 4.3. We note that there is insufficient understanding of how benthic resource and habitat changes affect critical ecosystem services provided to fishes, crabs, and shrimps. For example, if dredging transforms the character of bottom substrate by replacing sand with mud or, mud with sand, creating a very different macrobenthic community, we do not know which benthic community serves better to provide prey for the demersal predators, many of which are commercially and recreationally fished species.

A simplistic conceptual food web model of a hypothetical borrow area on the OCS can be used to examine which components (boxes) and processes (arrows) within an ecosystem (or a segment of an ecosystem) may be impacted by dredging (Figure 4.15). Immediately obvious is the simplification of the model relative to the real ecosystem complexity; however, modeling is a tool that allows one to focus on relevant ecosystem components (biotic or abiotic) and processes, and to suggest whether additional interactions within a given system should be examined. Here, each box represents a grouping of many taxonomically and trophically similar species and the arrows represent the flow of materials or energy (typically carbon or biomass in food webs) that is transferred by consumption of prey.

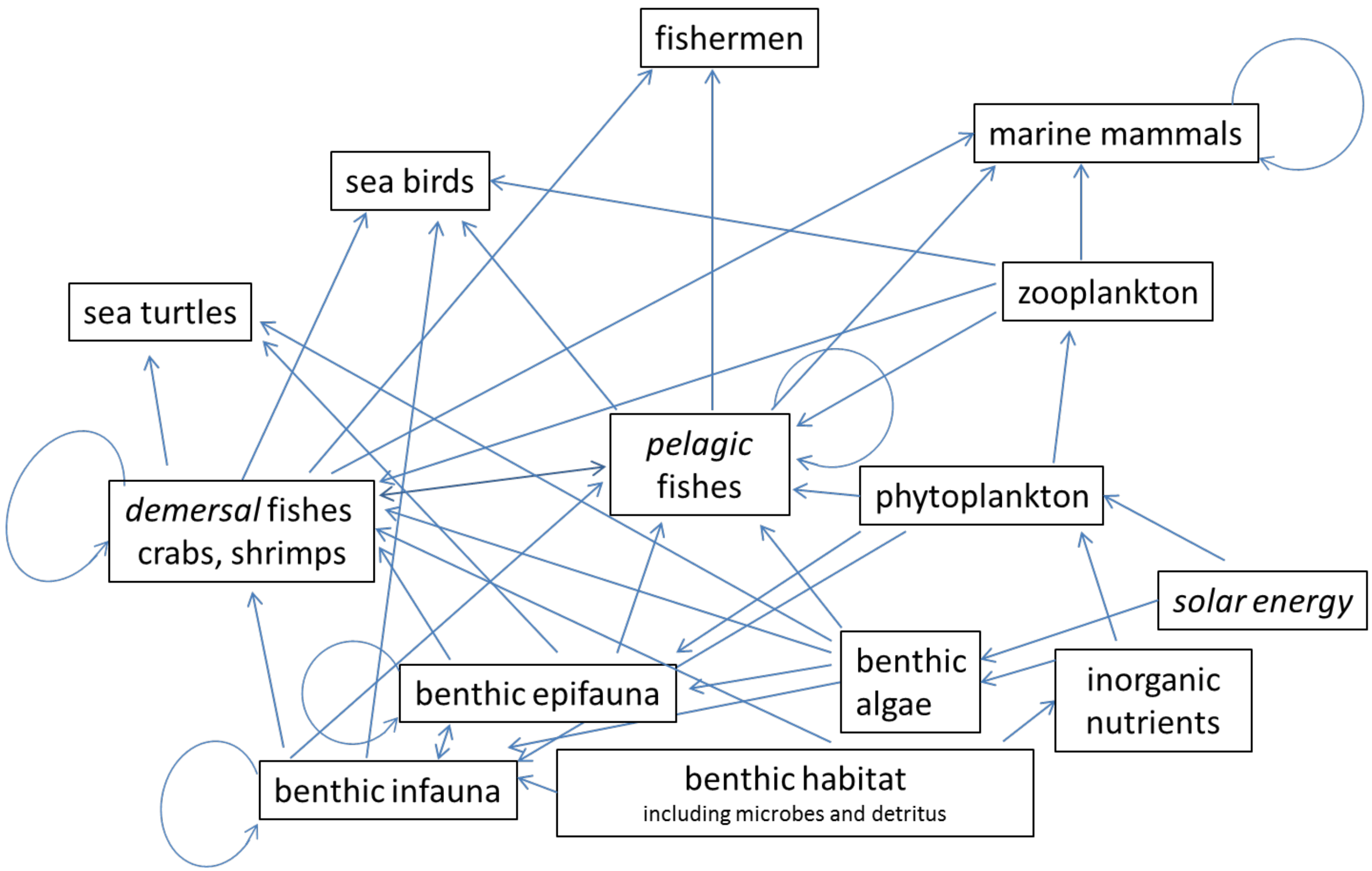


Figure 4.15 A generic conceptual food-web model of an U.S. Atlantic or Gulf of Mexico OCS borrow area. The spatial and temporal extents of models vary according to research questions and management needs.

A myriad of interactions occurs in natural marine food webs. Because of ontogenetic changes in size over multiple orders of magnitude and consequently in diet throughout species' lifespans, not only do most large pelagic predators, such as king mackerel, consume adult menhaden, but juvenile menhaden in turn consume larval stages of these predatory fishes. Figure 4.15 assumes the trophic relationships of adult organisms, except for some of the zooplankton. A similarly counterintuitive food web link is represented by circular arrows in Figure 4.15 that can represent cannibalism; this loop also represents interspecific consumption here. The importance of indirect interactions arising from direct dredging impacts can be illuminated by food-web models. For example, Baird and Ulanowicz (1989) in their network analysis of the Chesapeake Bay recognized the dissimilarity in diets and in the ultimate sources of energy for two important pelagic predatory fish–striped bass and bluefish. Although bluefish consume some fish and some benthic invertebrates and striped bass likewise consume many fish and some benthic invertebrates like blue crabs, the ultimate source of carbon passing upwards to these two predatory fish is derived from dramatically different sources. Bluefish rely strongly on carbon passing through benthic organisms (63% benthic bacteria and 48% polychaetes), whereas adult striped bass are largely dependent on pelagic phytoplankton, microzooplankton, and mesozooplankton (Baird and Ulanowicz 1989). In brief, food webs can facilitate an understanding of the links that comprise benthic-pelagic coupling (e.g., Rosenberg 2001), which can be important in tracing any indirect effects of sand dredging that are relayed up the food web. Food-web models reflect the interactions within a given ecosystem to the degree to which available data allow. Model scale is dependent upon the questions or hypotheses that are driving the investigation. Examining an ecosystem, or part of an ecosystem (if data availability necessitates), via models can also reveal where additional research is needed. Several such data needs are identified with suggested research focus in Section 5.0 of this report. The temporal and spatial scale of models may be constrained by a limited database as related to impacts of dredging activity. The selection of various types of ecosystem models for use in understanding and predicting the impacts of dredging disturbance is discussed further in Section 4.7.3 below.

Information is lacking to allow us to predict how modifications of bottom topography by mining sand on shoals and ridges will locally or regionally influence the interactions with fishes, crabs, and shrimps that may be targeting these habitats for reproductive processes such as mass spawning or using shoals to guide migrations (see Sections 5.1 and 5.2). Nevertheless, if maximum shoal height is restored by natural processes after dredging has extracted sand, as indicated for shoals in <30 m water depths in the mid-Atlantic region by modeling done by Dibajnia and Nairn (2011), then the effects on spawning and reproduction and on migration of exploited demersal predators that use shoals would be temporary–although with rate of recovery of maximum vertical height being very site-specific as a function of waves, currents, water depths, and dredging pattern (see Section 2.3; Dibajnia and Nairn 2011). If recovery of maximum height restores the functionality for facilitating spawning and reproductive activities and in guiding migrations, then effects of extraction of sand on peak shoal elevation would not be permanent, and may not qualify as a major effect. Yet, the seriousness of such an impact may need to be judged by combining site-specific data and population modeling to infer population-level consequences of temporary reductions in maximum shoal height. In addition, severity of impact is dependent upon the ecological uniqueness and connectivity of the habitat(s), as well as, the combined impact of cumulative stressors. For example, in regions where sand ridges are abundant, such as in the inner shelf of the Mid-Atlantic Bight, periodic dredging of few ridges likely has little impact on fish populations. However, where benthic habitat types are more heterogeneous, and sandy bottom is relatively rare, such as on the Southeast Florida shelf, sand ridges likely fulfill a unique ecological role. To the degree that sand shoals may more widely become classified as EFH because of population-level importance of impacts on spawning and reproduction, the impact rating for effects of dredging on spawning and reproduction of demersal fishes may be elevated to major, especially for such shoals in waters deeper than 30 m, where bottom energetics are insufficient to restore maximal elevation (Dibajnia and Nairn 2011). Information is presently inadequate to judge the population-level importance of modifying spawning and reproduction by shoal dredging at any depth. Although emergent sand shoals have been proposed to serve as topographic guideposts to seasonal migrations of fishes, blue crabs, or shrimps, even less is known about whether this interaction between shoal topography and migration exists, let alone possible impacts of shoal dredging on the process. See Section 5.4 for research proposed to address this data gap.

Another potentially key ecological interaction that is currently incompletely explored is whether sand shoals, particularly in the mid-Atlantic OCS region, are important habitat for foraging seabirds in winter and during migrations (see Section 4.4). Research on winter seabird distributions suggests that they use shallower sand shoals of the inner shelf, but more fine-scale data are needed to confirm this (Geo-Marine Inc. 2010; Silverman et al. 2011). Current research also indicates large temporal and spatial variation in use (that is, little site fidelity across years); however, little is known about whether these patterns reflect variations in prey distributions, climate, weather, or combinations thereof.

We discussed earlier (see Section 4.2.2.3) how impacts of siltation arising from turbidity caused by dredging projects and by deposition and remobilization of sand at rehandling sites could affect hard bottom, coral reefs, and their ecological support of valuable reef fishes and sea turtles, which would elevate the level of concern over impacts to the hard-bottom habitat alone.

In addition to considering how OCS dredging modifies ecological interactions that arise from direct and indirect effects on the benthic invertebrate resources, communities, and habitats, some other impacts of dredging have potential to trigger changes to ecological interactions. For example, a full consideration of sand mining impacts on ecological interactions would include assessment of potential cumulative impacts of sand mining in combination with other processes that modify the benthic resources, communities, and habitats–predominantly fishing impacts, but perhaps also renewable energy developments in the future. Furthermore, as the concept of ecosystem-based management of resources and habitats has become increasingly endorsed by land managers and natural resource managers (Christensen et al. 1996), models of ecosystem dynamics have been developed and applied to produce insights into how various factors influence ecological interactions directly and indirectly and, thereby, affect resources and habitats of concern to coastal zone and fisheries managers because of their high ecological value. Here we review the literature on ecological impacts of bottom-disturbing fishing gear and of exploitation of fish and place effects of sand mining into a broader, more integrative context to infer cumulative impacts of relevance to resource management. We also briefly review the use of energy-flow models and food-web models of OCS ecosystems to extract insights into how useful this approach may be in capturing how ecological interactions, especially trophic, may create networks of indirect impacts of specific ecosystem changes caused by OCS sand mining.

