7.0 POTENTIAL EFFECTS

One of the primary purposes of this project is to provide site-specific information for decisions on requests for non-competitive leases from other local, State, and Federal agencies. The information may be used to determine whether stipulations need to be applied to a lease. The information also may be incorporated into an Environmental Assessment (EA) or Environmental Impact Statement (EIS), if so required.

Environmental impact analyses of mining operations should be based on commodity-specific, technology-specific, and site-specific information, whenever possible (Hammer et al., 1993a,b). First, the specific mineral of interest and the technological operations for a specific mining operation need to be defined because these two parameters determine the impact producing factors that need to be considered. Once the impact producing factors are known, this information can be translated into statements concerning the impacts that might occur to the full suite of potentially affected environmental resources that may need to be addressed, including geology, chemical and physical oceanography, air quality, biology, and socioeconomics. Then, decisions can be made regarding the type of mitigation necessary to determine the preferred alternative for a specific marine mining operation to acquire project approval.

This section focuses on providing information on potential impacts related to physical processes and biological considerations of sand mining for beach nourishment from nine sand resource areas offshore central east Florida. Sand for beach replenishment is the commodity of interest. Two primary dredging technologies are available for offshore sand mining operations, depending on distance from source to project site, the quantity of sand being dredged, and the depth to which sand is extracted at a site (Herbich, 1992). They are 1) cutterhead suction dredge, where excavated sand is transported through a direct pipeline to shore, and 2) hopper dredge, where sand is pumped to the hopper, transported close to the replenishment site, and pumped to the site through a pipeline from the hopper or from a temporary offshore disposal area close to the beach fill site. As a general rule, cutterhead suction dredging is most effective for projects where the sand resource is close to shore (within 8 km), the dredging volumes are large (>8 mcm), and the excavation depth is on the order of 2.5 to 4 m (A. Taylor, 1999, pers. comm., Bean Stuyvesant, LLC). Hopper dredging becomes a more efficient procedure when the sand resource areas are greater than 8 km from shore, dredging volumes are relatively small (<2 mcm), and the excavation depth at the sand resource area is less than 2 m (A. Taylor, 1999, pers. comm.). Ultimately, a combination of these factors will be evaluated by dredgers to determine the most cost-effective method of sand extraction and beach replenishment for a given project. Availability of dredging equipment also may be a factor for determining the technique to be used; however, the number of cutterhead suction and hopper dredges in operation is about equal in the industry today (A. Taylor, 1999, pers. comm.). As such, both technologies will be evaluated for potential biological effects.

7.1 POTENTIAL SAND BORROW SITES

Nine potential sand resource areas were identified offshore central east Florida in Federal waters by the FGS and MMS. Each area has specific geological and geographical characteristics that make it more or less viable as a sand resource for specific segments of coast. Areas A1, B1, B2, C1, and D2 contain borrow sites with the greatest potential for future use.

All sand resource areas are very similar geologically (medium-to-coarse sand size ridge deposits with relief of 2 m or greater and resource volumes of at least 1 mcm). All identified potential sand borrow sites are of great interest to the State, primarily due to their proximity to eroding beaches critical for storm protection and recreation. Although six potential sand borrow sites were designated as ones with greatest potential, it is possible that sand could be dredged from intervening offshore areas and on other offshore shoals.

The amount of dredging that occurs at any site is a function of Federal, State, and local requirements for beach replenishment. It is impossible to predict the exact sand quantities needed in the foreseeable future, so a representative value for any given project was estimated based on discussions with MMS and State personnel. Preliminary analysis of short-term impacts (storm and normal conditions) at specific sites along the coast landward of sand borrow sites indicates that about 1 mcm of sand could be needed for a given beach replenishment event. Long-term shoreline change data suggest that a replenishment interval of about 10 to 20 years may be required to maintain beaches. This does not consider the potential for multiple storm events impacting the coast over a short time interval, nor does it consider longer time intervals without destructive storm events. Instead, the estimate represents average change over decades that is a reasonable measure for coastal management applications.

Given the quantity of 1 mcm of sand per beach replenishment event, the surface area covered for evaluating potential environmental impacts is a function of average dredging depth. Two factors should be considered when establishing dredging practice and depth limits for proposed extraction scenarios. First, regional shelf sediment transport dynamics should be evaluated to determine net transport directions and rates. It is good sand resource management practice to dredge the leading edge of a migrating shoal because infilling of dredged areas occurs more rapidly at these sites (Byrnes and Groat, 1991; Van Dolah et al., 1998). Second, shoal relief above the ambient shelf surface should be a determining factor controlling dredging depth. Geologically, shoals form and migrate on top of the ambient shelf surface, indicating a link between fluid dynamics, sedimentology, and environmental evolution (Swift, 1976). As such, average shoal relief is a reasonable threshold for maintaining environmentally-sound sand extraction procedures.

A primary question addressed by the modeling efforts relates to sediment transport and infilling estimates at potential borrow sites and the impact of dredging operations on these estimates. Combined wave-current interaction (waves mobilize the seabed and currents transport the sediment) at borrow sites results in a net direction of transport into and out of potential sand resource areas. Historical sediment transport dynamics suggest that the net direction of sediment movement is from north to south, and the rate at which sand moves along the shelf varies.

7.2 WAVE TRANSFORMATION MODELING

Excavation of borrow sites in the nearshore can affect offshore wave heights and the direction of wave propagation. The existence of offshore topographic relief can cause waves to refract toward the shallow edges of borrow sites. This alteration to the wave field by a borrow site may change local sediment transport rates, where some areas may experience a reduction in transport, while other areas may show an increase. To determine the potential physical impacts associated with dredging borrow sites offshore central east Florida, wave transformation modeling and sediment transport potential calculations were performed for existing and post-dredging bathymetric conditions. Comparison of results for existing and post-dredging conditions illustrated the relative impact of borrow site excavation on wave-induced coastal processes. Although the interpretation of wave modeling results was relatively straightforward, evaluating the significance of predicted changes for accepting or rejecting a borrow site was more complicated (see Section 4.0 for details).

7.2.1 Offshore Cape Canaveral

Canaveral Shoals, the complex of ridges and troughs that extend southeast from Cape Canaveral, caused significant increases in wave height as waves propagated over this area. As 1.0 m, 7.7 sec waves from the east-southeast (Case 3A) refracted around the shoals, wave heights increased by 0.5 m over offshore wave conditions. In the shoal field northeast of the Cape, wave heights increased by about 0.3 m above offshore wave heights. Wave direction changes also were observed in these areas. A greater degree of wave refraction was illustrated for longer period waves. For a 1.6 m, 14.3 sec wave propagating from the east-northeast (Case 6A), wave direction for some nearshore regions adjacent to the Cape changed more than 45 degrees, following the gradient in bathymetric contours (see Figure 4-17). Largest waves in the model domain occurred at shoals northeast of Port Canaveral (1.3 m higher than offshore waves). At shoals in the vicinity of the borrow site in Area A1, wave heights increased to a maximum of 2.8 m, 1.2 m above offshore conditions. Shoals tended to refract wave energy and caused focusing (wave convergence) near the Cape. However, the coast south of the Cape illustrated reduced wave heights (wave divergence).

Post-dredging wave height changes for Case 3A illustrated a maximum wave height increase of 0.2 m and maximum wave height decrease in the shadow zone of the site of 0.3 m. The overall area of influence for the borrow site in Area A1 extended approximately 14 km north of the Cape to about 4 km south of Port Canaveral. Similar wave height differences were illustrated for Case 6A. Maximum change in post-dredging wave heights was 0.7 m, substantially greater than change observed at other sites. The area of greatest wave height increase occurred at the northwest corner of the site. Wave heights did not increase by the same amount at the southwest corner, likely due to local bathymetry and geometry of the site. Deeper excavation depths at the northwest corner cause a greater degree of wave refraction. The longshore extent of influence was similar to that of Case 3A, but its location shifted slightly southward due to the direction of wave propagation. However, for all wave simulation cases, the impact of borrow site excavation on wave height and direction changes was minor relative to natural variability of the local wave climate and transport regime.

7.2.2 Offshore Sebastian Inlet

Wave model output for 1.9 m, 6.9 sec waves propagating from the north-northeast (Case 1B), offshore Sebastian Inlet at borrow sites in Areas B1 and B2, illustrated minor changes throughout the model domain. The shoal encompassing the borrow site in Area B1 had the greatest influence on wave propagation in the region, although effects were small because the shoal had a minimum depth of approximately 12 m NGVD. For wave Case 10B (H_s = 1.7 m, T_{peak} = 10.8 sec), wave height was similar but peak wave period was longer, resulting in greater wave refraction. Wave heights shoreward of the shoal were approximately 0.2 m greater than wave heights on the seaward side of the feature.

Changes in the wave field caused by dredging at borrow sites in Areas B1 and B2 illustrated minor impacts for the Area B model domain. For wave Case 1B, borrow sites had a limited influence on waves over a long section of coast (>30 km), but changes on the order of 0.01 m occurred along 2.5 km of coast landward of the borrow site in Area B1 (see Figure 4-22). Maximum change in wave height was approximately 0.10 m at Area B1 and 0.12 m at the borrow site in Area B2. Even though the borrow site in Area B2 was smaller than that in Area B1 (i.e., less sediment dredged), B2 had a slightly greater impact on local wave heights. This apparent paradox was due to subtle changes in bathymetry relative to borrow site geometry.

Wave differences computed for Case 10B indicated that changes to the wave field resulting from dredging at sand borrow sites in Areas B1 and B2 were more pronounced than for wave Case 1B. The length of shoreline influenced by changes in wave propagation from the two borrow sites was approximately 20 km; however, greatest changes occurred within a 12 km stretch of coast. At Area B1, maximum changes in wave height were 0.13 m, very similar to those computed for the borrow site in Area B2. Although the magnitude of maximum wave height change for wave Case 10B was only slightly larger than 1B, shoreline impacts associated with 10B were quite a bit greater. Long period waves of Case 10B were affected more by bathymetry in deeper water, causing larger areas of waves on the shoals to be impacted by dredging at borrow sites. This process resulted in a broader area of impacted shoreline. However, for all wave simulation cases offshore Sebastian Inlet, the impact of borrow site excavation on wave height and direction changes was minor relative to natural variability of the local wave climate and transport regime.

7.2.3 Offshore St. Lucie Inlet

For the wave model domain offshore St. Lucie Inlet, 1.5 m, 7.5 sec waves propagating from the northeast (Case 2C) illustrated slight wave focusing at shoals within the designated borrow site boundaries. The minimum depth at Site C1 north was 7.6 m NGVD, and 5.4 m NGVD was the minimum depth at Site C1 south. Because shallower depths exist in these areas, waves passing over the shoals refracted toward the shoreline sooner than in other areas the same distance offshore. Waves refracting over the shoals produced an area of increased wave heights landward of each shoal and a corresponding area of decreased wave heights immediately south of both sites. For C1 north, maximum wave height increase was 0.18 m, and the maximum decrease was 0.39 m. Similar changes were observed at C1 south, where the maximum increase in wave height was 0.13 m and the maximum decrease was 0.33 m. For wave Case 10C, a 1.1 m, 11.1 sec wave from the east, wave height changes at C1 north and C1 south were not as large as those for Case 2C, but wave energy was still focused behind the shoals. This focusing caused a zone of increased wave heights that extended to the shoreline. Unlike the results of Case 2C, where wave height changes at the borrow sites were more pronounced, the resulting wave shadow zone diffused more as it approached the shoreline (due to the shorter peak wavelength of Case 2C).

For wave Case 2C, wave height differences resulting from dredging Sites C1 north and C1 south indicated a strong interaction between the two sites because C1 south was partially within the shadow zone of C1 north. The alignment of borrow sites caused a single area of increased wave heights at the shoreline (approximately 4 km long) and a more diffuse zone of reduced wave heights (extending 12 km south toward St. Lucie Inlet). At the borrow sites, maximum wave height increase was 0.09 m, and the maximum wave height decrease was 0.15 m. Wave height differences for wave Case 10C illustrated that the borrow sites have an overlapping influence at the shoreline for waves propagating from the east, even though one site was not directly in the shadow of the other. The total length of affected shoreline was approximately 16 km, and changes at the borrow sites were similar in magnitude to those for Case 2C. The resulting wave shadow zone for the two borrow sites was less diffuse due to a longer peak wavelength for this model case.

7.2.4 Offshore Jupiter Inlet

The primary bathymetric feature impacting wave propagation in modeled Area D is located approximately 5.6 km offshore Jupiter Inlet. The shoal has a minimum water depth of 11.7 m NGVD, and the borrow site in and adjacent to Area D2 lies along the seaward margin of the shoal at the Federal-State boundary. For wave Case 1D (1.4 m, 6.9 sec wave from the NNE), the shoal produced a slight focusing of waves seaward of the shoal and an area of reduced wave heights 2.6 km along the shoreline north of Jupiter Inlet. Similar results were documented for wave Case 9D, a 1.3 m, 13.0 sec wave from the ENE. Wave heights increased behind the shoal, and a 4.9 km stretch of coastline north of Jupiter Inlet experienced increased wave heights. Maximum wave height increase caused by the shoal for Case 9D was 0.4 m, whereas Case 1D produced a 0.1 m change in wave height.

Wave height changes resulting from dredging Borrow Site D2 showed greatest change at the north end of the site where the deepest excavation occurred. The maximum increase and decrease in wave height that resulted for wave Case 1D was 0.04 and 0.05 m, respectively. This small change relative to changes at borrow sites to the north was due to greater water depths at and seaward of the borrow site. For wave Case 9D, two shadow areas of reduced wave heights propagated from two separate areas within the borrow site, but join to form one shadow on the shoreward side of the shoal. This change pattern occurred because the original bathymetry within Site D2 contained two elevation peaks approximately 1.5 m higher than the surrounding shoal surface. Overall, wave simulation cases offshore Jupiter Inlet illustrated minor wave height and direction changes in response to borrow site excavation relative to natural variability of the local wave climate and transport regime.

7.3 CURRENTS AND CIRCULATION

Circulation patterns along the central east Florida coast near potential offshore borrow sites were investigated using current meter observations obtained offshore St. Lucie Inlet and over Thomas Shoal, seaward of Sebastian Inlet. Analysis of historical data indicated that circulation patterns consisted predominantly of along-shelf currents that reversed direction approximately every 2 to 10 days. Current reversals were found weakly correlated with local wind stress; literature suggested that subtidal variability was due to meanders or spin-off eddies of the Florida Current. Peak speeds were on the order of 40 to 50 cm/sec at mid-shelf and inner-shelf locations and were directed either upshelf (to the north-northwest) or downshelf (to the south-southeast). Strongest currents were most commonly directed to the north. Tidal currents contributed significantly to inner-shelf current observations;

however, these observations were obtained near the tidally-dominated St. Lucie Inlet and may not be reflective of inner shelf regions removed from major coastal inlets.

ADCP measurements in the vicinity of Thomas Shoal offshore Sebastian Inlet also were dominated by along-shelf flows that correlated with seasonal changes in wind. May survey conditions were dominated by winds from the south, while September survey conditions were characterized by short wind events from the north. Current measurements illustrated a mean flow directed to the north during spring and to the south in fall. This seasonal directionality of flow was supported by historical data and literature regarding observations on the mid-shelf and inner-shelf where sand resource areas have been identified. Strongest currents flowed to the south at 30 cm/sec during the September survey in response to northerly winds.

Seasonal wind variations have been shown to induce downwelling in winter and upwelling in summer for central east Florida. Current variability not well explained by wind stress may be an indirect response to the Florida Current. The Florida Current flows northward past the study area on the outer shelf (Lee et al., 1985). Instabilities in the Florida Current create spin-off eddies that have been documented on the inner shelf (Smith, 1981). Potential influences of the Florida Current were observed in spring survey results, illustrated by the presence of a strong northward flowing bottom current in the presence of weak winds and surface currents. Florida Current effects may enhance northerly flows during winter and spring months in the study area.

In shallow waters, over shoals and adjacent to tide-dominated inlets such as St. Lucie, cross-shelf tides may influence current velocities. May and September field data showed onshore currents dominated across the shoal. During the May survey, onshore currents were enhanced by flood tide. Tidal dependence was not observed during the September survey. On the inner- to mid-shelf, in the vicinity of the sand resource areas, tidal effects are secondary to wind effects. In the presence of local bathymetric features, such as Thomas Shoal, steering and sheltering of flow across the shoal were observed. Under average conditions, currents were steered onshore across the shoal. In the presence of dominant winds, near-bottom currents flowed parallel to bathymetric contours.

The analysis of current patterns resulting from this study suggests proposed sand mining will have negligible impact on large-scale shelf circulation. The proposed sand mining locations are small relative to the entire shelf area, and it is anticipated that resulting dredging will not remove enough material to significantly alter major bathymetric features in the region. Therefore, the forces and/or geometric features that principally affect circulation patterns will remain relatively unchanged.

7.4 SEDIMENT TRANSPORT

Current measurements and analyses, and wave transformation modeling, provided baseline information on incident processes impacting coastal environments under existing conditions and with respect to proposed sand mining activities for beach replenishment. Ultimately, the most important information for understanding physical processes impacts from offshore sand extraction is changes in sediment transport dynamics resulting from potential sand extraction scenarios relative to existing conditions.

Three independent sediment transport analyses were completed to evaluate physical environmental impacts due to sand mining. First, historical sediment transport trends were quantified to document regional, long-term sediment movement throughout the study area using historical bathymetric data sets. Erosion and accretion patterns were documented, and sediment transport rates in the littoral zone and at offshore borrow sites were evaluated to assess potential changes due to offshore sand dredging activities. Second, sediment transport patterns at proposed offshore borrow sites were evaluated using wave modeling results and current measurements. Post-dredging wave model results were integrated with regional current measurements to estimate sediment transport trends for predicting borrow site infilling rates. Third, potential longshore sediment transport was computed using wave modeling output to estimate potential impacts along the coast (beach erosion and accretion). All three methods were compared for documenting consistency of measurements relative to predictions, and potential physical environmental impacts were identified.

7.4.1 Historical Sediment Transport Patterns

Regional geomorphic changes between 1877/83 to 2002 were analyzed for assessing long-term, net coastal sediment transport dynamics. Although these data did not provide information on potential impacts of sand dredging from proposed borrow sites, they did provide a means of verifying predictive sediment transport models relative to infilling rates at borrow sites and longshore sand transport.

Shoreline position and nearshore bathymetric change documented four important trends relative to study objectives. First, the predominant direction of sediment transport on the continental shelf and along the outer coast between Cape Canaveral and Jupiter Inlet was north to south. The greatest amount of shoreline change in this study was associated with beaches adjacent to Cape Canaveral, Port Canaveral Entrance, and beaches south of St. Lucie Inlet.

Second, the most dynamic features within the study area, in terms of nearshore sediment transport are the beaches and shoals associated with Cape Canaveral. Areas of significant erosion and accretion are documented between 1956 and 1996 at Cape Canaveral, reflecting wave and current dynamics and the contribution of littoral sand transport from the north to shoal and spit migration. Depositional zones also were prominent in the shoal regions along the inner shelf from Fort Pierce south to Jupiter Inlet. Large quantities of carbonate and shell fragments observed in sediment samples collected from shoals in this region indicated that much of the deposition in this portion of the study area may have been locally produced.

Third, alternating bands of erosion and accretion documented between 1956 and 1996 at Cape Canaveral illustrated steady reworking of the upper shelf surface as sand ridges migrated from north to south. The process by which this was occurring at Sand Resource Area A1 suggested that the borrow site in this region would fill with sand transported from the adjacent seafloor at rates ranging from 88,000 to 119,000 m^3/yr. Areas of erosion and accretion documented between 1929/31 and 1929/73 between Port Canaveral Entrance and Jupiter Inlet indicated the amount of sediment available for infilling sites south of Port Canaveral Entrance was between 38,000 and 113,000 m^3/yr.

Finally, net longshore transport rates determined from seafloor changes in the littoral zone between Cape Canaveral and Port Canaveral Entrance, in conjunction with dredging records for Port Canaveral entrance, indicated maximum transport rates near Cape Canaveral, with lower rates south of the entrance. Net longshore transport north of Port

Canaveral entrance was estimated at about 236,000 m^3/yr (308,000 cy/yr). South of the Port, rates have been estimated to range from 119,000 m^3/yr (155,000 cy/yr) immediately south of the entrance to 140,000 to 184,000 m^3/yr (183,000 to 240,000 cy/yr) between Fort Pierce and Jupiter Inlet.

7.4.2 Sediment Transport Modeling at Potential Borrow Sites

In addition to predicted modifications to the wave field, potential sand mining at offshore borrow sites resulted in minor changes in sediment transport pathways in and around potential dredging sites. Modifications to bathymetry caused by sand mining only influenced local hydrodynamic and sediment transport processes in the offshore area. Although wave heights changed at the dredged borrow sites, areas adjacent to these sites did not experience dramatic changes in wave or sediment transport characteristics.

Initially, it is anticipated that sediment transport at borrow sites will occur rapidly after sand dredging is completed. For water depths at the proposed borrow sites, minimal impacts to waves and regional sediment transport are expected during infilling. The characteristics of sediment that replaces borrow material during infilling will vary based on location, time of dredging, and storm characteristics following dredging episodes. Average transport rates ranged from a minimum of about 5,000 m^3/yr (Site D2) to a high of about 538,000 m^3/yr (Site A1), while the infilling time varied from 25 to >500 years. Site A1 had the greatest infilling rate due to its shallow water depth relative to the other sites and its large perimeter. Because Site A1 is in shallow water, wave-induced and wind-driven currents were larger than at deeper sites, and more sediment was mobile in the proximity of the borrow site. Furthermore, sites that have a larger surface area generally trap more sediment in a given time period.