4.7.2 Impacts of Bottom-Disturbing Fishing as a Proxy for Sand Mining Disturbance and Cumulative Effects of Both

Sand mining on the OCS takes place within an environment already greatly affected and transformed by many other ongoing human interventions, the most widespread and consequential being commercial, and to a lesser extent, recreational fishing. Commercial fishing modifies the coastal ecosystem in several fundamental ways (Botsford et al. 1997): 1) modifying abundances, size- and age-frequencies, and even genetics of targeted stocks; 2) reducing densities of other species, including seabirds, sea turtles, and marine mammals unintentionally caught and discarded as bycatch; 3) disturbing the seafloor habitat, creating changes to the sediment surface structure, causing turbidity, and modifying geochemical processes; and 4) inducing large modifications to benthic macrofaunal communities and thus to seafloor habitats and to prey for demersal predatory fishes, crabs, and shrimps. Bottom-disturbing fishing gear varies in depth of sediment penetration and, therefore, in scope of impacts on infaunal invertebrates, with the most damaging being heavy dredges and hydraulic dredge gear, followed by bottom trawls, which can include trawl doors and tickler chains that penetrate into the sediments, physically harming and extracting benthic invertebrates (Collie et al. 2000).

Dredging for sand excavates to deeper depths into the sediments than either type of bottom-disturbing fishing gear, trawls or dredges and uptakes and kills benthos, while fishing gear causes almost as much mortality of macrobenthic infauna but not by direct uptake. Dredging sand covers a far smaller OCS area than bottom-disturbing fishing gear, although to an unknown extent, and fishing disturbance is repeatedly applied even in a single fishing season, whereas dredge passes in sand mining are not annually superimposed. The bycatch injured by fishing gear, discards that sink, and the physically damaged and excavated invertebrates exposed on the seafloor attract demersal scavengers and predators (Caddy 1973), including benthic invertebrates, such as gastropods and crabs, as well as some demersal fishes. Although these subsidies to scavenging food chains end when a fishing season ends, enough biomass can be provided during open fishing seasons to enhance densities of such scavenging fishes over periods of years of seasonally repeated bottom disturbance and bycatch creation (Demestre et al. 2000).

Some of the impacts propagate further as indirect effects from this enhancement of populations of scavengers through seasonal food subsidies (Jennings and Kaiser 1998). For example, larger gulls are more aggressive and thus more successful scavengers of fish discarded by trawl fishermen onto the sea surface. Enhancement of abundances of these larger gulls via their consumption of trawl, dredge, gill net, and long-line discards can lead to their increased predation on eggs and chicks of other, typically smaller seabirds at nesting sites. By extension, similar indirect effects are likely to occur through enhancement of abundances of pelagic and demersal scavenging fish, but these subsurface scavengers are more difficult to study, leading to less direct, empirical evidence of consequences of subsidizing their diets and augmenting their abundances. Nevertheless, a substantial body of data demonstrates that fish extractions have selectively removed the highest trophic levels and largest fish (“fishing down the food web”: Pauly et al. 1998), which can lead to indirect effects through trophic cascades influencing species lower in the food-web, including species that support fisheries themselves (Myers et al. 2007).

The impacts of bottom-disturbing fishing gear vary greatly with geographic location, driven by changes in availability and value of available fishery resources, the presence of geological features that inhibit successful trawling or dredging, and various policies, laws, and regulations, including those affecting choice of fishing gear. The continental shelf of the Gulf of Mexico is especially intensely trawled for shrimp, with that trawling extending further offshore on the shallow continental shelf than on most of the Atlantic southeast coast. Commercially viable populations of penaeid shrimps extend northwards only through North Carolina, but flounder trawling and scallop and clam dredging represent locally intense fisheries further north and on both coasts of Florida that induce intense bottom disturbance and consequent ecological impacts on the benthic invertebrate community directly and on scavengers and predators indirectly.

The impacts of bottom-disturbing fishing gear on benthic invertebrate communities can be summarized now with reasonable confidence after completion of over 120 scientific studies during the past 20-30 years as concern over ecosystem-based fishery management has developed (Jennings and Kaiser 1998; Hall 1999; Watling 2010). Fishing with trawls and dredges impacts the composition of benthic invertebrate communities by selectivity of removals. Large epibiota is preferentially removed and killed, including many long-lived clonal invertebrates that provide emergent structural habitat, such as sponges, erect bryozoans, hydroids, soft corals, and stony corals (Collie et al. 1997; Sainsbury et al. 1997). Where waters are shallow enough to support seagrasses and macroalgae, they too are displaced by bottom-disturbing fishing gear (e.g., Peterson et al. 1987). Because of their life histories, the time scales of recovery from disturbance for these biogenic habitat providers is far longer than the interval between fishing seasons, meaning that these groups of organisms have become widely depleted over vast areas of the continental shelf where trawling is intense.

Sainsbury et al. (1997) inferred that recovery of sponge and coral habitat damaged by bottom trawling in Australia would require more than fifteen years free from bottom-disturbing fishing. When intact, such emergent, biogenic habitat functions to provide settlement sites for many fishes (Auster 2008) and offers physical, and perhaps chemically defended, shelter for juvenile fishes and crustaceans from predation. Trawling and dredging collapse the burrows created by various crustaceans. Such burrowers are also “ecosystem engineers,” providing subsurface, oxygenated habitat structure for commensal invertebrates and smaller, often juvenile fishes that occupy the burrows and burrow walls. Additionally, trawling preferentially destroys tube-builders, such as the polychaete *Lanice* in UK waters, which function to enhance benthic invertebrate abundances and biodiversity through stabilization of otherwise mobile sediments. Reise and Schubert (1987) resampled areas of the Wadden Sea after sixty years of intense bottom-disturbing fishing, finding that the clonal tube-building polychaete *Sabellaria spinulosa* and oyster reefs had been virtually eliminated, whereas mobile polychaetes had increased greatly. Emergent tubes of tube-building polychaetes also contribute to habitat complexity.

The loss of habitat complexity following bottom disturbance by fishing gear leads to depressed biodiversity of fishes and other taxa, with potential consequences on fisheries yields (Auster and Langton 1998; Norse and Watling 1999). The benthic species that have come to predominate in sediments that are repeatedly disturbed by fishing with trawls and dredges are smaller, short-lived opportunistic polychaetes and other taxa that have high reproductive rates. This fishing-induced substitution of abundant, small-bodied, shallow-burrowing invertebrates in place of emergent, long-lived epibiota and long-lived infaunal bivalves is widely hypothesized to promote production of shrimp, which prey on small, surface-dwelling invertebrates. Shrimp support valuable fisheries, especially in the Gulf of Mexico and southeast Atlantic. Consequently, the bottom disturbance from trawling for shrimp and demersal fishes cultivates the bottom in favor of small opportunistic invertebrate prey for shrimp, probably enhancing that fishery at the expense of a diverse fish community and fishery on finfish species associated with biogenic structural habitat provided by the historic baseline community of epifaunal and burrow-constructing ecosystem engineers. This hypothesis has yet to be tested rigorously.

Impacts of repeated intense fishing by trawls and dredges vary in intensity among different sedimentary habitats on the seafloor, mainly as a consequence of differing recovery rates of the species dominating the benthos. In sandy sediments that undergo regular reworking, invertebrates tend to be short-lived, opportunistic species, including especially polychaetes and amphipods and other small crustaceans (Brooks et al. 2004; 2006), which are the very ones that recover most quickly from disturbances, including bottom disturbance by fishing gear. Thus, the sandy bottoms can show relatively rapid recovery of total abundance and total numbers of species of benthic invertebrates after trawling or dredging (Brooks et al. 2004; 2006). Nevertheless, recovery times for total abundance of benthos after bottom disturbance from fishing ceased was never less than 3 months and the most reliable studies showed about 16.5 months durations for recovery of total macrobenthic abundance after dredge fishing (Collie et al. 2000). Similarly, recovery of total benthic macrofaunal abundance after sand mining ranges from 3-30 months, although community composition occasionally fails to exhibit recovery even after multiple years (reviews in Brooks et al. 2004; 2006). Analyses of separate species (or genera) showed that, despite relatively rapid recovery of total abundance of benthos, sandy sediments commonly harbored one or two long-lived taxa that remained numerically depressed for longer periods of time after fishing ceased (Collie et al. 2000).

Analyses of impacts of bottom-disturbing fishing on population densities of individual species (or genera) also revealed the most negative effects in gravel-sized and muddy sand sediments, which represent typically stable habitats. The most stable benthic habitats, which are those created by biogenic structures, especially emergent clonal invertebrates on the OCS but also within reefs of tubiculous polychaetes, have been inadequately studied to characterize their recovery times after bottom disturbance by trawls and dredges (Collie et al. 2000). Nonetheless, the evidence in Sainsbury et al. (1997) that trawling damage to biogenic habitat provided by emergent sponges and corals requires greater than fifteen years to recover is not contradicted by any comparable studies revealing rapid recovery of such a biogenic habitat.

Auster (2008) and Auster and Langton (1998) provide models of how fish populations may be expected to suffer from loss of biogenic habitat complexity provided by emergent epibiota, such as sponges, erect brozoans, and gorgonians, after disturbance induced by bottom trawling and dredging. Effects of sand extraction in borrow areas would be expected to be similar, except to the degree that previous bottom-disturbing fishing practices in sandy sediments may have already cleared the bottom of these habitat-providing clonal invertebrates. Mean mortality of all benthic macroinvertebrates after trawl and dredge fishing is 55% (Collie et al. 2000) versus a mortality rate from sand extraction from borrow areas ranging from 40-95% (Hill et al. 2011). Collie et al. (2000) demonstrated that areas on the Georges Banks that are intensely trawled and dredged by fishing boats are characterized by greatly reduced abundance of “bush-like” benthic epibiota and typically associated mobile demersal megafauna like shrimps, polychaetes, brittle stars, mussels, and small fishes as compared with undisturbed areas. Several species of demersal fish, some of which exhibit lower densities on the disturbed fishing grounds, regularly consume members of the invertebrate megafauna. The conceptual models of Auster (2008) and Auster and Langton (1998) propose high importance of this biogenic complexity to settlement of fish that cue on particular species or groups of benthic organisms and to post-settlement survival of fish that depend on sheltering within emergent structural habitat elements. Consequently, these studies of fishing disturbance on benthic communities provide insight into how demersal fishes respond indirectly to bottom disturbance, with direct applicability to impacts of sand extraction on the benthic community, implying that demersal fishes may exhibit similar responses to sand mining, even if these effects have not yet been sufficiently explored at mining areas.

In summary, the now-extensive literature on biological impacts of intense trawling and dredging on the benthic communities and habitat services to demersal fish, shrimps, and crabs of the sedimentary seafloor not only provides insight into impacts likely held in common with the disturbance of sand mining, but in addition places these sand mining impacts in a broader context of potential cumulative human impacts on biological systems of the continental shelf. Specifically, the direct take of harvested fishes and untargeted losses from bycatch over wide areas of the continental shelf implies that fish densities and biomass are generally diminished below baseline abundances over the continental shelf of the Atlantic and Gulf coasts even before consideration of sand mining for beach nourishment and other purposes. The community composition of fishes has been transformed, with apex consumers most intensely diminished (Myers et al. 2007) and smaller species and individuals predominating (Pauly et al. 1998). Relative abundances of scavengers are probably higher across the shelf in response to provision of food subsidies from bycatch discards. Sand mining is, therefore, occurring under conditions of lower abundance and biomass of the larger, high-order predatory fishes preferred by fishermen, of a fish community dominated by smaller fishes generally, and of modified benthic invertebrate prey communities and functionally diminished epibenthic habitat across virtually the entire sedimentary portion of the continental shelf from Massachusetts to Texas. Bottom disturbance of the benthic prey and biogenic habitat-providers induced by sand mining thus adds to this already widespread modification of the coastal shelf ecosystem.

On the other hand, to the degree that the sand shoals and mud-capped sand deposit locations are already regularly subjected to bottom-disturbing fishing practices, much of the habitat damage and biological transformation of the fish communities and the marine coastal ecosystem has already occurred. The reasons why sand mining impacts may persist for longer than impacts from bottom-disturbing fishing practices relate in part to the differing geomorphological and sedimentological legacies of each. Bottom disturbance by fishing rearranges the sediments, causing no net extraction. In contrast, sand mining extracts sediments by intent, and benthos inadvertently. More importantly, excavating deeper depressions, which act as depositional basins in an engineering context, runs a high risk of multi-year transformations or in the worst-case scenario of anoxia, suppression of the benthic invertebrate community in the presence of finer sediments, with energy transfers to demersal fishes, shrimps, and crabs affected in unknown ways by development of a transformed, mud-loving community or, for the worst case, blocked by absence of benthos. It is possible that the benthic community occupying the siltier sediments passes more energy upwards to the higher trophic levels than the preexisting sand community, except where oxygen depletion inhibits settlement and survival of the benthos.

Some such sand shoals may be valued by fisheries managers as potential spawning habitat for high-demand targets of recreational and commercial fishermen (e.g., Luczkovich et al. 1999; Slacum et al. 2010). If fish biomass is now generally reduced by overfishing throughout the continental shelf, then overall demand for benthic invertebrate prey should be lower than in the historic pristine system; however, because of widespread and intense bottom-disturbing fishing practices, benthic invertebrate biomass is almost certainly lower too. Nevertheless, if the immediate large loss of benthos in dredged areas induces demersal predators to forage elsewhere, they will likely encounter lower benthic food supplies than would have been expected before the present-day intensity of bottom-disturbing fishing. This may represent a problematic cumulative impact between commercial fishing and sand mining. However, because bottom disturbance does lead rapidly to higher abundances of small opportunistic invertebrates, typical of succession Phase I, smaller demersal predators, such as penaeid shrimps displaced from borrow areas by sand mining may experience higher prey biomass densities on bottom "cultivated" by trawling and dredging disturbance. Consequently, mitigation to replace benthic invertebrate prey killed by sand mining may be unnecessary if smaller, rapidly reproducing invertebrates that shrimp prefer to eat are enhanced by bottom disturbance associated with bottom disturbance by fishing gear as well as by sand dredging. These processes are hypothetical and need to be confirmed by appropriate observations in a rigorous design.

Mitigating for the benthic invertebrate impacts of sand mining, specifically for mortality of benthic invertebrate prey favored by demersal fishes and crabs, as well as mortality of bivalve mollusks that scoters prey upon, is probably not necessary given that commercial trawling for shrimps and bottom fishes is so widespread on the OCS of the Gulf and southeast Atlantic coasts that borrow areas disturbed by sand mining are likely to already have been impacted by trawl fishing. Furthermore, based on the data from the U.K., the vast area disturbed by trawl and dredge fishing on the OCS probably greatly exceeds the area affected by sand mining, so the additional impacts of sand mining are relatively trivial. Foden et al. (2009) indicated that the area of the continental shelf disturbed by bottom-disturbing fishing gear in the U.K. is more than 100 times larger than the area disturbed by sand and aggregate mining. A similar comparison is needed for the U.S. Gulf and Atlantic coasts, but the disparity in areas impacted seems likely to be much greater. Benthic impacts of repeated passes of trawling gear and of dredges have important consequences for the benthos that are similar to those of sand mining in selectively removing habitat-providing emergent epibiota from sedimentary bottoms and inducing high and relatively non-selective rates of infaunal invertebrate mortality. In both the case of disturbance by bottom-disturbing fishing gear and the case of sand extraction by dredging, the disturbance is so great that secondary succession is induced with recovery occurring in stages beginning with small, surface-dwelling opportunists and ending with more deeply living, long-lived climax species (Newell et al. 1998; Hill et al. 2011). Because a given individual cannot be killed twice, the passage of a dredge while mining sand causes limited benthic invertebrate mortality where bottom-disturbing fishing has recently taken place. To assess quantitatively how these two types of bottom-disturbing activities interact, spatio-temporal maps of OCS fishing intensity by location need to be created, perhaps using data from the NMFS Vessel Monitoring System, and compared to locations of sand shoals and other viable sand resources. Delineating sand ridge and swale complexes from shoals would be a challenge in this project.

4.7.3 Employing models to provide insight into disturbance responses and interactions at ecosystem scales

The most obvious impact of sand and gravel extraction is the removal of the substrate and its infaunal and epifaunal biota (ICES 1992). Because fishes convert a significant amount of benthic food resources into biomass, the greatest potential effect to fish populations utilizing sand borrow areas may be the alteration of energy flow (production of species through trophic relationships in the food-web). Alteration of benthic-pelagic coupling can reduce fish biomass directly (reduced food availability) or indirectly (displacement of fishes to alternate feeding habitats where competition with existing fishes may be greater). To date, it is unclear if the temporary organic enrichment from damaged fauna in the outwash produced during sand extraction (and likely confined to the footprint of the dredge area) partially mitigates for the loss of food availability to demersal predators, and if the replacement of large, long-lived benthic species like clonal epibiota and infaunal bivalves with small crustaceans and other opportunistic invertebrates like polychaetes during the early recolonization phase represents provision of equivalent or improved food sources for fish species (Tillin et al. 2011). Studies following dredging at borrow areas have documented changes in the abundances of representative invertebrate groups and shifts in diversity and richness patterns, with recoveries of total abundance varying from months to a few years, but with some long-lived species populations not showing recovery after a decade or more (see Section 4.2.2.1). Because benthic invertebrate communities on the OCS naturally vary in space and time, fish and overwintering seabird populations must exhibit some flexible responses to fluctuations in food resources, a characteristic that provides some degree of resilience to disturbances associated with sand mining. However, key ecological questions regarding the energy transfer efficiency of the post-dredging benthic community to higher trophic levels compared to the original community remain unanswered. Although the effects of sand extraction on higher trophic levels are difficult to assess and are not well understood, efforts have been made to address this issue. Because BOEM recognizes the importance of trophic interactions in the OCS, these types of studies are currently being considered in their future planning of research funding.

Sutton and Boyd (2009) looked at the impact of aggregate extraction from the Dieppe extraction site in European waters on the feeding habits of common benthic and demersal fish species. The Dieppe extraction site (2 km^2; 10-15 m depth) is located 5.6 km off Dieppe, France, in the Eastern English Channel, where extraction reached ca. 0.4-0.8 million tons per year from 1980 to 1985, followed by a decline in extraction (ca. 0.1 million tons per year) by 1992 (Desprez and Duhamel 1993 cited in Sutton and Boyd 2009). Fish species sampled for abundance and collected within and near the Dieppe extraction site for stomach content analysis showed some differences in response to extraction activities. Dredging activity adversely impacted plaice and skate (reduced abundance), but not red mullet, with differences attributed to the diet plasticity of the latter. In contrast, other species (black sea bream and cod) were attracted to dredged areas as these became colonized by opportunistic benthic species. This assessment (Sutton and Boyd 2009) implied that aggregate extraction may have greater negative effects on species with more specialized diets and, less resilience to prey changes, while inducing compensatory effects on species with less specialized diets. However, changes in fish species abundance and composition and their overall positive or negative impact on the energy budget need to be weighed against the functional roles of these species in the ecosystem.

To better appreciate the competing uses for benthic habitat on the OCS, one might consider the ecological role of this habitat and its community within the context of the larger coastal ocean ecosystem. Such ecosystem functions vary depending on: depth–which determines the amount of light reaching the benthic surface; ocean temperatures–which determine the species that are physiologically suited to exploit the habitat; and substrate type–whether sedimentary or hard. In addition to providing habitat for spawning and refuge, perhaps the most important ecological role of the benthic community is its contribution to marine food-webs. A large amount of primary literature on sand and aggregate extraction focuses on direct impacts to the geology of the seafloor and resulting alterations of the benthic community, while indirect effects and their temporal and spatial scales are much less completely investigated. Models should be validated and designed to be appropriate for their intended use (Rykiel 1996; Loehle 1997). Model outputs are best treated as hypotheses that are subject to testing through observations of a change in ecosystem driver, whether this occurs through a management action or via a natural change. Ecosystem modeling, in combination with empirical impact assessments, has a potential to elucidate both direct effects and indirect impacts that result from sediment extraction.

The key objective of modeling is to construct a representation of the organisms and processes that occur within the ecosystem that one wishes to investigate, such that one can manipulate components and the relationships between components to explore possible scenarios of ecosystem responses to perturbations. The utility of such models is dependent upon the body of knowledge available for an ecosystem. The conceptual and empirical understanding of an ecosystem develops in parallel. There are four major types of ecosystem models: 1) the basic "box and arrow" diagram that describes relationships (e.g., trophic) between organisms (boxes) and the general flow (arrows) of energy or elements; 2) coupled models, such as bio-physical models, that represent both the physical environment and organism responses to ecosystem alterations; 3) network analysis models that use a detailed version of the "box and arrow" model to examine both direct and indirect interactions between organisms, as well as a suite of system-level metrics such as the number and strength of interactions, the flow of energy or elements throughout the system, and temporal development of the system; and 4) the holistic "end-to-end" models that consider the physical, biological, and social (e.g., economic, political, management) mechanisms and relationships within an ecosystem. Models can represent an ecosystem as a snap-shot in time (steady state) or through time (simulation).

Marine ecosystem models offer a tool to examine both direct and indirect impacts of sand extraction in coastal ocean ecosystems. Relatively few such models have attempted to examine the impacts of sediment extraction. Of those published, each has emphasized the lack of sufficient information to adequately represent the suite of interactions that result from a sediment extraction disturbance, thereby creating significant uncertainty in the model results (CEFAS 2007; Cooper et al. 2008; Austen et al. 2009; Daskalov et al. 2011). These studies are based upon a general knowledge of ecological interactions found within the primary literature with some incorporation of site- or species-specific empirical data. CEFAS (2007) developed a predictive, three-dimensional, cellular automata model that coupled the physical and ecological impacts of dredging the U.K. seafloor. The CEFAS (2007) physical model generally reproduced the seabed changes observed at typical case study sites, including steady-state topography, height of modeled bedforms, and the distance between and movement of bedforms. The CEFAS (2007) biological model was able to reproduce species responses to changes in sedimentary environments as observed from case studies, but lacked detailed information on species interactions within the benthos and between the benthos and demersal predators.