Total infilling times were computed using the total design excavated volume divided by the computed infilling rates, and thus represent the length of time required to fill a site that was excavated to the total design depth during a single dredging event. Site D2 has the longest total infilling time, resulting from relatively deep water depths and the low infilling volume rate computed for the area. Even though Site A1 had the largest sand extraction volume, the infilling time was shortest due to its large sediment infilling rate. The analysis of borrow site infilling time assumed a constant rate of transport from each direction and does not include the effects of modified bathymetry. For example, as a dredged site begins to fill, sediment transport dynamics may change. As such, sediment transport rates will fluctuate as a borrow site evolves during infilling. These estimated infilling times are most useful as a relative guide for borrow site infilling rather than an absolute indicator of exactly how long it takes for the borrow site to fill. The analysis performed provided a reasonable estimate of infilling times for resource management purposes.

7.4.3 Nearshore Sediment Transport Potential

Comparisons of average annual sediment transport potential were performed for existing and post-dredging conditions to indicate the relative impact of dredging to longshore sediment transport processes. Mean sediment transport potential calculated for the shoreline south of Port Canaveral indicated strong net southerly transport of approximately 500,000 m^3/yr, which gradually reduced to approximately 300,000 m^3/yr at the southern limit of the model grid. The transport significance envelope was largest (approximately $\pm$300,000 m^3/yr) north of Cape Canaveral and in the southern half of the modeled area (see Figure 4-32).

Mean transport potential computed for Area B (offshore Sebastian Inlet) indicated that net transport potential was generally less than 100,000 m^3/yr to the south. There is an approximate ±500,000 m^3/yr range in net transport rates. Computations indicated that it was possible in some years for net transport potential to be northward directed. Near Vero Beach, net transport potential was to the south at around 500,000 m^3/yr and annual variation in net transport potential was similar (approximately ±500,000 m^3/yr). This may be due to a change in shoreline orientation that occurred at this point.

Computed mean annual transport potential for modeled Area C (just north of St. Lucie Inlet) was to the south, ranging from approximately 400,000 m^3/yr at the northern boundary of the grid to approximately 100,000 m^3/yr at the southern limit near St. Lucie Inlet (see Figure 4-36). Sand transport potential calculations for 20 individual years indicated that the annual variability in transport potential had a range of approximately ±400,000 m^3/yr to the north that gradually decreases to approximately ±200,000 m^3/yr at the southern limit of the modeled area. Along some sections of the modeled shoreline, it was possible to have net northerly-directed transport during some years.

Net transport along the coastline adjacent to Jupiter Inlet varied from about 200,000 m^3/yr to the south near the northern limit of the area to about 500,000 m^3/yr to the south near Jupiter Inlet (see Figure 4-38). Results from the 20 individual modeled years showed that the annual variability ranged from approximately ±150,000 m^3/yr in the northern part of the area to approximately ±300,000 m^3/yr at the southern extent of the model grid. Similar to modeled areas to the north, the year with greatest modeled southerly transport was 1980, and the year with greatest northerly transport was 1990. As with the entire study area south of Cape Canaveral, net transport potential was always to the south and transport variability was large.

The significance of changes to longshore transport along the modeled shoreline resulting from dredging proposed borrow sites to their maximum design depths was determined using the method described in Kelley et al. (2004). Model output for the region south of Cape Canaveral (Area A) indicated that the significance envelope was approximately 20% of the mean computed net transport potential in the area of greatest impact from the borrow site in Area A1. The maximum modeled decrease in south-directed transport for post-dredging conditions was about a 40,000 m^3/yr (within the transport significance range), just south of Port Canaveral. For the Area B borrow sites (adjacent to Sebastian Inlet), the transport significance range was nearly consistent at about ±100,000 m^3/yr. The impacts that resulted from numerically dredging Borrow Sites B1 and B2 are within this transport range, indicating that these sites would not produce significant modifications to coastal processes along the shoreline. The largest calculated differences between existing and post-dredging transport potential occurred north of Sebastian Inlet (where the transport rate becomes more southerly by 30,000 m^3/yr) and just south of the inlet (where transport rates become less southerly by 30,000 m^3/yr).

For Borrow Sites C1 north and C1 south (north of St. Lucie Inlet), the computed longshore transport significance range was approximately ±100,000 m^3/yr at the northern limit of the area and ±50,000 m^3/yr at the southern limit. Potential impacts from dredging Sites C1 north and C1 south to a maximum excavation depth of -12 m indicated that the significance envelope was exceeded along a 2-km length of shoreline approximately 18 km north of St. Lucie Inlet. At the point of maximum dredging-induced change along the shoreline, the significance level was ±60,000 m^3/yr, and the computed change in transport potential was 85,000 m^3/yr. As designed, borrow site configuration may not be acceptable. If a borrow site redesign were required, the most likely change would be a reduction in maximum dredging depth to reduce site impacts.

The envelope of significant change in potential longshore transport rates under natural wave propagation conditions for Borrow Site D2 ranged from approximately $\pm$50,000 m^3/yr in the north to $\pm$100,000 m^3/yr in the south, with a maximum of approximately $\pm$150,000 m^3/yr occurring just north of Jupiter Inlet. Modeled dredging impacts to transport potential for Site D2 were minimal; predicted changes were well within the transport variability significance range. Maximum dredging impacts to transport potential were approximately $\pm$10,000 m^3/yr. Small impacts for this area (compared with previous modeled areas) resulted from larger borrow site depths, smaller excavation volume, and the sheltering effect of the shoal landward of D2.

Overall, it was determined that no significant changes in longshore sediment transport potential would result from modeled borrow site configurations for Areas A, B, and D. However, the proposed sites in Area C do have significant impacts to transport potential along the shoreline. Therefore, Area C sites should be redesigned so impacts are within acceptable limits, most likely by reducing the maximum depth of excavation at the sites.

7.5 BENTHIC ENVIRONMENT

The purpose of this section is to address potential effects of offshore sand dredging on benthic organisms, including analyses of recolonization periods and success following cessation of dredging. This section is divided into three parts. The first two parts provide reviews of information from existing literature on effects and recolonization. The first part (Section 7.5.1) summarizes potential impacts to benthic organisms from physical disturbance of dredging, which causes removal, suspension/dispersion, and deposition of sediments. The second part (Section 7.5.2) is a synthesis of information concerning recolonization periods and success. Finally, the third part (Section 7.5.3) provides predictions of impacts and recolonization relative to the central east Florida sand resource areas.

Ecological effects of marine mining and beach nourishment operations have been reviewed by numerous authors (Thompson, 1973; Naqvi and Pullen, 1982; Nelson, 1985; Cruickshank et al., 1987; Goldberg, 1989; Grober, 1992; Hammer et al., 1993a,b; National Research Council, 1995). Effects vary from detrimental to beneficial, short to long term, and direct to indirect (National Research Council, 1995).

Most reviews on the effects of beach nourishment operations have focused on potential impacts at the beach. Comprehensive assessments of effects on biological resources at open ocean sand borrow sites have been limited (National Research Council, 1995). Alterations to biological resources in offshore sand borrow sites are generally of longer duration, and the consequences of those changes have not been well-defined (National Research Council, 1995). The remainder of this section focuses on potential impacts of dredging operations at offshore sand resource areas.

7.5.1 Effects of Offshore Dredging on Benthic Biota

The primary impact producing factor relative to dredging offshore sand borrow sites is mechanical disturbance of the seabed. This physical disruption includes removal, suspension/dispersion, and deposition of dredged material, which may make the benthic environment less suitable for some species and better for other biota. The following subsections focus on potential effects of these physical processes on benthic biota.

7.5.1.1 Sediment Removal

Physical removal of sediments from a borrow site removes benthic habitat along with infauna and epibiota that are incapable of avoiding the dredge, resulting in drastic reductions in number of individuals, number of species, and biomass. Extraction of habitat and biological resources may in turn disrupt the functioning of existing communities. Removal of benthic resources is of concern because the resources are important in the food web for commercially and recreationally important fishes and invertebrates and contribute to the biodiversity of the pelagic environment through benthic-pelagic coupling mechanisms. These mechanisms include larval transport and diurnal migrations of organisms, which may have substantial impact on food availability, feeding strategies, and behavioral patterns of other members of the assemblage (Hammer and Zimmerman, 1979; Hammer, 1981). In some cases, dredging borrow sites may create new and different habitats from surrounding substrates, which could result in beneficial impacts in terms of increased habitat complexity and biodiversity of an area.

The influence of sediment composition on benthic community composition has been recognized since the pioneer studies of Peterson (1913), Jones (1950), Thorson (1957), and Sanders (1958). However, more recent reviews suggest that precise relationships between benthic assemblages and specific sediment characteristics are poorly understood (Gray, 1974; Snelgrove and Butman, 1994; Newell et al., 1998). Sediment grain size, chemistry, and organic content may influence recolonization of benthic organisms (McNulty et al., 1962; Thorson, 1966; Snelgrove and Butman, 1994), although the effects of sediment composition on recolonization patterns of various species are not always significant (Zajac and Whitlatch, 1982). Because the complexity of soft-sediment communities may defy any simple paradigm relating to any single factor, Hall (1994) and Snelgrove and Butman (1994) proposed a shift in focus towards understanding relationships between organism distributions and the dynamic sedimentary and hydrodynamic environments. It is likely that the composition of benthic assemblages is controlled by a wide array of physical, chemical, and biological factors that interact in complex ways and are variable with time.

Removal of sand resources can expose underlying sediments and change the sediment structure and composition of a borrow site, consequently altering its suitability for burrowing, feeding, or larval settlement of some benthic organisms. Many studies show decreases in mean grain size, and in some cases, increases in silt and clay in borrow sites following dredging (National Research Council, 1995). Changes in sediment composition could potentially prevent recovery to an assemblage similar to that which occurred in the borrow site prior to dredging and could by implication affect the nature and abundance of food organisms for commercial and recreational fishery stocks (Coastline Surveys Limited, 1998; Newell et al., 1998). Selective bottom feeders could be affected due to removal of specific prey species from borrow sites. The State of Florida and Florida Keys National Marine Sanctuary (FKNMS) prohibited collection of “live sand” (i.e., sand material, typically containing a high diversity of algal, bacterial, and macroinvertebrate species, used in the aquarium trade industry) within the FKNMS because the sand substrate is an important habitat for grazers and detritovores and removal of this habitat was determined to adversely impact marine productivity, fisheries, wildlife habitat, and water quality (Ruebsamen, 2003).