Cooper et al. (2008) compared the effects of dredging disturbance on ecosystem functions using indices that characterize the richness and evenness of functions, rather than of species (as done in traditional techniques). Austen et al. (2009) offer a cost-benefit framework for examining the impact of sediment extraction on the delivery of the ecosystem services; however, these authors admit to the lack of available data for their analysis, a high level of uncertainty in valuation methods, and the likelihood of under-estimating the value of ecosystem services.

Dasakalov et al. (2011) created a spatio-temporal, ecosystem-level food-web model (ECOSPACE, network analysis) of consequences of sediment dredging in the Eastern English Channel to assess the effects of changes in species abundance and distribution on energy flow through food-webs and to fisheries. This model used two dredge-effects scenarios: mortality on zoobenthos alone, and direct mortality on both zoobenthos and demersal fishes (e.g., cod, plaice, and whiting). Simulations were run assuming various (2.5-30%) percent removal of biomass under each scenario, with changes in the food-web assessed for 30 years. Species-specific diets and species interactions were calibrated against historical surveys, and spatial simulations for the entire licensed area were run in segments with and without dredging. This food-web model showed direct (negative) and compensatory (positive) food-web responses through feeding and competition, and appeared to be sensitive to food-web complexities. The 30% zoobenthos-only mortality simulation showed a gradual decline in biomass in dredged areas, but included a 9.5% and 7.8% increase in suspension and deposit feeders, respectively. Biomass increases in these latter groups were attributed to organic enrichment (as per Poiner and Kennedy 1984; Newell et al. 1999). The 30% zoobenthos-and-fish mortality simulation on fish showed variable results. Most demersal fishes decreased in biomass, yet flatfish showed a slight increase in biomass, attributed to a release from competition. Overall, for fishes: 1) direct mortality induced greater biomass decreases than did benthic prey reductions; 2) dredging mortality resulted in a general decrease in fisheries catch, especially in dredged areas, except for pelagic fishes, whose catch increased; and 3) at 30% mortality, total fisheries catch decreased by 3.5%. Dredging effects were greatest in the dredged footprint and these effects diminished over the wider study area (Daskalov et al. 2011). Subsequent modeling exercises could benefit from better site-specific empirical knowledge on quantitative changes of key parameters (mortality, feeding relationships and spatial and temporal distribution) that characterize the impact of the sand extraction on benthic and fish populations.

In another study, biological indices were calculated for a North Sea site with low and high intensity of aggregate extraction (gravel) operations between 1996 and 1999 (Cooper et al. 2011). Somatic production, a metric of the quantity of energy potentially available as food for the next trophic level, was higher at the reference site compared to both the low and high intensity dredged sites, five years after dredging ceased (Cooper et al. 2011). Other indices also suggested that a return to baseline conditions would take more than five years. In contrast to typical U.S. OCS sand borrow areas that are mined only once every 3-10 years, the North Sea extraction sites (100 m x 100 m) were dredged more frequently, between 1-14 hours each year during the four-year study, making results only somewhat comparable. Although the North Sea study did not quantify impacts on fishes, alteration of the benthic community structure and changes in benthic production may have had effects on resident fish populations. A related study reported in Cooper et al. 2011) also observed decreased levels of somatic production as natural physical disturbance increased (combined impact of tide and wave action on the seabed) and as the proportion of gravel decreased (change in particle size resulting from aggregate dredging).

Other efforts have been made using basic energetic conversions of organic matter (ICES 1992). Rough estimates of the annual impacts of excavation of sediment from a 175×10^6 m^2 area in the North Sea translated into a gross loss of macrobenthic production of 525×10^6 gC/year, which is equivalent to a drop of 294×10^6 gC/year wet weight of demersal fishes (under the assumption of 10% ecological efficiency between benthos and demersal fishes). If, by contrast, all primary production were lost from this area of the North Sea, the gross loss of demersal and pelagic fish production would be 22.7 and 79.9×10^6 gC/year, respectively. This particular energetic conversions exercise did not account for losses/gains of specific species, but rather looked at functional groups (benthos, demersal, and pelagic fish). Similarly, trophic food-web models estimate that as much as 30% of the total biomass of exploitable fishes in waters of the North Sea is derived from benthic food sources (Steele 1965; Newell et al. 1998 cited in Newell and Seiderer 2003). Therefore, indirect effects on fish stocks from dredging activities may reflect the reduction and modifications in the foods available from benthic resources. Van Dolah et al. (1994b) also documented changes in the benthic fauna associated with changes in the sediment composition at a borrow area off Folly Beach, South Carolina. This assessment noted a shift from an amphipod-dominated to a polychaete/mollusk-dominated assemblage, with recovery to pre-dredge conditions requiring at least twelve months.

LMEs are regions of ocean space that encompass coastal areas from river basins and estuaries out to the seaward boundary of continental shelves and the outer margins of coastal boundary currents. LMEs are characterized by distinct bathymetry, hydrology, productivity, and trophic interactions, and these regions produce approximately 95% of the world's annual fish catches (Sherman 2001). LME food-webs may offer insight into the large-scale, often indirect impacts of sediment extraction in the OCS. Our search of the LME and marine food-web literature revealed no investigations that included impacts of sand or aggregate extraction; however, LME food-webs that examine other anthropogenic disturbances exist (e.g., Pauly et al. 1998; Watling and Norse 1998; Jackson et al. 2001; Thrush and Dayton 2002). Using network analysis, Link (2002) found a high degree of connectivity in his analysis of a LME food web of the Northeast U.S. Shelf ecosystem (Cape Hatteras, NC to the Gulf of Maine) that used a 25-year database of dietary information for over 120 species of fishes and invertebrates. Connectivity or connectance–the proportion of all possible links between species that are realized (e.g., Pimm 1984), and linkage density–number of links per species or interaction richness (e.g., Dunne et al. 2002) indicate ecosystem complexity or degree of interdependency and, therefore, may imply ecosystem resilience to disturbances (e.g., Holling 1973; Levin and Lubchenco 2008) and thus stability (Dunne et al. 2004; Gravel et al. 2011). Link's (2002) findings of high connectivity and linkage density in the Northeast U.S. Shelf LME were attributed to the openness of marine ecosystems, the relatively high proportion of generalist feeders, as well as the long lifespans and the ontogenetic changes in size and diet across the life histories of many marine species. Link (2002) questioned whether marine ecosystems are truly more connected or whether his long-term database and high sampling intensity may have identified and incorporated more species interactions than other food-web data bases, yielding unrepresentatively high connectance and linkage values. Although Dunne et al. (2004) disagreed with Link's (2002) computation of connectance, claiming that it effectively double counted cannibalism and mutual predation, their examination of the trophic structure of food-webs also found marine ecosystems to have large trophic interaction richness, thus yielding higher degrees of connectance than food-webs of terrestrial and freshwater aquatic (lake/pond, stream and estuary) ecosystems. Yet unanswered is whether the enduring change in benthic community composition that is observed in borrow areas (see Section 4.2) limits the energy flow to some fish, sea turtle, or bird species or whether relevant species are generalists and thrive with an altered diet (see Section 5.3). Also unknown is the extent to which the small-scale alterations of borrow areas influence the broader ecosystem.

The relevance of this LME modeling to inferring impacts of sand dredging on the affected OCS ecosystems depends on resolution of conflicting viewpoints on the relationship between system connectance and resilience to perturbations. One consequence of high food-web connectance is that strong perturbations are transmitted more widely through an ecosystem (Link 2002; Dunne et al. 2004) such that the high number of links may dampen effects of disturbance (Vázquez et al. 2007), whereas the alternative consequence may be the transmission of disturbance effects cascading throughout the system and affecting many instead of only few species (Dunne et al. 2004). With either interpretation, one might reasonably conclude that with such apparently high connectance in OCS ecosystems, some of the impacts of sediment extraction are more likely to be indirect and to extend beyond the borrow areas. Such conceptual questions remain unanswered. The omnivorous nature and high dietary overlap of the species, as well as their diet switching (Sissenwine et al. 1982; Garrison and Link 2000; Overholtz et al. 2000; Link and Garrison 2002), imply that few species are tightly coupled to another individual species (Link 2002). This high degree of functional redundancy, which confers greater stability (Peterson et al. 1998), may mask anthropogenic impacts (Link 2002). Nevertheless, some declines in function may not be discernible for many years (Jackson et al. 2001; Myers et al. 2007). The most complex ecosystem models currently available, such as the Atlantis end-to-end model (e.g., Kaplan et al. 2012), may help resolve the conflicting interpretations on how connectance affects resilience, as this is applied to sand mining and fishing disturbance. As with most scientific fields, conceptual models precede the empirical study of ecosystems. Although outputs of such grand-scale models are necessarily uncertain, ecosystem modeling still has value to management of sand mining and other perturbations through using the outputs to generate testable hypotheses about how species and environmental interactions create unexpected, emergent impacts that would not have been anticipated without the model outputs. Modeling is best linked with empirical hypothesis testing, which can provide the rigor necessary for taking management actions.

Overall, there is lack of empirical information on the impacts on OCS sand mining by impacting mechanism, and as a whole, on the ecological interactions within these habitats. A greater understanding of the ecological role of these habitats to critical biological and prey resources (i.e., invertebrates and fishes), and the consequences of each relevant impacting mechanism on sand shoal resources will likely enhance our understanding of the overall and general impacts of sand mining on ecological interactions across natural resource communities.

5.0 SUMMARY OF FINDINGS, DATA GAPS, AND RECOMMENDATIONS FOR STUDIES TO ADDRESS THE MAJOR DATA GAPS

Section 4 included summaries of the findings of relevant studies, mentioned important data gaps, and briefly suggested studies to address these gaps within each resource category: benthic resources, fishes and essential fish habitat, foraging seabirds, marine mammals, and sea turtles. In this section, major and integrative data gaps are posed as questions or overarching concerns that we first discuss in the context of the current state of knowledge, which then leads to recommendations for feasible studies to address these major data gaps. We attempt to prioritize these studies by suggesting ordinal rankings within each resource category and overall rankings of their relative importance to sustaining a growing program of environmentally sound OCS sand mining (Table 5.1).

Priority rankings were based upon the following factors: 1) value of potential results to support changes to the present management of the BOEM marine minerals program in enhancing sustainability of the sand resource, physical habitat features, and biological resources; 2) opportunity to resolve current conflicting scientific theories or hypotheses by gathering empirical evidence; 3) the presumption that testing effectiveness of mitigation actions is important so as to avoid wasting resources or putting other resources at risk if some mitigation actions prove ineffective; 4) continuity with ongoing BOEM projects such that a logical follow-up study would provide additional new insights; 5) engaging new, potentially high-value approaches to the inter-disciplinary and multidisciplinary study of OCS seafloor processes and resource dynamics; and 6) certain scientific uncertainty to address resource agency information needs.

It is of interest to compare these recommendations with those by Michel et al. (2007), who conducted a critical technical review and evaluation of the techniques used for site-specific studies sponsored by the MMP. Research to address some of the prior recommendations has begun, such as the work to improve the understanding of the current patterns and morphologic response of dredged areas (e.g., Nairn et al. 2007), though such studies should continue. Many of the recommendations in Table 5.1 are multidisciplinary, which was one of the key recommendations for BOEM research in Michel et al. (2007). Studies to identify and test the effectiveness of mitigation measures continue to be of importance, with Table 5.1 identifying more specific measures that need well-designed studies to be able to achieve definitive results.