Removal of sediments from borrow sites can alter seabed topography, creating pits that may refill rapidly or cause detrimental impacts for extended periods of time. The term "borrow site" can be misleading because often material is returned only by natural sediment transport processes. Nearly 12 years may be required for some offshore borrow sites to refill to pre-dredge profiles (Wright, 1977), and other borrow sites have been known to remain well-defined 8 years after dredging (Marsh and Turbeville, 1981; Turbeville and Marsh, 1982). Intentionally locating borrow sites in highly depositional areas may dramatically reduce the time for refilling (Van Dolah et al., 1998). In general, shallow dredging over large areas causes less harm than small but deep pits, particularly pits opening into a different substrate surface (Thompson, 1973; Applied Biology, Inc., 1979). Deep pits also can hamper commercial trawling activities and harm level-bottom communities (Thompson, 1973). If borrow pits are deep, current velocity is reduced at the bottom, which can lead to deposition of fine particulate matter and in turn a biological assemblage much different in composition than the original. Increasing water depths and turbidity from dredging may reduce the photic zone for benthic primary producers. Recovery of the physical environment and benthic assemblages to pre-dredging conditions will probably take decades for a deep pit dredged 3.6 km offshore Coney Island (Barry A Vittor & Associates, Inc., 1999). Deep holes may decrease dissolved oxygen to hypoxic or anoxic levels and increase hydrogen sulfide levels (Murawski, 1969; Saloman, 1974; National Research Council, 1995). Not all impacts from dredge pits are detrimental. Borrow pits are known to attract numerous fishes (Gustafson, 1972; Michals, 1997; Weakley, 2001), even to the extent that some dredge holes offshore east Florida have been referred to as "reefs in reverse" (Weakley, 2001). Borrow pits also provide resting places for loggerhead sea turtles (Michals, 1997).

Seabed topography and benthic communities can be altered when sediment is removed by dredging bathymetric peaks such as ridges or shoals rather than level sea bottoms or depressions. Little information exists regarding the relationship between biological assemblages and removal of shoals by dredging. Numerous benthic organisms and fishes inhabit offshore shoal areas, but specifics regarding species, assemblages, and ecological interrelationships between the topographic features and associated biota are not well known. Potential long-term physical and biological impacts could occur if dredging significantly changes the physiography of shoals. The MMS has funded several studies to address environmental questions concerning use of shoals by fishes and mobile invertebrates, potential impacts to these species from offshore sand dredging, and ways to preclude or minimize long-term impacts. Burlas et al. (2001) monitored borrow sites with bathymetric high points off northern New Jersey and found that essentially all infaunal assemblage patterns recovered within 1 year after dredging disturbance except recovery of average sand dollar weight and biomass composition, which required 2.5 years.

Mechanical damages to hard bottom habitats and biota have occurred in the past from dredges digging into and equipment (e.g., anchors, cables, pipelines, etc.) being dragged across reefs (Courtenay et al., 1972, 1974; Britt & Associates, Inc., 1979; Marszalek, 1981; Blair and Flynn, 1988; Goldberg, 1989; Blair et al., 1990). These occurrences often are unnecessary and avoidable if borrow sites and adjacent areas are adequately surveyed for hard bottom prior to dredging, then mitigation and monitoring are implemented. Reef destruction can lead to shifts from coral to algal dominance (de Sylva, 1994; Umar et al., 1998; McCook et al., 2001). Randall (1958) pointed to the correlation between availability of new surfaces in the reef environment, rapid growth of algae, and development of ciguatera (toxicity in normally edible reef fishes causing human health problems). Dredging, filling, anchoring, and other anthropogenic activities leading to changes in a reef environment or coral reef destruction may increase the incidence of ciguatera, known to occur irrespective of season along the Florida east coast, including locations near the study area (de Sylva, 1994).

7.5.1.2 Sediment Suspension/Dispersion

Dredging causes suspension of sediments, which increases turbidity over the bottom. This turbidity undergoes dispersion in a plume that drifts with water currents, then suspended sediments from dredging settle. The extent of suspension/dispersion depends on a multitude of factors, incluing the type of dredging equipment, techniques for operating the equipment, amount of dredging, thickness of the dredged layer, sediment composition, and sediment transport processes. Although turbidity plumes associated with dredging often are short lived and affect relatively small areas (Cronin et al., 1970; Nichols et al., 1990), resuspension and redispersion of dredged sediments by subsequent currents and waves can propagate dredge-related turbidity for extended periods after dredging ends (Onuf, 1994). Biological responses to turbidity depend on all of these physical factors coupled with the type of organism, geographic locations, and the time of year.

Herbich and Brahme (1991) and Herbich (1992) reviewed sediment suspension caused by existing dredging equipment, and discussed potential technologies and techniques to reduce suspension and associated environmental impacts. In general, cutterhead suction dredges produce less turbidity than hopper dredges. A cutterhead suction dredge consists of a rotating cutterhead, positioned at the end of a ladder, which excavates the bottom sediment. The cutterhead is swung in a wide arc from side to side as the dredge is stepped forward on pivoting spuds, and excavated material is lifted by a suction pipe and transferred by pipeline as a slurry (Hrabovsky, 1990; LaSalle et al., 1991). Sediment suspension is caused by rotating action of the cutterhead and swinging action of the ladder (Herbich, 1992). Well-designed and properly operated cutterhead dredges can limit sediment suspension to the lower portion of the water column (Herbich and Brahme, 1991; Herbich, 1992). Turbidity can be reduced by selecting an appropriate cutterhead for a given sediment, determining the best relationship between cutterhead rotational speed and hydraulic suction magnitude, establishing a suitable swing rate for the cutterhead, and using hooded intakes, although these conditions are rarely achieved (Herbich, 1992). Measurements around properly operated cutterhead dredges show that suspended sediments can be confined to the immediate vicinity of the cutterhead and dissipate rapidly with little turbidity reaching surface waters (Herbich and Brahme, 1991; LaSalle et al., 1991; Herbich, 1992). Maximum suspended sediment concentrations typically occur within 3 m above the cutterhead and decline exponentially to the sea surface (LaSalle et al., 1991). Suspended sediments in near-bottom waters may occur several hundred meters laterally from the cutterhead location (LaSalle et al., 1991).

A hopper dredge consists of one, two, or more dragarms and attached dragheads mounted on a ship-type hull or barge with hoppers to hold material dredged from the bottom (Herbich and Brahme, 1991). As the hopper dredge moves forward, sediments are hydraulically lifted through the dragarm and stored in hopper bins on the dredge (Taylor, 1990; LaSalle et al., 1991). Hopper dredging operations produce turbidity as the dragheads are pulled through bottom sediments. However, the main source of turbidity during hopper dredging operations is sediment release during hopper overflow (Herbich and Brahme, 1991; LaSalle et al., 1991; Herbich, 1992). A plume may occasionally be visible at distances of 1,200 m or more (LaSalle et al., 1991).

Much attention has been given to turbidity effects from dredging, although most reviews have concerned estuaries, embayments, and enclosed waters (e.g., Sherk and Cronin, 1970; Sherk, 1971; Sherk et al., 1975; Moore, 1977; Peddicord and McFarland, 1978; Stern and Stickle, 1978; Herbich and Brahme, 1991; LaSalle et al., 1991; Kerr, 1995; Newcombe and Jensen, 1996; Wilber and Clarke, 2001). Turbidity effects may be less important in unprotected offshore areas for several reasons. Offshore sands tend to be coarser with less clay and silt than inshore areas. The open ocean environment also provides more dynamic physical oceanographic conditions, which minimize settling effects. In addition, offshore organisms are adapted to sediment transport processes, which create scouring, natural turbidity, and sedimentation effects under normal conditions. Impacts should be evaluated in light of natural variability as well as high level disturbances associated with such events as storms, trawling, floods, hypoxia/anoxia, etc. (Sosnowski, 1984; Herbich, 1992). Physical disturbance of the bottom and resulting biological impacts from dredging are similar to those of storms and trawling but at a much smaller spatial scale. The following suggestions from Hughes and Connell (1999) also are instructive regarding the complexities of analyzing effects of multiple stressors (broadly defined as natural or man-made disturbances). Long-term approaches are necessary to understand biological responses to multiple stressors because studying single events in isolation can be misleading. The effects of a particular disturbance often depend critically on impacts from previous perturbations. Consequently, even the same type of recurrent stressor can have different effects at different times, depending on history. Accordingly, when the added dimension of time is considered, the distinction between single and multiple stressors becomes blurred (Hughes and Connell, 1999).

Turbidity from dredging can elicit a variety of benthic responses primarily because attributes of the physical environment are affected (Wilber and Clarke, 2001). Large quantities of bottom material placed in suspension decrease light penetration and change the proportion of wavelengths of light reaching the bottom, leading to decreases in photosynthesis and primary productivity of benthic organisms such as algae, seagrasses, and zooxanthellae (symbiotic algae) associated with corals (Phinney, 1959; Courtenay et al., 1972; Owen, 1977; Onuf, 1994). Light has long been known as an ecological factor affecting dispersal and settlement of marine invertebrate larvae (Thorson, 1964). Suspended materials can prevent growth of benthic organisms such as corals and plants that provide habitat complexity and biological structures used by many other species for shelter and egg attachment (Phinney, 1959; Cronin et al., 1969; Owen, 1977; Nelson, 1989; Connell, 1997). Although coral reefs are adapted to transient increases in turbidity, a continuous reduction in light penetration from dredging may drastically reduce respiration and productivity, cause bleaching and death, and lead to severe alterations of community structure and function, particularly in deep reef zones where light is already limiting (Rogers, 1979). From laboratory experiments that did not reduce light intensity or significantly alter spectral quality, Telesnicki and Goldberg (1995) concluded that turbidity induces increases in respiration rather than decreases in photosynthesis of two common scleractinian coral species from Florida, and suggested that adherence to turbidity-related water quality standards in Florida may result in short-term stress and long-term decline in at least some coral species.

Turbidity can affect food availability for benthic organisms. Changes in light penetration and wavelengths due to turbidity can affect visibility and may be detrimental or beneficial, depending on whether an organism is predator or prey. Suspension and dispersion processes uncover and displace benthic organisms, temporarily providing extra food for bottom feeding species (Centre for Cold Ocean Resources Engineering, 1995).

Turbidity can interfere with food gathering processes of filter feeders and organisms that feed by sight by inundation with nonnutritive particles. In addition to altered feeding rates, other biological responses to turbidity include reduced hatching success, slowed growth, abnormal development, tissue abrasion, and increased mortality (Wilber and Clarke, 2001). In general, egg and larval stages are more sensitive to turbidity effects than older life history stages. Although a considerable amount of information is available on the effects of sediment suspension and dispersion to some benthic organisms, little or no information exists for many other species, particularly those associated with hard bottom (Dodge and Vaisnys, 1977; Bak, 1978; Nelson, 1989; Rogers, 1990; Kerr, 1995; Renaud et al., 1996, 1997).

Suspension and dispersion of sediments may cause changes in sediment and water chemistry as nutrients and other substances are released from the substratum and dissolved during dredging. For aggregate mining operations using hopper dredges, the far-field visible plume contains an organic mixture of fats, lipids, and carbohydrates from organisms entrained and fragmented during the dredging process and discharged with the overflow (Coastline Surveys Limited, 1998; Newell et al., 1999). Dredging may produce localized hypoxia or anoxia in the water column due to oxygen consumption of suspended sediments (LaSalle et al., 1991). Flocculation of suspended sediments can mechanically trap inorganic and organic particles and plankton and carry them to the bottom (Bartsch, 1960 as cited in Levin, 1970).

7.5.1.3 Sediment Deposition

Suspended sediments settle and are deposited nearby or some distance from dredged sites. The extent of deposition and boundaries of biological impact are dependent on the type and amount of suspended sediments and physical oceanographic characteristics of the area.