As BOEM proceeds with developing studies to support its mission, several of the Michel et al. (2007) recommendations provide useful guidance including: improve the scope of work of site-specific study projects; continue and enhance the peer-review process of both proposals and draft reports; require some standard protocols for data collection and analysis, or include justification for other methods; and require biophysical habitat mapping.

Table 5.1
Prioritizing research gaps and recommendations.

Research Recommendations to Address Data Gaps		Overall Rank
	Benthic Resources, Communities, and Habitats	
1	Resample previously studied borrow-area pits along with environmental factors to better observe and model recovery patterns beyond 2-3 years	3
2	Determine the population-level and fisheries importance of reproductive output of blue crabs on Ship Shoal	12
3	Determine the degree of site specificity of specific individual shoals for spawning by valuable fishes and invertebrates	1
	Trophic Interactions	
1	Determine whether demersal fish and crabs become food-limited during discrete (pulse) disturbance events on the OCS that significantly reduce benthic infaunal prey	5
2	Determine how topographic high ground on the seafloor may lead to enhanced water-column abundances and production of fish based upon fluid dynamics	6
3	Use new approaches to assess impacts of sand mining on fish use of dredged areas and unmodified areas by following acoustically tagged fish movements	7
4	Collect and analyze spatially explicit data on commercial and recreational fishing effort and relate effort to locations of sand resources	8
5	Test how feeding and energy transfer rates from benthos to fish, crabs, and shrimp vary between sandy and muddy bottoms within dredged areas	9
	Dredging Practices/Mitigation Measures	
1	Model and validate the sediment morphodynamics of sediment-extraction scenarios of shoals to optimize both sediment exploitation and the sustainability of OCS resources	2
2	Determine whether dredging in strips influences recovery of sediment composition and/or benthic communities in borrow areas	4
3	Determine the need for and effectiveness of relocation trawling in the OCS as a method to reduce entrainment of sea turtles; assess impacts of bycatch on other protected species	13
	Sound Impacts	
1	Collaborate with other agencies to determine the behavioral responses and physiological effects of dredging-associated sounds on marine mammals and sea turtles	10
2	Gain a better understanding of the underwater hearing of seabirds	14
	Seabirds	
1	Better understand the habitat value of OCS shoals for seabirds	11

5.1 What is Known about Dredging Impacts on Benthic Resources, Communities, and Habitats and How can Emerging Data Gaps be Filled?

Much is known already about the factors that influence the resilience of benthic resources, communities, and habitats to impacts of OCS sand mining, but further observational, experimental, and modeling studies are still needed to improve the process of protecting resources of value. Benthic biological recovery to pre-dredging conditions of abundance, biomass, species richness, and community composition is typically faster where natural sediment disturbance by waves is greater, where sand transport rates are higher, where sedimentology inside borrow areas is not greatly transformed by the dredging and subsequent in-filling processes, and where dredging does not excavate deep pits (Foden et al. 2009). Each of these variables is determined by choices in planning and permitting the dredging design. Dredging for sand resources resembles other major seafloor disturbances, such as massive sediment disposal (McCall 1977) and widespread, repeated bottom disturbance by fishing with trawls and dredges (Collie et al. 2000), in reducing abundances, biomass, and species diversity of the benthos, and thereby initiating a process of secondary succession. Benthic environments whose surface sediments undergo high natural rates of physical disturbance, like clean sand in relatively shallow shelf waters, are characterized by more opportunistic, short-lived benthic organisms, which can recover quickly after disturbance, either natural or dredging (Newell et al. 1998; Collie et al. 2000; Hill et al. 2011).

Where intrinsic value of benthic biological resources may be high enough to elevate concerns to the level of a potential major impact can be envisioned under two possible conditions as discussed below.

First, few benthic resources are listed under the ESA, which requires substantive avoidance of further risks. For the Atlantic and Gulf coast OCS, the only organisms that presently fit into this category are two species of stony corals, staghorn and elkhorn corals, which do not range further north than central Florida on the Atlantic east coast and do not occur in U.S. waters of the western Gulf coast. Both species require hard-bottom benthic habitat, so any risks to either of these species should be removed by permitting sand mining only at sufficient distances away from hard bottom to prevent siltation, sedimentation, and burial of existing staghorn or elkhorn colonies and hard-bottom habitat suitable for their colonization and occupation. Pipeline corridors can be routed to avoid these habitats, or mitigation can be used to offset unavoidable impacts.

Second, because benthic resources, communities, and habitats are generally much more valuable for the ecosystem services that they provide than for the intrinsic value of the macrobenthos itself, high risks to such ecosystem service functions could conceivably be elevated to major impacts if the service at risk is judged irreplaceable or perhaps even sufficiently valuable. Specifically, the benthos serves important roles in promoting production of many demersal predators of ecological and economic value as targets of fisheries, such as blue crabs, penaeid shrimps, and several demersal fishes (Foden et al. 2009). In addition, the benthos serves to facilitate global biogeochemical cycling of elements that sustains life and influences global climate (Foden et al. 2009). This second major ecosystem service of the benthos is unlikely to be modified substantially by sand mining, which affects such a small fraction of the global or even coastal seafloor area. This first major functional role of the benthos–facilitating production that also includes producing forage fishes and invertebrates that support feeding of seabird, sea turtle, marine mammal, and larger predatory fish populations–could be influenced by sand mining. In particular, sand shoals may represent habitats associated with reproductive activities of key species acting through one or more processes associated with sand shoal habitat. For example, sand shoals, as illustrated best by Ship Shoal, can provide elevated ground that remains in the surface mixed zone, and thereby remains bathed by oxygenated waters and providing perhaps critical benthic prey for blue crabs when bottom-water anoxia develops over deeper shelf habitats (Grippo et al. 2009, 2011) and forms a seasonal "dead zone" in which sessile benthos is killed (Rabalais et al. 2001a). Ship Shoal hosts foraging by large numbers of female blue crabs during the egg-development and release processes, serving to supply the larvae that then return to coastal bays and estuaries and sustain coastal and estuarine blue crab fisheries (Gelpi 2012; Gelpi et al. 2009). Other sand shoals have been demonstrated to host spawning aggregations of various fishes of value in commercial and recreational fisheries, such as speckled trout, gray trout, and red drum on ebb tidal shoals off Ocracoke Inlet, North Carolina (Luczkovich et al. 1999) and gray trout on deeper shoals off Delaware and Maryland (Slacum et al. 2010). This raises the questions of what percentage of the reproductive activities of affected fishery species is associated with sand shoals, which shoals, and at what water depths, and how flexible these species are in choosing alternative spawning sites if traditional spawning sites were degraded. On the basis of limited current observations, the value of sand shoals as spawning habitat in the OCS seems unlikely to suffice to raise sand mining risks to a level of even moderate concern. Yet data addressing these questions of how dredging may interfere with successful spawning aggregation and reproduction are needed to make sufficiently informed decisions about the need to manage sand mining to protect propagation of commercially important fishery species.

The design of the sand dredging pattern has been discussed by scientists and managers concerned with minimizing or mitigating the impacts of sand extraction on benthic resources and habitat, hypothesizing that dredging in strips or otherwise leaving undredged areas may dramatically speed up recovery (CSA et al. 2010; Dibajnia and Nairn 2011). Benthic ecologists presume that the magnitude of impacts on benthic resources, communities, and habitats increases with surface area disturbed by dredging. Consequently, excavating more deeply over less total area would reduce immediate direct effects of sand and benthos extraction. However, the subsequent recovery process may be slower in the more deeply dredged depressions into the seafloor. Thus, any benefits of minimizing the area of the dredging footprint must be weighed against costs associated with potentially protracted recovery. Because follow-up surveys after prior sand mining have revealed several examples of relatively deep pits that have exhibited slow in-filling rates–sometimes implying a recovery process lasting, in the extreme cases, a decade or more, sand extraction designs now tend to avoid excavation of deep pits as a means of mitigating against long recovery times. We recognize further that even relatively shallow pits, such as have been dredged on South Carolina State waters, can exhibit long times of infilling with muddy sediments and slow benthic biological recovery times in some locations while not in others (Bergquist et al. 2011a). This raises the possibility that further data on availability of nearby sand resources, hydrodynamic bedload transport rates of sand sediments (as included in the recovery model of Boyd et al. 2005), and delivery rate of suspended fine sediments and syntheses of available information on recovery rate of previously dredged pits may allow accurate identification of conditions and locations where pits could be expected to fill quickly and original macrobenthic communities to reestablish rapidly. Assembly of data (including new data from follow-up surveys of past borrow areas) and syntheses of patterns of benthic recovery in a context of sediment transport environments may provide an important means of mitigating against potentially long recovery times for even modest pits excavated during sand extraction. Such a study could also assess the trade-offs between excavating larger shallower depressions versus smaller deeper depressions.

Below, we provide a brief rationale for the need for each of several studies designed to address important data gaps, with each study given a priority ranking within groupings by topic areas and overall across all topic areas.

Benthic Resources, Community, and Habitat Data Gap #1 *Overall rank: 1*

Determine the degree of site specificity of specific individual shoals for spawning by valuable fishes and invertebrates

Some studies have revealed that certain sand shoals are sites of spawning for commercially and recreationally valuable species of fishes and crabs. It is presently unclear which types of sand features are predominately used for this function, although they vary from ebb tidal delta shoals to the huge Ship Shoal in Louisiana. For example, sand ridges off Delaware and Maryland are used by gray trout (weakfish) for spawning (Slacum et al. 2010) and Luczkovich et al. (1999) confirmed spawning by red drum, speckled trout, and weakfish on ebb tidal shoals off Ocracoke Inlet, North Carolina. These studies, when combined with the Dubois et al. (2009) and Gelpi (2012) demonstrations of the value of Ship Shoal off the Mississippi Delta in Louisiana to blue crab reproduction, raise the question of what fraction of the total regional spawning or reproduction by each of these species occurs on sand shoals. Sand shoals appear to be foci of spawning aggregations for many coastal fishes (Kaiser et al. 2004), although for subsequent feeding in nursery habitats, many demersal fishes use broad expanses of inner continental shelf to the same degree that they use estuarine bottom (Woodland et al. 2011). Critical information is needed to assess how fixed or flexible each species is in spawning site selection. Many tropical reef fishes (Johannes 1981; Sala et al. 2001) and temperate demersal fishes such as red drum that utilize sedimentary habitats (Pearson 1929) use specific sites for mass, often multi-specific, spawning year after year, such that degrading the spawning site or modifying it in any way that prevented the fishes from using it as their spawning location could have lasting impacts on population size and production. Further studies, including use of acoustic telemetry, to determine whether various sand shoal types and whether certain shoals of concern are used exclusively or predominantly by any of these commercially valuable species for their regional spawning are needed to be able to characterize the level of concern associated with sand mining on sand shoals serving as spawning sites. In 2014, BOEM is hosting a workshop entitled “Workshop and Research Planning to Improve Understanding of the Habitat Value and Function of Shoal/Ridge/Trough Complexes to Fish and Fisheries on the Outer Continental Shelf” with the goal to bring together a select group with a broad knowledge base to characterize the present scientific understanding of the fish-related functions of shoal habitats. This workshop and the workshop final white paper could be essential to address critical gaps in understanding the habitat uniqueness, functions, and values of ridge/swale systems and shoals (individually and within a region) related to fishes and to identify studies to fill those gaps.