Deposition of sediments can suffocate and bury hard bottom and soft bottom benthic biota, although some mobile soft bottom organisms are able to migrate vertically to the new surface (Maurer et al., 1986; Nelson, 1988). Unlike most soft bottom biota, many hard bottom organisms are sessile and unable to burrow up through sediment overburden (Nelson, 1989; Wesseling et al., 1999). Heavy sedimentation can result in acute stress and death, and chronic high turbidity can cause stress responses and reductions in health and growth of algae, corals, and other filter feeding organisms (Dodge et al., 1974; Dodge and Vaisnys, 1977; Bak, 1978). Corals and algae with shapes that enable accumulation of sediments are particularly sensitive to depositional effects from mining (Courtenay et al., 1972; Owen, 1977; Bak, 1978; Goldberg, 1985, 1989; Hubbard, 1986; Chansang, 1988; Rogers, 1990; Riegl, 1995). Sediment deposition can negatively affect photosynthetic activity of zooxanthellae and thus the viability of corals (Yentsch et al., 2002; Philipp and Fabricius, 2003). Substantial deposition of sediments in areas of coral growth is of concern, even though many corals can withstand some sedimentation through active removal (Levin, 1970; Courtenay et al., 1972; Rice and Hunter, 1992; Stafford-Smith and Ormond, 1992; Stafford-Smith, 1993; Torres and Morelock, 2002). Corals lose the ability to clean sediments when exposed to extended periods of high turbidity (Clarke et al., 1993). Sediment removal by organisms requires time and energy that otherwise could be used for growth, food capture, reproduction, etc. (Dodge and Vaisnys, 1977; Riegl and Branch, 1995; Dustan, 1999). Growth rates were reduced for some coral species that are efficient sediment rejectors, and colonies of another species lost their symbiotic zooxanthellae and died as a result of sediment cover that they were unable to remove (Bak, 1978). Heavy sedimentation can result in decreased calcification and net productivity of corals, fewer coral species, greater abundance of branching forms, less live coral, lower coral growth rates, reduced coral recruitment, and slower rates of reef accretion (Rogers, 1990). Increased sediment loads can contribute to coral reef degradation and shifts from coral to algal dominance (de Sylva, 1994; Umar et al., 1998; McCook et al., 2001).

Sediment deposition can inhibit larvae of numerous invertebrate species that need hard surfaces to settle and develop (Thorson, 1966; Rogers, 1990). Herrnkind et al. (1988) suggested that large-scale siltation resulting from dredging, mineral mining, and other human activities must be viewed as potentially deleterious to spiny lobster recruitment.

Dredging effects are not necessarily limited to the borrow site alone. Far-field impacts from suspension, dispersion, and deposition of sediments during dredging can be detrimental or beneficial. Johnson and Nelson (1985) found decreases in infaunal abundances and numbers of taxa at nondredged stations, although these decreases were not as extreme as those observed in the borrow site. McCaully et al. (1977; as cited by Johnson and Nelson, 1985) also observed that dredging effects can extend to other nearby areas, and noted decreases in infaunal abundances ranging from 34% to 70% at undredged stations within 100 m of a dredged site. Conversely, benthos may show increased biodiversity downstream from dredged sites (Centre for Cold Ocean Resources Engineering, 1995). In some areas, population density and species composition of benthic invertebrates increased rapidly outside dredged sites, with the level of enhancement decreasing with increasing distance from the dredged site up to a distance of 2 km (Stephenson et al., 1978; Jones and Candy, 1981; Poiner and Kennedy, 1984). The enhancement was ascribed to release of organic nutrients from the dredge plume, a process known from other studies (Ingle, 1952; Biggs, 1968; Sherk, 1972; Oviatt et al., 1982; Coastline Surveys Limited, 1998; Newell et al., 1998, 1999). This suggestion was supported by records of nutrient releases from benthic areas during intermittent, wind-driven bottom resuspension events (Walker and O'Donnell, 1981), significant increases in water column nutrients from simulated storm events in the laboratory (Oviatt et al., 1982), and review of the literature indicating a major restructuring force in infaunal communities is the response of species to resources released from sediments by periodic disturbance (Thistle, 1981). Rosenfeld et al. (1999) also suggested a positive role of turbidity and sedimentation relative to the ability of corals to digest the sediment's organic fraction as a supplementary food source. Fishing may improve temporarily down current of the dredging site and continue for some months (Centre for Cold Ocean Resources Engineering, 1995). Additional far-field impacts can occur by resuspension, redispersion, and redeposition of fine dredged materials by wave and current actions long after dredging has been completed.

7.5.2 Recolonization Periods and Success

7.5.2.1 Adaptations for Recolonization and Succession

In dynamic areas that undergo frequent perturbations, benthic invertebrates tend to be small bodied, short lived, and adapted for maximum rate of population increase with high fecundity, efficient dispersal mechanisms, dense settlement, and rapid growth rates. In contrast, organisms in stable areas tend to be relatively larger and longer lived with low fecundity, poor dispersal mechanisms, slow growth rates, and adaptations for non-reproductive processes such as competition and predator avoidance. Recolonization of a disturbed area often is initiated by organisms that have adaptive characteristics for rapid invasion and colonization of habitats where space is available due to some natural or man-induced disturbance. These early colonizers frequently are replaced during the course of succession through competition by other organisms, unless the habitat is unstable or frequently perturbed (MacArthur, 1960; MacArthur and Wilson, 1967; Odum, 1969; Pianka, 1970; Grassle and Grassle, 1974).

Although the distinction between the adaptive strategies is somewhat arbitrary and is blurred in habitats that are subject to only mild disturbance, the lifestyle differences are fundamentally important because they help explain variations in succession and recolonization periods and success following disturbance (Coastline Surveys Limited, 1998; Newell et al., 1998). Knowledge of faunal component lifestyles allows some predictions of dredging impacts and subsequent recolonization and recovery of community composition (Coastline Surveys Limited, 1998; Newell et al., 1998).

7.5.2.2 Successional Stages

When discussing succession in soft bottom habitats, it is important to point out that most past studies have concerned silt-clay bottoms rather than sand habitats. Little is known about succession in sand bottoms of offshore borrow areas.

Successional theory states that organism-sediment interactions result in a predictable sequence of benthic invertebrates belonging to specific functional types following a major seafloor disturbance (Rhoads and Germano, 1982, 1986). Because functional types are the biological units of interest, the succession definition does not rely on the sequential appearance of particular species or genera (Rhoads and Boyer, 1982). This continuum of change in benthic communities has been divided arbitrarily into three stages (Rhoads et al., 1978; Rhoads and Boyer, 1982; Rhoads and Germano, 1982):

Stage I is the initial pioneering community of tiny, densely populated organisms that appears within days of a natural or anthropogenic disturbance. Stage I communities are composed of opportunistic species that have high tolerance for and can indicate disturbance by physical disruption, organic enrichment, and chemical contamination of sediments. The organisms have high rates of recruitment and ontogenetic growth. Stage I communities tend to physically bind sediments, making them less susceptible to resuspension and transport. For example, Stage I communities often include tube-dwelling polychaetes or oligochaetes that produce mucous to build their tubes, which stabilizes the sediment surface. Stage I communities include suspension or surface deposit-feeding animals that feed at or near the sediment-water interface. The Stage I initial community may reach population densities of 10^4 to 10^6 individuals per m^2;

Stage II is the beginning of the transition to burrowing, head-down deposit feeders that rework the sediment deeper with time and mix oxygen from the overlying water into the sediment. Stage II animals may include tubiculous amphipods, polychaetes, and mollusks. These animals are larger and have very low population densities compared to Stage I animals; and

Stage III is the mature and stable community of deep-dwelling, head-down deposit feeders. In contrast to Stage I organisms, these animals rework the sediments to depths of 3 to 20 cm or more, loosening the sedimentary fabric and increasing the water content of the sediment. They also actively recycle nutrients because of the high exchange rate with the overlying water resulting from their burrowing and feeding activities. Presence of Stage III taxa can be a good indication that the sediment surrounding these organisms has not been severely disturbed recently, resulting in high benthic stability and health. Loss of Stage III species results in loss of sediment stirring and aeration and may be followed by a build-up of organic matter (eutrophication) in the sediment. Because Stage III species tend to have relatively low rates of recruitment and ontogenetic growth, they may not reappear for several years once they are excluded from an area. These inferences are based on past work, primarily in temperate latitudes, showing that Stage III species are relatively intolerant to physical disturbance, organic enrichment, and chemical contamination of sediments. Population densities are low (10 to 10^2 individuals per m^2) compared to Stage I.

The general pattern of succession of benthic species in a marine sediment following cessation of dredging or other environmental disturbance begins with initial recolonization. Initial recolonization occurs relatively rapidly by small opportunistic species that may reach peak population densities within months of a new habitat becoming available after catastrophic mortality of the previous assemblage. As the disturbed area is invaded by additional larger species, the population density of initial colonizers declines. This transitional period and assemblage with higher species diversity and a wide range of functional types may last for years, depending on numerous environmental factors. Provided environmental conditions remain stable, some members of the transitional assemblage are eliminated by competition, and the species assemblage forms a recovered community composed of larger, long-lived, and slow-growing species that have complex biological interactions with one another.

7.5.2.3 Recolonization Periods

The rate of recolonization is dependent on numerous physical and biological factors and their interactions. Physical factors include time of year, dredging technologies and techniques, borrow site dimensions, water currents, water quality, sediment composition, bedload transport, temperature, salinity, natural energy levels in the area, frequency of disturbance, latitude, etc. Recovery times may be shorter in warmer waters at lower latitudes as compared to colder waters at higher latitudes (Coastline Surveys Limited, 1998; Newell et al., 1998). Spatial and temporal variability in physical conditions may in some cases exert more influence on initial stages of recolonization than biological responses of species considered to be opportunists (Zajac and Whitlatch, 1982).

Biological factors influencing the rate of recolonization include the size of the pool of available colonists (Bonsdorff, 1983; Hall, 1994) and life history characteristics of colonizing species (Whitlatch et al., 1998). Recolonization of borrow sites may occur by transport of eggs, larvae, juveniles, and adults from neighboring populations by currents, immigration of motile species from adjacent areas, organisms contained in sediment slumping from the sides of pits, or return of undamaged organisms from the dredge plume. Other biological factors such as competition and predation also determine the rate of recolonization and the composition of resulting benthic communities. Timing of dredging is important because many benthic species have distinct peak periods of reproduction and recruitment. Because larval recruitment and adult migration are the primary recolonization mechanisms, biological recovery from physical impacts generally should be most rapid if dredging is completed before seasonal increases in larval abundance and adult activity (Herbich, 1992). Recovery of a community disturbed after peak recruitment, therefore, will be slower than one disturbed prior to peak recruitment (LaSalle et al., 1991).

Benthic recolonization and succession have been reviewed to varying extents for a wide variety of habitats throughout the world (e.g., Thistle, 1981; Thayer, 1983; Hall, 1994; Coastline Surveys Limited, 1998; Newell et al., 1998). Recolonization is highly variable, depending on the habitat type and other physical and biological factors. Focusing on dredging, Coastline Surveys Limited (1998) and Newell et al. (1998) suggested that, in general, recovery times of 6 to 8 months are characteristic for many estuarine muds, 2 to 3 years for sand and gravel, and 5 to 10 years as the deposits become coarser.

The Centre for Cold Ocean Resources Engineering (1995) estimated times for recovery of a reasonable biodiversity (number of species and number of individuals) based on sediment type. In this study, recovery was defined as attaining a successional community of opportunistic species providing evidence of progression towards a community equivalent to that previously present or at non-impacted sites. Fine-grained sediments may need only 1 year before achieving a recovery level biodiversity, medium-grained deposits 1 to 3 years, and coarse-grained deposits 5 or more years. For a hypothetical borrow site dredging scenario off Ocean City, Maryland, the Centre for Cold Ocean Resources Engineering (1995) stated that virtually all benthic species would be lost, but there may be temporary improvement of fishing due to release of nutrients. Recolonization would start within weeks of closure and moderate biodiversity would occur within 1 year. The borrow site would be colonized initially by a very different species complex than originally present. An estimate of 2 to 3 years was given for the community to begin to show succession to pre-impact sand habitat species.

Recolonization of a borrow site was studied 3 km offshore of Great Egg Harbor Inlet near Ocean City, New Jersey (Scott and Kelley. 1998). Macrobenthic organisms were able to colonize the borrow site rapidly. Approximately 2 years after the last dredging, the number of taxa, diversity, and abundance in the borrow site recovered to conditions that existed in other borrow sites and undisturbed areas before dredging. The community composition within the borrow site may have changed, although the community change was described as not significant and not a result of dredging because the community composition of the borrow site was similar to the composition observed at the adjacent stations. Good juvenile surf clam recruitment occurred in the borrow site, but the population may not have reached size levels in nearby undisturbed sites 2 years after the last dredging. Although biomass and size of surf clams appeared diminished, there was no indication that the population would not stabilize given additional time. As dredging events were conducted in all seasons and no apparent effect was detected, no changes in the timing of dredging appeared to be necessary (Scott and Kelley, 1998).

Recolonization also was studied by Burlas et al. (2001) at borrow areas near Sites H1 and H2. Similar to the present study, their borrow areas were bathymetric high points on the seascape with strong currents and sand movement. Burlas et al. (2001) summarized their results by stating that abundance, biomass, richness, and the average size of the biomass dominant, which was the sand dollar *Echinarachnius parma,* declined immediately after dredging. Abundance, biomass, and richness recovered quickly after the first dredging operation with no detectable difference between dredged and undisturbed areas by the following spring. Abundance also recovered quickly after a second dredging operation, but biomass and richness were still reduced the next spring. Species and biomass composition were altered in similar manners by each operation. Immediately after dredging, the relative contribution of echinoderm biomass declined and the abundance of the spionid polychaete *Spiophanes bombyx* increased. Changes in biomass composition were longer lasting with the assemblage taking 1.5 to 2.5 years to return to undredged conditions.

Studies of recolonization listed and discussed by Grober (1992) and the National Research Council (1995) indicate that recolonization of offshore borrow sites is highly variable. This variability is not surprising considering differences among studies in geographic locations, oceanographic conditions, sampling methods and times, etc. Part of the problem in determining recolonization patterns is seasonal and year to year fluctuations in benthic community characteristics and composition. Without adequate seasonal and yearly data prior to dredging, it is difficult to determine whether differences in community characteristics and composition are due to temporal changes or dredging disturbance.

Results and conclusions from these offshore borrow site studies indicate that recolonization usually begins soon after dredging ends. Recolonization periods range in duration from a few months (Saloman et al., 1982; Jutte et al., 2002) for shallow dredging to possibly decades for deep pits (Barry A. Vittor & Associates, Inc., 1999). Although abundance and diversity of benthic infauna within borrow sites often returned to levels comparable to pre-dredging or reference conditions within less than 1 year, several studies documented changes in benthic species composition that lasted much longer, particularly where sediment composition was altered (e.g., Saloman, 1974; Wright, 1977; Johnson and Nelson, 1985; Bowen and Marsh, 1988; Van Dolah et al., 1992, 1993; Wilber and Stern, 1992; Barry A. Vittor & Associates, Inc., 1999).

Most recolonization studies of borrow sites concentrated on three main features of infaunal communities: number of individuals (population density), number of species (diversity), and weight (biomass as an index of growth). Dredging is usually accompanied by an immediate and significant decrease in the number of individuals, species, and biomass of benthic infauna. Using biological community parameters (e.g., total taxa, total number of individuals, species diversity, evenness, richness, etc.), some previous studies tend to indicate that recovery of borrow sites occurs in approximately 1 year after dredging. However, these parameters do not necessarily reflect the complex changes in community structure and composition that occur during the recovery process. Major changes in species assemblages and community composition usually occur shortly after dredging such that a different type of community exists. Although the number of individuals, species, and biomass of benthic infauna may approach pre-dredging levels within a relatively short time after dredging, recovery of community composition may take longer.

7.5.2.4 Recolonization Success and Recovery

Assessing dredging impacts and borrow site recolonization and recovery is difficult because most biological communities are complex associations of species that often undergo major changes in population densities and community composition, even in areas that are far removed and unaffected by dredging and other disturbances. Recolonization success and recovery do not necessarily mean that communities should be expected to return to the pre-dredged species composition. To gauge recovery, it is important to compare the community composition of dredged sites with control areas during the same seasons because community composition changes with time.

When long-term alterations in sediment structure and composition occur as a result of dredging, long-term differences in the composition of benthic assemblages inhabiting those sites may occur as well. The recovery time of benthic assemblages after dredging can depend in large measure on the degree and duration of sediment alteration from sand borrowing (Van Dolah, 1996). Recolonization success and recovery also are controlled by compaction and stabilization processes involving complex interactions between particle size, water currents, waves, and biological activities of the benthos following sediment deposition (Oakwood Environmental Ltd., 1999). While the abundance and diversity of infaunal assemblages may recover relatively rapidly in dredged sites, it may take years to recover in terms of sediment and species composition.

One conclusion commonly held is that perturbations to infaunal communities in borrow sites are negligible because organisms recolonize rapidly (Wilber and Stern, 1992). This conclusion often is based on measures including densities, species diversity/evenness indices, relative distribution of classes or phyla, and species-level dendrograms. For example, many researchers have recognized that borrow site and reference area infaunal communities can differ considerably at the species level, although these differences usually are considered insignificant because species diversity is high. According to Wilber and Stern (1992), reliance on these studies may lead to a premature conclusion that impacts to borrow site infauna are minimal because these measures are relatively superficial and ambiguous characteristics of infaunal communities. Wilber and Stern (1992) reexamined infaunal data from four borrow site projects by grouping species into functional groups called ecological guilds based on similarities in feeding mode, locomotory ability, and sediment depth occurrence. Their analyses showed that infaunal communities in borrow and control areas can differ in several ways and that these differences can last several years. Polychaetes and amphipods that recolonize borrow sites are small-bodied and confine their movement and feeding to the surface sediment or the interface between the sediment and water column. In contrast, control areas have well-developed infaunal communities commonly consisting of large-bodied organisms that move and feed deep in the sediment (Wilber and Stern, 1992). They concluded that infaunal communities recolonizing borrow sites may remain in an early successional stage for 2 to 3 years or longer as opposed to being completely recovered in shorter time frames.

The conclusions of Wilber and Stern (1992) coincide with the model of succession discussed previously. The model states pioneering or opportunistic species are the first to colonize an area after a physical disturbance to the bottom (e.g., dredging borrow sites). Pioneering species tend to share several ecological traits, including a tendency to confine activities to the sediment-water interface, possibly because subsurface conditions cannot support a significant number of organisms. The subsurface environment changes with time after the disturbance, possibly by actions of early colonizers, and becomes suitable for deposit feeders and mid-depth burrowers. The relative absence of deposit feeders and mid-depth burrowers is interpreted to mean an area is still in the state of recovery.

Although most literature on recolonization periods and success in borrow sites concerns infauna, some information exists for soft bottom epifauna. Numbers of taxa and individuals collected by trawls in a borrow site off Duval County, Florida greatly exceeded control area numbers 4 months after dredging and were generally higher 7 and 13 months after dredging (Applied Biology, Inc., 1979). There were no detectable differences between pre-dredging and post-dredging (8 and 16 months) epifaunal communities in a borrow site surveyed by otter trawl and video camera off Egmont Key, Florida (Blake et al., 1995).

In general, hard bottom species take longer to recolonize their respective habitats than soft bottom species. This is particularly true for large reef-building corals living at the extreme northern end of their distributional range (Courtenay et al., 1972). When a reef community is destroyed, ecological conditions that follow cannot be expected to coincide with those that initially developed the community, and it cannot be assumed that the reef community will replace itself (Johannes, 1970 as cited by Levin, 1970). Connell (1997) cautioned about judging recovery of coral assemblages, in a similar way that Wilber and Stern (1992) did for soft bottom infaunal assemblages: recovery in coral abundance does not necessarily imply that the assemblage has recovered in several other characteristics, such as species composition, diversity, rates of reproduction and growth, colony size structure, etc. Recovery in abundance is only one aspect of recovery of a coral assemblage (Connell, 1997). Brown and Howard (1985) stated that generalizations concerning recolonization and recovery of reef corals are dangerous, and recommended consideration of each case individually. Because hard bottom species tend to be slow growing and direct mechanical damages to hard bottom habitats from dredging have occurred in the past (Courtenay et al., 1972, 1974; Britt & Associates, Inc., 1979; Marszalek, 1981; Blair and Flynn, 1988; Goldberg, 1989; Blair et al., 1990), surveys should be conducted in the future prior to dredging in and near specific borrow sites to determine if hard bottom is present and protective measures are necessary.

7.5.3 Predictions Relative to the Sand Resource Areas

7.5.3.1 Potential Soft Bottom Benthic Effects

Sediment Removal

The immediate impact of excavating upper sediments of a sand resource area will be removal of portions of the benthic invertebrate populations that inhabit shelf sediments, especially those fauna with sessile and slow-moving lifestyles. Surveys within and adjacent to the sand resource areas, as well as benthic investigations of nearby waters, reveal that benthic invertebrate assemblages of open shelf waters of the study area include crustaceans, echinoderms, mollusks, and polychaetous annelids.

The expected loss of benthic fauna due to sediment excavation could be considered to represent a negligible impact on the ecosystem when evaluating the impact on a broad spatial scale. Specific shoals within each sand resource area are targeted for excavation based on particular sedimentary and bathymetric characteristics. A significant extent of non-dredged areas will surround borrow sites. These undisturbed areas would be a primary source of colonizing fauna for the excavated site (Van Dolah et al., 1984; Jutte et al., 2002) and would complement colonization of altered substrata via larval recruitment. The great densities and fecundity of invertebrate populations, along with the relatively small areas of impact proposed, would preclude significant long-term negative effects on benthic populations. Impacts are expected to be localized and short-term.

Correlation between sediment composition and the composition of infaunal assemblages has been demonstrated in numerous environmental surveys, including this study. Invertebrate populations inhabiting marine soft bottoms in the study area exhibit heterogeneous distributions that largely are the result of the local sedimentary regime. Modification of surficial sediments and local bathymetry could result in an alteration of the areal extent and relative distribution of infaunal assemblages by altering the distribution of sediment types capable of supporting those assemblages.