Benthic Resource, Community, and Habitat Data Gap #2 *Overall rank: 3*

Resample previously studied borrow-area pits along with environmental factors to better observe and model recovery patterns beyond 2-3 years

Although we have started the process of reviewing the history of the process of recovery of sediments and of the benthic resources and communities within relatively deep dredge pits (Section 4.2.2.3), more data collection on recovery of previously sampled dredge pits, sediment transport modeling, and interdisciplinary synthesis is needed to design dredging to quantify losses of benthic resources and habitat ecosystem services associated with excavating deeper pits or even relatively shallow pits, where siltation dominates infilling. Much variability exists among deeper dredge pits in the time required for infilling and for redeveloping their benthic invertebrate communities (Jutte and Van Dolah 2000). Provided that processes influencing recovery of dredge pits in shallower state waters and closer to shore than OCS borrow areas are reasonable analogs for deeper-water processes, the large set of previous sampling studies of multiple dredge pits and their recovery off South Carolina (Bergquist and Crowe 2009; Table 4.1) makes that area a good candidate for resampling and for developing methods of assessing physical transport of sandy sediments versus siltation and their relationships to driving factors. The Sandy Point borrow area used for the Pelican Island project is an area of active research on causes of variation in rates of infilling (see Figure 2.12). Some pits may remain unfilled for as long as a decade or more, and any pit that does not fill in relatively rapidly through transport of sandy sediments by local current flows becomes filled only gradually with fine sediments dropping out of suspension in the water column, thereby failing to replicate preexisting sedimentology (e.g., Scott and Burton 2005). Benthic colonization of such muddy sediments is unlikely to produce the same community composition as maintained pre-dredging on sandy sediments (e.g., CEC 2003). We presume, as concluded by Foden et al. (2009) based on synthesis of many studies, that once the local topography has recovered and ongoing horizontal transport of coarser sediments has restored surface sediment sizes, a macrobenthic community will redevelop that resembles the initial sand-loving benthos present before dredging. However, this presumption requires confirmation by follow-up sampling of a suite of previously documented slowly recovering dredge pits. Furthermore, resampling is required to document the length of time required for infilling of many of the deeper pits where previous monitoring failed to demonstrate convergence back to preexisting bathymetry, sedimentology, water quality, and macrobenthic community composition. Even more important is conducting physical measurements on areas where pits filled in quickly and those where infilling has been slow that are sufficient to generate reliable sediment transport models that can explain and predict recovery time of pits excavated during sand mining. One goal of such modeling would be developing indicators that could be confidently and accurately applied to recovery of sand sheets, relatively flat sand bodies, and incised paleochannel areas to determine causes of differing infilling times of dredge pits of varying depths so that areas with more rapid recovery rates can be selected to mitigate against protracted recovery and longer durations of lost ecosystem services.

Benthic Resource, Community, and Habitat Data Gap #3 O*verall rank: 12*

Determine the population-level and fisheries importance of reproductive output of blue crabs on Ship Shoal

A new, follow-up study focus for continued research on Ship Shoal, Tiger Shoal, and Trinity Shoal is needed to determine to what degree the blue crab stock along the entire north-central Gulf of Mexico coast depends on the reproductive output of the female blue crabs on Ship Shoal (see Gelpi et al. 2009; Gelpi 2012). In addition, tests are needed of whether sand extraction on some portions of Ship Shoal would result in reduced reproductive output of blue crabs or if the same numbers of blue crabs would succeed in foraging, egg production, and surviving even if they were concentrated into a smaller fraction of undisturbed Ship Shoal. This assessment should develop a sufficient understanding of the process of concentrating blue crabs on smaller shoal areas so that blue crab impacts can be reliably modeled as a function of the percentage of Ship Shoal dredged for sand to address where cumulative impacts of sequential sand mining may become important.

5.2 How Does Dredging Affect Trophic Interactions?

Although much is known about how sediment disturbance and extraction associated with sand mining affects the benthic macroinvertebrate resources, communities, and habitats on the OCS seafloor, the knowledge falls short of what is needed to manage and sustain the most important ecosystem service of soft-bottom (sedimentary habitat) benthos–the delivery of food and thus trophic transfers from benthic resources to demersal consumers of benthos. In other words, more data must be gathered on how sand mining on the OCS influences the ecological interactions of energy transfer from benthic macroinvertebrates (and meiofauna) to the predators that consume them, especially to those of commercial value and of value as prey for higher trophic-level resources, such as sea turtles, marine mammals, larger fishes, and some diving seabirds.

Many past studies of multiple types of disturbance to seafloor habitats from sediment disposal, organic enrichment, bottom-disturbing fishing gears, and from dredging and extracting sand and gravel resources provide well-accepted models of the impacts on the benthic macroinvertebrate community. Disturbance, and especially extraction of sand and gravel, immediately and dramatically reduces abundance, biomass, and diversity (species counts and information-theoretic diversity) of macrobenthos. Following this disturbance, the macrobenthic community undergoes the process of succession during recovery from disturbance, leading back towards the undisturbed community state–provided that dredging and extraction have not fundamentally altered the sedimentology on the seafloor at extraction sites or depositional sites nearby, or altered the physical-chemical environment in important ways. These studies focus on the macrobenthos and only occasionally include meiofaunal-sized benthic invertebrates (Brooks et al. 2006). If dredging modifies the granulometry, then the benthic community composition in the newly developing community will differ from the original community prior to dredging. Biological diversity may also change, increasing if sediment heterogeneity increases (MESL 2007). The succession models that have been derived from the many studies of macrofaunal responses to sand and gravel extraction and other bottom disturbances are community-level models. As such, they deal only with the benthic invertebrates and only tangentially and indirectly with higher trophic levels.

The intense focus on benthic invertebrate communities in assessing impacts of sand and mineral extraction from the seabed is based upon a long history of using the benthic community to monitor environmental impacts. Warwick (1993) presents the scientific justification for widespread monitoring of benthic communities as a method of inferring environmental impacts and recovery. The benthos is relatively stationary, so spatial patterns of disturbance can be readily associated with the locations and scale of the disturbance in question. The taxonomy of benthic invertebrates is well described so identifications can be confidently made at a species level. Evidence is strong, nonetheless, that robust inferences about the status of the benthic community can still be made without much loss of insight by making identifications only to the family level and thereby sparing high costs of taxonomic resolution (Somerfield and Clarke 1995; Warwick 1988). Because of the largely stationary nature of the benthos, treatment and control sites can be monitored for years afterwards to infer recovery rates and characterize recovery dynamics without confounding mixing of benthos between treatments and controls. Powerful multivariate analytic methods and supporting software have been developed (e.g., PRIMER and PERMANOVA), allowing highly resolved distinctions among different community states. Finally, the benthos is of recognized importance to higher trophic levels so when the benthic community recovers to its undisturbed state, it presumably delivers levels of food-web services to demersal consumers equivalent to what the undisturbed benthos had provided.

The problems arising from the present heavy reliance on macrobenthic community composition as a metric of broader habitat and even ecosystem status arises when comparing communities that differ in composition. The vast literature on disturbance, including especially extraction of sand and benthos, reveals repeatable and thus largely predictable patterns in the post-disturbance behavior of community parameters (e.g., McCall 1977; Rhoads et al. 1978; Newell et al. 1998; Hill et al. 2011). Abundance, biomass, and species richness of the macrobenthic community drops dramatically as sediments and associated infaunal invertebrates are extracted. Recovery from that disturbance progresses through a succession of stages of varying community composition, characterized by varying life-history and demographic characteristics of the dominant benthic invertebrates in each successional stage (MESL 2007; Fig. 4.1). Among the four major community metrics, abundance recovers most rapidly as small-bodied, surface-dwelling, opportunistic species settle in the first stage of recovery (Bolam and Rees 2003; Hill et al. 2011). As these species continue to settle and the earlier settlers grow in size, biomass then exhibits a dramatic increase. Both abundance and biomass typically increase to levels substantially greater than those that characterized the benthic community prior to the disturbance of extraction of sand and associated benthos (Pearson and Rosenberg 1978; Newell et al. 1998; Hill et al. 2011). Each of these two community metrics then exhibits a subsequent crash, as the opportunistic species exhaust their food supplies through competition and are consumed by predators. Further settlement includes more long-lived species in a second stage of succession – species that burrow more deeply into sediments that have begun to recover their pre-disturbance oxygenation through bioturbation by the early colonists. Species diversity increases from the first to this second successional stage, despite the large declines in total benthic abundance and biomass. Community composition begins to approach that of the pre-disturbance benthos, provided no major modification of sedimentology. Finally as the climax community state is approached, early opportunists disappear and more long-lived, deep-burrowing species are established as the final Phase III climax community emerges. Most benthic ecologists predict a slight decline in species richness as this climax community is approached, following the intermediate disturbance hypothesis in which in the least disturbed environments competition for resources leads to exclusion of some competitively inferior species

(Connell 1978), but this response is not a universally demonstrated feature of benthic succession (Naqvi and Pullen 1982; Bergquist and Crowe 2009). Community composition is expected to converge with that of control areas that were not disturbed by the sediment extraction. As described, this three-stage process of benthic community recovery (Figure 4.1) assumes that the pre-extraction community existed in an environment with little natural disturbance so a climax community was maintained. This assumption is not always true because some sand resources targeted for mining may be found in more physically disturbed locations where a Phase II community state is naturally maintained and thus recovery needs only proceed to that stage to be complete.

Interaction Data Gap #1 *Overall rank: 5*

Determine whether demersal fish and crabs become food-limited during discrete (pulse) disturbance events on the OCS that significantly reduce benthic infaunal prey

Although the benthic infauna of sediments on the OCS seafloor is known to provide food for demersal fishes, blue crabs, and penaeid shrimps, many of which also have value to commercial and/or recreational fisheries, what is unknown is whether these predators are food-limited during their use of OCS sand-bottom habitats containing suitable sand resources for mining and during the period(s) of residence more generally on the OCS. Further data are needed to answer the question of whether foraging demersal fishes, blue crabs, and shrimps of several species could be displaced by mining from seafloor areas of historic use for foraging, yet suffer no reduction in food intake or overall fitness because, when displaced to other locations nearby, they find prey to be equally available. This equates to a need to evaluate whether populations of these consumers exhibit density-dependent individual growth and population production during the life stage(s) when residing and feeding on the OCS in sand bottom habitats targeted for sand removal.