It is possible that a change in surficial sediment composition within excavated areas could become a long-term result of dredging. Several factors could contribute to such an outcome, primarily the type of sediments exposed by dredging, the degree of deposition of fine sediments into dredged areas, and bathymetric alteration that results in hypoxic or anoxic conditions. These factors would depend primarily on the depth of excavation, which would be determined by the vertical relief of the sand shoal to be excavated, the vertical extent of those sediments suitable for coastal renourishment projects, and the volume of sand required.

Because the inner shelf ecosystem of the east Florida shelf exhibits some heterogeneity in sediment types and their associated infaunal assemblages, those assemblages that initially colonize dredged areas likely would be similar to some naturally occurring assemblages that inhabit nearby non-dredged areas, especially areas with finer sediments. When viewed within a context of scale, removal of sediments from portions of the inner shelf would at most minimally alter the existing spatial balance of habitat (sediment) types. Moreover, those habitats that have relatively high amounts of finer sediments are not uninhabitable, or necessarily less functional in an ecological sense, when compared to sand or gravel substrata. Various sedimentary habitats merely differ in their level of suitability for certain types of infaunal taxa. Changes in habitat suitability that result from sand removal likely would be ephemeral and inconsequential in the shelf ecosystem, a system where both infaunal assemblage types and sedimentary parameters often are temporally and spatially variable.

Motile populations, including non-migratory foragers, would be less stressed by sediment removal than infauna or sessile epibiota. Most epibiotal and demersal fish populations would have a low probability of being adversely impacted directly by the dredging of surficial sediments. Slow-moving or burrowing sessile epibiota inhabiting the study area include echinoderm and decapod taxa, and local populations of these types of benthic organisms would most likely experience a reduction in density due to sediment removal. Motile epifauna generally are migratory and are not restricted to the borrow areas. Most demersal populations exhibit naturally dynamic distributions and are distributed over a wide geographic area. However, there have been questions regarding the importance of shoal areas as orientation sites, staging areas, or aggregating sites for pelagic and demersal fishes (Research Planning, Inc. et al., 2001). Unfortunately, scientific data are lacking.

Most impacts of sediment removal on epibenthic and demersal fish taxa would be indirect in nature, through habitat alteration. A reduction of infaunal biomass resulting from sediment removal could have an indirect effect on the distribution of certain demersal fishes and other epibenthic predators by interrupting established energy pathways to the higher trophic levels represented by these foraging taxa. Reductions in densities of the preferred prey of bottom-feeding taxa could induce migration of foragers to unimpacted areas. However, a relatively small percentage of infaunal prey items that typically are consumed by demersal taxa would be rendered unavailable for consumption as a result of prey removal along with sediments. Benthic predators would select alternative areas in which to forage. Because excavated areas are expected to recover relatively rapidly after dredging, loss of infaunal biomass due to sediment excavation is unlikely to adversely affect normal energy flow in dredged areas.

In addition to widely documented spatial variation, the location and extent of some inner shelf-inhabiting infaunal and demersal populations vary seasonally in the study area. Seasonal variability should be considered when evaluating potential impacts due to sand removal. The timing of sand removal would seem to be less critical for minimizing the impact on infauna than for other faunal categories of concern (e.g., key pelagic species) due to the great abundance and reproductive potential of infaunal populations. Many numerically dominant infaunal taxa inhabiting the study area are known to exhibit either year-round or late winter-early spring periods of recruitment. Because of these patterns of recruitment and lower winter densities, removal of sand between late fall and early spring would result in less stress on benthic populations.

Sediment Suspension/Dispersion

Whether cutterhead suction dredging or hopper dredging ultimately is utilized for sand mining, the amount of sediment suspension that results from these excavation methods is not anticipated to be of a scale that would cause significant negative impacts to the benthic community. Central east Florida sand resource areas are characterized by a relatively limited amount of very fine sediments, indicating that the region encompassing those areas currently is not a depositional environment, but is hydrologically dynamic. In general, benthic assemblages of the inner central east Florida shelf probably are adapted to periodic reworking of surficial sediments caused by tropical and extra-tropical storms. Impacts of dredging-induced elevations in turbidity (associated mainly with hopper dredging) would be short-term and localized. Motile taxa could avoid turbid areas and are unlikely to be affected by sediment resuspension.

Sediment Deposition

Of the various faunal categories, infaunal and sessile epibiotal populations would be most negatively affected by significant deposition of sediments; however, efficient methods of sediment excavation would preclude all but a relatively minor amount of sediment deposition. Suspension and transport of sediments away from dredging sites should be minimal, and any subsequent deposition will be insignificant in degree. In the unlikely event that significant dredging-related deposition of fine-grained sediments were to occur, the deposited sediments likely would not persist at sites of initial redeposition because of the high-energy inner shelf environment. However, some low or depressional areas of the seafloor could receive substantial deposition of fine sediments under this scenario. Given the relatively small amount of sediment suspension anticipated to occur during dredging, the degree of burial should be substantially less than would be required to impact negatively on infaunal populations.

Potential Recolonization Periods and Success

Germano (1999) has suggested that, despite all advances in theoretical ecology over the last half century and huge amounts of data that have been collected in various marine monitoring programs, we still do not know enough about how marine ecosystems function to be able to make valid predictions of impacts before they occur. The relative lack of understanding of complex ecological systems may in some cases even preclude our ability to observe significant negative environmental effects of activities of concern. However, review of previous studies does provide some evidence as to how certain activities, such as dredging, might affect benthic communities.

The period and nature of post-dredging recovery of benthic assemblages within an excavated borrow site will depend primarily on the depth of sand excavation. While surface area of impact could be minimized by excavating a shoal to a greater depth, deep excavation likely would require a greater length of time for complete recovery of infaunal assemblages within the impacted area. Creation of a bathymetrically abrupt pit has potential to inhibit water current flow through such a feature, possibly resulting in a "dead zone" characterized by deposition of fine particles and hypoxia or anoxia. This scenario would extend the duration of ecological impact beyond that which would occur with a shallower cut over a much larger area.

Results of long-term environmental monitoring of a borrow site located 3.6 km offshore Coney Island, New York have demonstrated potential consequences of dredging an abrupt pit feature (Barry A. Vittor & Associates, Inc., 1999). A nearby reference area also was sampled before (1992) and after (1995 through 1998) dredging. Prior to dredging, average water depths were approximately 3 to 4 m at the Coney Island borrow site and in the reference area. After the last dredging in 1995, and up to the last monitoring event (1998), depths of borrow site stations varied from 6 to 15 m, while the average depth of reference area stations did not change during the study period. Prior to dredging, sediments at the borrow site were 55% medium to coarse sands, but by 1995 were fine to medium sands (<20% medium to coarse sand). By 1998, the silt/clay fraction (>20%) of borrow site sediments was significantly greater than in reference area sediments (4%). During each year following the last dredging event, infaunal assemblage composition at the borrow site was numerically dominated by deposit-feeding polychaetes (*Spio setosa* and *Streblospio benedicti*) and mollusks (primarily *Tellina agilis*); none of these species were ever observed in the non-dredged reference area. Although hypoxic conditions have not been detected at the Coney Island borrow site, bathymetric alteration and subsequent deposition of fine sediments resulted in persistent alteration of its infaunal assemblage.

While the initial impact on benthic assemblages would increase with a greater surface area of sand removal, the persistence of ecological impact that would occur with a relatively shallow excavation would be less than that of a deep pit. Central east Florida sand resource areas exhibit natural inter-ridge trough features. These bathymetric depressions can be depositional areas for fine sediments and often support benthic assemblages that are different from nearby assemblages inhabiting gravel and sand (Camp et al., 1977; Lyons, 1989). Ultimately, though, it is expected that only the leading edge of each shoal will be dredged and that depth of dredging will not substantially exceed the level of the ambient shelf surface.

The length of time required for reestablishment of predredging infaunal assemblages within excavated sites depends in part on the length of time required for refilling of those mined areas. Shallow waters of the central east Florida inner shelf are strongly influenced by factors such as tidal currents, circulation, and storms. These same forces would tend to modify dredged sites toward predredging morphology. The rate of reestablishment of natural benthic conditions at dredged sites may depend especially on the extent of storm-induced sediment transport, which can be substantial at the relatively shallow depths of the sand resource areas. The length of time required to reestablish infaunal assemblages also depends in large measure on the sediments exposed by dredging. Shoals tend to consist of well-sorted sands and be vertically uniform in sediment composition. Sediments exposed by dredging probably would not differ substantially from existing surficial sediments.

Assuming that the depth of sand excavation would not be so great as to substantially alter local hydrological characteristics, removal of benthic organisms along with sediments should quickly be followed by initial recolonization of dredged areas by opportunistic infaunal taxa. Early-stage succession tends to begin within days of sediment removal through settlement of larval recruits, primarily annelids and bivalves (Grassle and Grassle, 1974;

Simon and Dauer, 1977). Initial larval recruits likely would be dominated by populations of deposit feeding, opportunistic taxa, such as those collected from muddy sediment stations offshore central east Florida. These taxa may include polychaetes such as *Magelona* sp. H, *Mediomastus*, and *Paraprionospio pinnata*, and bivalves such as *Lucina* and *Tellina*. These species are well adapted to environmental stress and exploit suitable habitat when it becomes available. Later successional stages of benthic recolonization will be more gradual and involve taxa that generally are less opportunistic and longer-lived. Immigration of motile annelids, crustaceans, and echinoderms into impacted areas also will begin soon after excavation. Dredging of only a small portion of each sand resource area will ensure that a supply of non-transitional, motile taxa will be available for rapid migration into dredged areas.

Because sediment shoals in the central east Florida sand resource areas tend to be vertically uniform in terms of sediment composition, recolonization of exposed sediments by later successional stages likely will proceed even if dredged shoals are not completely reestablished, particularly if the depth of dredging does not cut below ambient grade. While community composition may differ for a period of time after the last dredging, the infaunal assemblage type that exists in mined areas will be similar to naturally occurring assemblages in the study area, particularly those assemblages inhabiting inter-ridge troughs. Johnson and Nelson (1985) documented changes in benthos following excavation of a nearshore borrow site close to Fort Pierce Inlet. They found that relatively large reductions in abundance, but not number of species, occurred in the borrow site after dredging and that both parameters approximated predredging levels in from 9 to 12 months after the last dredging. Based on previous observations of infaunal reestablishment in dredged areas, the infaunal community in central east Florida offshore borrow sites most likely will become reestablished within 2 years, and will exhibit levels of infaunal abundance, diversity, and composition comparable to nearby nondredged areas.