Fisheries science has a long tradition of using population models combined with empirical data on size-at-age as a function of stock density to determine whether individual growth and population production exhibit any relationship to stock size (biomass). Such relationships can include positive or negative relationships with stock size (population density) or can show general density independence. Data analyses can reveal thresholds above which or below which (Alee effects) responses are expressed. Density dependence can vary in importance from year to year, based largely on changes in available food resources. Such inter-annual variation among demersal predators of relevance to sand mining could provide important insights into whether benthic infauna as prey limit individual growth, production, and survival of these predatory species. Using available, or collecting and using necessary new, information to conduct rigorous, compelling tests for density dependence among the species of concern because of known foraging use of habitats containing sand resources could help fill the gap in our understanding of the sensitivity of these predator populations to potential displacement from extraction sites during and perhaps for some time after sand mining is conducted.

Interaction Data Gap #2 *Overall rank: 6*

Determine how topographic high ground on the seafloor may lead to enhanced water-column abundances and production of fish based upon fluid dynamics

Emergent structural features on the OCS seafloor such as ridge and swale complexes or other topographically elevated sand shoals have the potential to serve as ecological foci of enhanced trophic interactions and thus energy flows passing up pelagic ocean food chains to even the apex predators of the coastal ocean. We propose a mechanistic hypothesis to explain how the topographic high grounds on the seafloor may lead to enhanced water-column concentrations of suspended plankton and other particulates based upon fluid dynamics. Topographic high grounds such as sand ridges between swales interact with wave-driven, tidal, or wind-driven currents by inducing microscale to mesoscale (<1 km to <10 km) eddies (van Rijn 1993) that can enhance feeding capacity of planktivorous forage fishes, such as anchovies, silversides, menhaden, and thread herring, at the base of the consumer food chains of the pelagic ocean. Feeding enhancement is hypothesized to develop from these eddies creating local concentrations of phytoplankton and zooplankton that make planktivory more efficient. Eddy flows behind emergent sand features would also be expected to modify biogeochemical processes, such as breaking down near-bottom stratification, bringing oxygen to the seafloor, and allowing nutrients to be mixed upwards. Local concentrations of these schooling forage fishes in the lee of emergent sand features similarly attract schools of piscivorous predators such as Spanish and king mackerel. Furthermore, piscivorous seabirds such as northern gannets, cormorants, loons, and several terns and gulls also respond to the high concentrations of forage fish, leading to enhanced energy flow to seabirds. Even baleen whales such as right whales and humpbacks that make coastal migrations in nearshore travel corridors may take advantage of feeding on concentrations of both zooplankton and small baitfishes in the lee of larger emergent sand bodies.

This scenario, proposed to explain why pelagic foraging hot spots can develop over some seafloor sand shoals, is plausible but not yet fully supported by necessary field observations and associated fluid dynamics modeling. According to basic fluid dynamics theory (van Rijn 1993) and Nairn (pers. comm., 2012) some form of eddy or circulation pattern is possible and likely, particularly evident on the steeper side of the shoals during stronger flow conditions. The first step could be to develop a 3D model with fine enough resolution to evaluate eddy development, then use existing data on current profiles measured at selected offshore sand shoals (e.g., Dibajnia and Nairn 2011) to validate and refine the model. Existing current profile flow data collected offshore of Maryland at the NW and SE leading edge of the Isle of Wight Shoal may not be recorded at fine enough scales, but an examination is still appropriate. The results, when combined with field observations on locations and feeding activities by planktivorous fishes, could indicate if eddy formation is likely a process by which these hot spots of pelagic trophic transfers develop. If eddy formation is determined to be an important process, modeling can then be applied to answer the question of whether extracting sand resources from such shoal borrow areas would be expected to suppress the development of the eddies that concentrate plankton as the first step in promoting food-web energy transfers. The model predictions themselves should validated using observations under varying flow regimes before and after sand extraction. Such an interaction study could lead to the mitigation of negative effects on food chain transfers at such ecological hotspots by limiting dredging cut-depths or volumes and or conducting spatial patterns of extraction so as to retain the physical topography required to induce effective eddy development.

Interaction Data Gap #3 *Overall rank: 7*

Use new approaches to assess impacts of sand mining on fish use of dredged areas and unmodified areas by following acoustically tagged fish movements

The current data sets estimating the spatial scale of foraging by demersal fishes, blue crabs, and penaeid shrimps on the OCS are insufficient to estimate the distances and areas over which these commercially important and valuable fishery stocks move during foraging. Such information is critical to estimation of how seriously localized and temporally changing degradation of benthic infaunal resources might be expected to impact feeding by key predators. Previously conducted studies that included efforts to test how demersal fishes may use areas mined for sand as foraging grounds failed to use technology that might provide an answer to that question without ambiguity. For example, the USACE study of impacts of the Asbury Park to Manasquan Inlet beach nourishment project (Burlas et al. 2001) employed otter trawling to sample and compare fishes at extraction sites versus control sites. Unfortunately, this sampling merely revealed that the spatial scale of mobility of the most important and abundant demersal fishes was extensive enough to intermingle fishes among the extraction and control areas. Demersal predatory fishes captured at the mining areas tended to have gut contents dominated by anemone siphons, all of which were absent in benthic samples of the borrow areas, presumably because of anemone injury and subsequent mortality during extraction along with uptake of sand during dredging. Hence, the trawling results failed to assess whether feeding was suppressed in mining sites because fishes trawled up over mined areas were so mobile that even their most recent feeding, as evidenced by gut contents, ranged beyond the extraction area.

Advances in acoustic tracking systems now make possible the placement of individual transmitters on demersal fishes so that their patterns of movement along the bottom can be tracked by a network of receivers. While that would not necessarily confirm feeding, one could use data from replicate transmitting fish to determine the time spent in recently dredged areas versus the time spent in undisturbed bottom nearby, thereby creating a measure of relative use of these two bottom types. This procedure could also be done before and after dredging so as to document use and degrees of site fidelity of key demersal fishes before and after sand extractions. Traditional trawling data are simply inadequate for this purpose because of the scale of movement and intermingling of fishes among dredged and control areas and the snap-shot in time that trawl sampling produces.

Interaction Data Gap #4 *Overall rank: 8*

Collect and analyze spatially explicit data on commercial and recreational fishing effort and relate effort to locations of sand resources

At present, the degree to which commercial and recreational fishermen target specific localities for their fishing that contain extractable sand resources, as a function of type of sand habitat feature fished by bottom-disturbing gear versus other methods, has not been adequately assessed. Tomlinson et al. (2007) include ethnographic data on fishing and fishermen's concerns with sand mining for two Florida localities, one east coast and one west coast. While useful in developing an understanding of conflicts, this study does not give fishing effort, broken down by bottom-disturbing gear versus other techniques and organized by sand resource habitat type, and the data set is restricted geographically. For some other sand mining projects, BOEM has done surveys or interviews with fishermen before dredging to be able to recognize and minimize user conflicts, but this too involved ethnographic data rather than spatially explicit, quantitative fishing effort measures by bottom type and sand feature, separately by gear types. Site selection for fishing, especially commercial fishing, tends to reflect abundant use of those areas preferred by targeted fish species, therefore providing indirect evidence of relative utilization among a variety of habitat types. Many fisheries, such as shrimp trawling, have had a history of federal fisheries observers, from whose log records and especially Vessel Monitoring Systems data records collected by NMFS (which are confidential, thus this analysis would have be done internally by BOEM), when combined habitat maps of sand shoals versus other types of habitats, researchers could assemble data on whether fishermen target sand bottom habitats of various types and with what gear and effort. This pattern, depending on how intense, would indicate a conflict between existing and future human uses, as well as suggest the intensity of disturbance from bottom disturbing gear use on various habitats by fishermen seeking commercially harvested species of value. The quantification of spatial overlap between different types of fishing for various exploited fishery organisms and sand resources on the OCS within each geographic area would aid in assessment of cumulative impacts and offer insight into the degree to which commercial fishing with bottom-disturbing fishing gear such as dredges and trawls may have modified the benthic resources, communities, and habitats before sand mining occurs.

Interaction Data Gap #5 *Overall rank: 9*

Test how feeding and energy transfer rates from benthos to fish, crabs, and shrimp vary between sandy and muddy bottoms within dredged areas

Although fully regaining the pre-existing community composition, benthic abundance, biomass, and species richness of macrobenthos may imply a conclusion that the benthic service of feeding shrimps, crabs, and demersal fishes has been restored, data are lacking from which to infer how this service varies among recovering benthic communities of differing composition at different phases of recovery. This same question of how demersal predators are served by different benthic infaunal communities also arises in the context of how communities on sandy sediments differ from those developing or developed on muddy sediments. These data are needed to quantify and evaluate food web support differences between sandy bottoms and muddier bottoms, specifically where dredging leaves behind a legacy of altered surface sediment character, such as mud in deeper dredge pits, which may only slowly return to surface sand cover after pits have first been filled by fine sediments. There are rational reasons to hypothesize that the benthic infauna of some earlier successional stages may even provide more food to demersal predators than the climax successional Phase III community, so new studies may prove insightful. For example, commercially valuable penaeid shrimp feed upon small, surface-dwelling benthic invertebrates, including meiobenthos. Because the benthos of Phase I successional communities is dominated by invertebrates small in size, surface-dwelling, and highly abundant, the value of such Phase I communities as prey for penaeid shrimps may be higher than the value of a climax community of benthic infauna. Furthermore, the crash in abundance and biomass of the surface-dwelling, small opportunistic benthic invertebrates that marks the transition between recovery Phase I and Phase II can be explained by either overpopulation of benthos depleting their food supply or by predation. These two alternative explanations have very different implications for the importance of the benthos in feeding demersal marine predators. Similarly, benthic infauna associated with muddier sediments instead of clean sand may differ in food value among the predator groups of demersal fishes, blue crabs, and penaeid shrimps. Penaid shrimps are more likely to prey on smaller benthos, including meiofauna. The paucity of studies of dredging impacts that even sample the meiofauna represents a major data gap inhibiting determination of whether early-succession benthic communities developing on muddy bottom or on natural sand bottoms provide valuable prey sources in the form of meiofauna for shrimps and other smaller consumers, such as juvenile fish and crabs.

This gap could be filled by conducting caging experiments in which various appropriate demersal predators would be enclosed with benthic infaunal communities that differ among sites in successional phase and in sediment type (sand versus muddy). In such experiments, diver-conducted hand coring would sample benthic meio- and macro-infauna before and at several times after introduction of each type of demersal predator (at least one species of demersal penaeid shrimp, the blue crab, and one or more treatments of demersal fish, such as gray trout and croaker, of various sizes). Cages would be large enough to hold replicate individuals of the assigned predator. Data could be taken on how composition of the benthic infaunal community changes inside inclusion cages and control cages lacking predators, and on how metrics of physiological condition of each predator change over the course of the experiment. Predators could be sacrificed periodically so as to assess gut contents and how predator physiology changes over the course of time in the experiment. Only by making such measurements in an experimental context can necessary rigorous data be provided to fill the gap in our understanding of food value of different benthic infaunal communities. Predator inclusion experiments have a productive history in benthic ecology (e.g., see Eby et al. 2005). Although caging inhibits large-scale mobility of the predators, any such caging effect is held in common among treatments of differing benthic communities. If these experiments can be sustained for more than about a month, growth rates of the predators can be measured as a response variable along with assessments of type and biomass of gut contents.

5.3 WHAT ARE THE BEST CONCEPTUAL DREDGING PRACTICES TO SPEED RECOVERY OF BENTHOS AND MAINTAIN THE PHYSICAL INTEGRITY OF OCS SAND RIDGE AND SHOAL COMPLEXES?

Dredging Design Data Gap #1 *Overall rank: 2*

Model and validate the sediment morphodynamics of sediment-extraction scenarios of shoals to optimize both sediment exploitation and the sustainability of OCS resources

As discussed in the different resources in Section 4, little is known about the ecological importance of OCS sand ridge and shoal complexes (individually and regionally) as fish habitat and as foraging areas for overwintering and migrating seabirds, sea turtles, and marine mammals. Until more is known about the relationship of geomorphology to marine resources, to habitat functions, and to functional ecological roles of sand ridges and shoals, the approach has been to identify and implement dredging guidelines to maintain the geomorphic integrity of shoals. Most of the sand shoals in the OCS are relict in that they are not connected to the littoral system; thus, sand removed during dredging will not be replaced and individual dredged shoals will eventually become smaller and their maximum height will eventually decline with repeated removals. OCS sand ridges (e.g., Fenwick Shoal) and shoal bodies (Sandbridge Shoal) in the Mid-Atlantic region are of particular concern because they have been identified as long-term sources of sand for critical shoreline protection sites. BOEM funded two modeling studies to predict how Mid-Atlantic OCS sand ridges respond to different dredging scenarios, with somewhat different recommendations based on the use of different models and inputs. Both CSA et al. (CSA 2010) and Dibajnia and Nairn (2011) recommended dredging from depositional areas such as the leading edge of the migrating shoal, so that the crest will reform by the continued transport of sand. They also cautioned against dredging of the trailing or erosional areas. The Holocene ravinement is often expressed as coarser lag deposits of sediment on the surface in this environment over back-barrier sediments or inlet-fill sequences, leaving sediment that is too coarse for use on beaches. Both groups also agreed that there is little risk that a shoal will "deflate" or "unravel" after repeated sand removal. However, there are differences in recommendations as to whether or not to dredge the shoal crest, where to dredge along the crest, and how much material can be removed from the crest without reducing the shoal elevation. Based on analysis of 180 shoals in the region, Dibajnia and Nairn (2011) developed a metric based on the Relative Shoal Height as a predictor of shoals that would be more likely to rebuild themselves to the same height after being dredged. They also said that shoals in water greater than 30 m would not likely rebuild to their original height.

These modeling studies have provided valuable guidelines for dredging sand ridges in the Mid-Atlantic OCS to maintain their physical integrity, and they are being applied to current dredging projects in the region. However, post-dredging bathymetric surveys will be needed to validate these guidelines and refine future recommendations. Also, studies similar to those conducted by Dibajnia and Nairn (2011) value in other regions, the next of which is already funded and being conducted by Applied Research and Coastal Engineering off the South Carolina coast. Dibajnia and Nairn (2011) provided guidelines for more such follow-up studies based on their work in the Mid-Atlantic region.

Dredging Design Data Gap #2 *Overall rank: 4*

Determine whether dredging in strips influences recovery of sediment composition and/or benthic communities in borrow areas

Many coastal experts (e.g., Whitlatch et al. 1998; Diaz et al. 2004; CSA et al. 2010) have suggested that dredging in strips would speed up both filling in of the dredged depressions by sediment slumping and macrobenthic recovery by lateral movement of nearby benthic organisms from undredged areas into the depressions (see data on slumping in Cooper et al. 2007). Project sponsors have been very reluctant to agree to dredge in strips because of the significant added costs in terms of dredging time and fuel consumption. Leaving unmodified strips of about 50 m would be feasible, but unless the sand feature is very large with a large volume of sand resource, such a pattern with 50 m strips is wasteful of resources. Dredged strips are already achieved in single pass TSHD dredging of large footprint sand areas that take low sediment volumes, so further study could perhaps be designed around such projects. Other resource managers are concerned that increased dredging time could lead to increased sea turtle takes.

Unfortunately, the suggested protocol of dredging in strips has yet to be subjected to sufficient rigorous field testing, so more empirical data are needed to assess whether this dredge design can mitigate for some benthic impacts by accelerating recovery. On short time scales, slumping and slope equilibration of the sides of dredged depressions will kill additional benthic invertebrates by burial under the slumping sediments, by augmented turbidity, and by abrasion, which further extends the macrobenthic mortality. For example, Barry et al. (2010) used empirical data from Kenny and Rees (1994, 1996) to model and quantify loss of macrobenthic individuals in the case of a U.K. dredging project done in strips; although only 70% of the borrow area was mined, 90% of individuals were lost because of these processes of slumping, burial, abrasion, and turbidity generation that affected benthos beyond the mined strips. Nevertheless, the influx of other benthic invertebrates that survive inside the dredged depression may speed up reproduction and biotic recovery. In a review of dredging impacts and recovery from sand mining in the U.S., Brooks et al. (2004, 2006) demonstrated that some studies show apparently faster recovery of sediments and macrobenthos when dredging is done in a design that leaves islands of undredged bottom to serve through slumping as sources of both sediments and benthos to jump-start recovery. Nonetheless, a more comprehensive observational study and meta-analyses of the impacts of leaving undredged strips are still needed so that the balance between greater initial impacts and possibly faster recovery can be quantitatively assessed with rigor, thereby replacing current opinions and supporting a better analysis of the tradeoffs.

5.4 Do Sounds Generated During Dredging Operations Affect Protected Species?

There are very limited field data on the sound levels generated during dredging operations in the OCS. The Wallops Island field study results will be the first comprehensive measurements of U.S. operations on the OCS (Reine et al. In prep). These results should provide the input needed for propagation and transmission-loss models of the received levels for different species of concern (cetaceans, manatees, and sea turtles). These results should also allow selection and validation of propagation models that best represent conditions at OCS borrow areas.

Sound Impact Data Gap #1 *Overall rank: 10*

Collaborate with other agencies to determine the behavioral responses and physiological effects of dredging-associated sounds on marine mammals and sea turtles

In addition to the acoustic characteristics of the sound, the audibility and behavioral responses of marine mammals and sea turtles are dependent on many factors, such as the physical environment (e.g., water depth, substrate type), existing ambient sound, hearing ability of the animal, and behavioral context of the animal (e.g., feeding, migrating, resting). Thus, additional studies are needed to determine the behavioral responses to sound exposures and those sound levels that cause physiological effects for marine mammals and sea turtles. Other agencies and researchers are conducting such studies for a wide range of conditions and sound sources. BOEM should continue to be engaged in these studies and participate with NOAA in determining appropriately revised acoustic thresholds.

BOEM should also work closely with NOAA during their development of revised acoustic thresholds for marine mammals. As more data on sound levels generated during dredging operations and the effects of sound on animal behavior and health are available, BOEM and NOAA need to agree on the best model(s) and input parameters to use for assessing potential impacts of dredging sounds on protected species.

Sound Impact Data Gap #2 *Overall rank: 14*

Gain a better understanding of the underwater hearing of seabirds

No literature was found that assessed the potential impacts of sound from OCS sand dredging and conveyance operations on foraging seabirds. There are no measures of underwater hearing of any diving bird, though studies are being conducted to train captive scaup to respond to sound underwater so behavioral audiograms can be compared with other measurements. BOEM should encourage and participate in the design of appropriate studies that include exposures to sound characteristic of dredging operations.

5.5 WHAT ARE THE IMPACTS OF OCS DREDGING ON FORAGING SEABIRDS?

The Mid-Atlantic is an important wintering and migration area for large numbers of foraging seabirds, such as scoters and long-tailed ducks (benthos eaters) and northern gannet, red-throated loon, common loon, gulls, terns, and cormorants (fish eaters). Studies have shown that seabirds do avoid areas of high-intensity vessel traffic, and that flocks will be flushed by the passage of vessels traveling within 500-1,000 m of the flock. Where and when seabirds concentrate around offshore sand shoals, they could be disturbed by dredging operations for the weeks and months of nearly continuous operations at the borrow area. However, it is likely that flocks would move to areas outside the dredging operations. Dredges and associated vessel traffic to/from the site could also temporarily disturb flocks along the transit routes, though studies have shown that they return quickly after vessel passage. Thus, any impacts are thought to be temporary; however, so little is known that the consequences are uncertain.

Seabird Impact Data Gap #1 *Overall rank: 11*

Better understand the habitat value of OCS shoals for seabirds

The potential, long-term impacts on seabirds of repeated dredging and ultimate deepening or removal of offshore shoals cannot be assessed with the current limited understanding of the importance of sand shoals as foraging habitat. Forsell and Watson (2006) concluded that seabirds are attracted to the shoals, but wanted to know why, with the assumption being that the shoals were important feeding grounds. However, little is known about the patterns of use of the OCS shoals. There is a concerted effort by the Sea Duck Joint Venture to conduct field studies of the linkages among breeding, overwintering, staging, and molting areas in the Atlantic region, to better understand sea duck declines and limiting factors. One of their study objectives is to identify the most important wintering and staging areas for sea ducks and estimate degrees of site fidelity, focusing on surf scoter, black scoter, white-winged scoter, and long-tailed duck. They have collected four years of data on wintering and migratory areas along the Atlantic coast. Finer-scale surveys aimed at more precise determination of flock locations and sampled more comprehensively over time, tides, weather conditions, different sea states, etc. are needed to better understand the existing data on seabird distributions and short-term behavior relative to sand shoals on the OCS. In addition, more information is needed to understand the prey consumed by key species and how sand mining may affect these species.

5.6 RELOCATION TRAWLING IN THE OCS: WHAT ARE THE EFFECTS AND EFFECTIVENESS?

Dredging Mitigation Measure Data Gap #3 *Overall rank: 13*

Determine the need for and effectiveness of relocation trawling in the OCS as a method to reduce entrainment of sea turtles; assess impacts of bycatch on other protected species

Based on the available information, relocation trawling appears to be an appropriate mitigation tool to reduce incidental sea turtle take in areas with abundant sea turtles present at the time of dredging, and particularly for navigational dredging. However, relocation trawling does have potential negative consequences on sea turtles directly, as well as potential impacts to other species captured as bycatch, increase in the dredging duration when sea conditions do not allow trawling, and vessel safety risks from snags and collisions. Therefore, relocation trawling should be considered where it will be most effective. An analysis of relocation trawling data for OCS dredging projects, similar to that conducted by Dickerson et al. (2007) for USACE dredging projects in navigational channels, would provide more data on which BOEM and NMFS could evaluate the effectiveness of these mitigation measures in the OCS in general and in specific geographic regions.

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The Department of the Interior Mission

As the Nation's principal conservation agency, the Department of the Interior has responsibility for most of our nationally owned public lands and natural resources. This includes fostering the sound use of our land and water resources; protecting our fish, wildlife, and biological diversity; preserving the environmental and cultural values of our national parks and historical places; and providing for the enjoyment of life through outdoor recreation. The Department assesses our energy and mineral resources and works to ensure that their development is in the best interests of all our people by encouraging stewardship and citizen participation in their care. The Department also has a major responsibility for American Indian reservation communities and for people who live in island communities.

The Bureau of Ocean Energy Management Mission

The Bureau of Ocean Energy Management (BOEM) promotes energy independence, environmental protection, and economic development through responsible, science-based management of offshore conventional and renewable energy.