7.5.3.2 Potential Hard Bottom Benthic Effects

Sediment Removal

Equipment used in the sediment removal process (e.g., cutterheads, dragheads, cables, anchors, pipelines, etc.) can physically damage hard bottom areas occurring close to sand borrow sites. For the sand resource areas studied in this report, hard bottom was documented in Areas B1, D1, D2, and near C2. Epibiotal assemblages observed on the hard bottom in and around Area D2 supported large sponges and octocorals that could be easily sheared off by mishandled anchor cables. Similar damage could occur to lower profile organisms (algae, small sponges, hydrozoans) characteristic of hard bottom assemblages in or near the other sand resource areas. Anchors and cutterheads could damage hard bottom structures that provide substrate for epibiota and fishes. Such impacts can be avoided by conducting hard bottom surveys prior to dredging. If hard bottom is found, detailed maps can be used in conjunction with precise positioning of all mechanical components that could potentially impact the seafloor.

Sediment Suspension/Dispersion

Suspended sediment affects sessile epibiota by interrupting photosynthesis in algae and organisms with symbiotic algae (e.g., some octocorals and scleractinian corals). High suspended sediment also can affect respiration causing metabolic stress in hard corals and other epibiota. Effects of turbidity generated in local areas will depend on background turbidity levels and therefore levels normally experienced by organisms composing local hard bottom assemblages. Because of the narrowness of the continental shelf and proximity of the Gulf Stream current, water clarity is consistently high in the southern portion of the study area. Epibiota on hard bottom outcrops in and around Areas D1 and D2 are expected to be least adapted to turbid water when compared to epibiota in and around the other sand resource areas where background turbidity is generally higher. Turbidity excess generated during dredging projects should be of short duration (acute) and restricted to small areas relative to the regional continental shelf. Chronic resuspension could occur in areas where deposits of fine sediments are exposed to waves and currents by dredging projects.

Sediment Deposition

High levels of sedimentation can impact hard bottom organisms by burying them, thereby preventing photosynthesis by algae, seagrasses, and coral zooxanthellae; clogging filter-feeding organisms such as sponges; or causing octocorals and scleractinian corals to bleach or expend large amounts of energy producing mucous to clear sediment from their surfaces. High sedimentation also can reduce recruitment of hard bottom organisms by covering potential substrate and burying newly settled juveniles. Sedimentation effects on hard bottom habitats in and around the sand resource areas will depend on sediment composition; distance from the dredging site to hard bottom areas; and prevailing tides, currents, wind, and local weather conditions. Dredge-related sediment deposition should be confined to areas close to the actual excavation points, thus avoiding hard bottom areas through pre-project surveys.

Potential Recolonization Periods and Success

Physical damage to hard bottom areas may occur through accidental contact of dredging equipment. Recolonization of a damaged hard bottom area within or near any of the sand resource areas off central east Florida would depend on the spatial extent of the damage, timing with respect to larval availability, latitudinal location of the damaged area, and composition of the impacted epibiotal assemblage.

Hard bottom assemblages in the vicinity of the sand resource areas are composed of algae, sponges, octocorals, hydrozoans, scleractinian corals, and other sessile organisms. Most members of these groups colonize disturbed or newly open hard bottom areas by settlement of planktonic larvae. Thus, the assembly of organisms on newly exposed hard bottom areas can be highly variable and depend on life history characteristics of individual species coupled with local circulation patterns. Because of spatial and temporal variation in these biotic and physical factors, colonization of impacted hard bottom may not follow the orderly successional process described in Section 7.5.2.2 for infauna. Timing relative to larval availability (spawning times) is a key aspect of hard bottom colonization that sets the starting point of species assembly. Recovery of hard bottom assemblages consisting of large sponges, octocorals, and scleractinian corals can take years or decades (Fizhardinge and Bailey-Brock, 1989). Recovery of scleractinian corals on damaged coral reefs takes years and in some cases decades and depends on the type of disturbance, coral species, ecological setting, and other factors (Connell, 1997).

If an area composed of large sponges and octocorals near the southern sand resource areas (particularly Area D2) was damaged, recovery would likely take years and possibly decades. A similar sized area covered mostly by algae and encrusting sponges such as those observed in sand resource areas north of Area D2 would likely recover more rapidly because algae and encrusting sponges grow rapidly and are adapted to conditions where space is limited and sediment movement is dynamic (Renaud et al., 1997).

7.6 PELAGIC ENVIRONMENT

7.6.1 Fishes

Potential impact producing factors from dredging operations in the sand resource areas that may affect pelagic fishes offshore of central east Florida include physical injury, attraction, and turbidity. These factors along with potential impacts are described in following subsections. Project scheduling considerations and essential fish habitat also are discussed in separate subsections.

7.6.1.1 Physical Injury

Physical injury through entrainment of adult fishes by hydraulic dredging has been reported for several projects (Larson and Moehl, 1988; McGraw and Armstrong, 1988; Reine et al., 1998). The most comprehensive study of fish entrainment took place in Grays Harbor, Washington during a 10-year period when 27 fish taxa were entrained (McGraw and Armstrong, 1988). Most entrained fishes were demersal species such as flatfishes, sand lance, and sculpin; however, three pelagic species (anchovy, herring, and smelt) were recorded. Entrainment rates for the pelagic species were very low, ranging from 1 to 18 fishes/1,000 cy (McGraw and Armstrong, 1988). Comparisons between relative numbers of entrained fishes with numbers captured by trawling showed that some pelagic species were avoiding the dredge. Another entrainment study conducted near the mouth of the Columbia River, Washington reported 14 fish taxa entrained at an average rate of 0.008 to 0.341 fishes/cy (Larson and Moehl, 1988). Few of the coastal pelagic fishes occurring offshore of Florida should become entrained because the dredge's suction field exists near the bottom and many pelagic species have sufficient mobility to avoid the suction field.

7.6.1.2 Attraction

Even though dredges are temporary structures, they can still attract roving pelagic fishes. This attraction would be similar to an artificial reef effect, where both small and large coastal pelagic fishes become associated with fixed structures. This may temporarily disrupt migratory routes for some members of the stock, but it is unlikely that there would be an appreciable negative effect.

7.6.1.3 Turbidity

Turbidity can cause feeding impairment, avoidance and attraction movements, and physiological changes in adult pelagic fishes. As discussed for larval fishes, pelagic species are primarily visual feeders, and when turbidity reduces light penetration, the fishes reactive distance decreases (Vinyard and O' Brien, 1976). Light scattering caused by suspended sediment also can affect a visual predator's ability to perceive and capture prey (Benfield and Minello, 1996). Some fishes have demonstrated the ability to capture prey at various turbidity levels, but the density of prey and light penetration are important factors (Grecay and Targett, 1996).

Some species will actively avoid or be attracted to turbid water. Experiments with pelagic kawakawa (*Euthynnus affinis*) and yellowfin tuna (*Thunnus albacares*) demonstrated that these species would actively avoid experimental turbidity clouds, but also would swim directly through them during some trials (Barry, 1978). Turbidity plumes emanating from coastal rivers may retard or affect movements of some pelagic species.

Gill cavities can be abraded and clogged by suspended sediment, preventing normal respiration and mechanically affecting food gathering in planktivorous species (Bruton, 1985). High suspended sediment levels generated by storms have contributed to the death of nearshore and offshore fishes by clogging gill cavities and eroding gill lamellae (Robins, 1957). High concentrations of fine sediments can coat respiratory surfaces of the gills, preventing gas exchange (Wilber and Clarke, 2001).

Understanding and predicting effects of suspended sediments on fishes requires some information on the range and variation of turbidity levels found at a project site prior to dredging (Wilber and Clarke, 2001). The spatial and temporal extents of turbidity plumes from either cutterhead or hopper dredges are expected to be limited. Therefore, there should be negligible impact on adult pelagic fishes. However, removal of coarse sediment from borrow sites could promote chronic turbidity if finer underlying sedimentary layers are exposed and resuspended.

7.6.1.4 Underwater Noise

Noise associated with all aspects of the dredging process may affect organisms in several ways. Some reef fish larvae have been shown to respond to sound stimuli as a sensory que to settlement sites (Stobutzki and Bellwood, 1998; Tolimieri et al., 2000). Alterations of background noise could impair the ability of newly settled fishes to locate preferred substrate. Changes in noise levels also may affect feeding or reproductive activities of reef fishes that depend on sound for these activities (Myrberg and Fuiman, 2002). Continental Shelf Associates, Inc. (2004) reviewed effects of noise on fishes. This report stated that all fish species investigated can hear, with varying degrees of sensitivity, within the frequency range of sound produced by cutterhead dredges, hopper dredges, and clamshell excavators. These sounds can mask the sounds normally used by fishes in their normal acoustic behaviors at levels as low as 60 to 80 dB (just above detection thresholds for many species). Levels as high as 160 dB may cause receiving fish to change their behaviors and movements that may temporarily affect the usual distribution of animals and commercial fishing. Continuous, long-term exposure to levels above 180 dB has been shown to cause damage to the hair cells of the ears of some fishes under some circumstances. These effects may not be permanent because damaged hair cells are repaired and/or regenerated in fishes. None of the dredge types proposed for this project produce continuous sounds above 120 dB (Richardson et al., 1995). Due to the short duration of most dredging projects, the effects of underwater noise on fish populations should be minimal.

7.6.1.5 Project Scheduling Considerations

It is uncertain whether hydraulic dredging will present a significant problem for pelagic fishes offshore of central east Florida or not. Temporal scheduling of environmental windows as means to avoid impacts is practical if the organisms in question are highly concentrated in an area during some specific time period. The only current window was established to protect nesting sea turtles. This window allows beach nourishment projects to operate from November to March, a time period when Spanish mackerel migrate into the study area to overwinter. If a project was conducted in the vicinity of an important gathering area for Spanish mackerel, there could be some temporary impact. Unfortunately, quantitative data are lacking to support the use of an environmental window to lessen effects on pelagic fishes.

7.6.1.6 Essential Fish Habitat

The Magnuson-Stevens Fishery Conservation and Management Act (16 United States Code [U.S.C.] § 1801-1882) established regional Fishery Management Councils and mandated that Fishery Management Plans (FMPs) be developed to responsibly manage exploited fish and invertebrate species in Federal waters of the U.S. When Congress reauthorized this act in 1996 as the Sustainable Fisheries Act, several reforms were made. One change was to charge the NMFS with designating and conserving Essential Fish Habitat (EFH) for species managed under existing FMPs. This is intended to minimize, to the extent practicable, any adverse effects on habitat caused by fishing or non-fishing activities, and to identify other actions to encourage the conservation and enhancement of such habitat.

EFH is defined as “those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity” (16 U.S.C. § 1801[10]). The EFH interim final rule summarizing EFH regulations (62 Federal Register 66531-66559) outlines additional interpretation of the EFH definition. Waters, as defined previously, include “aquatic areas and their associated physical, chemical, and biological properties that are used by fish, and may include aquatic areas historically used by fish where appropriate.” Substrate includes “sediment, hard bottom, structures underlying the waters, and associated biological communities.” Necessary is defined as “the habitat required to support a sustainable fishery and the managed species’ contribution to a healthy ecosystem.” “Fish” includes “finfish, mollusks, crustaceans, and all other forms of marine animal and plant life other than marine mammals and birds,” whereas “spawning, breeding, feeding or growth to maturity” cover the complete life cycle of those species of interest.

The SAFMC has produced several FMPs for single and mixed groups of species. All of these FMPs were recently amended in a single document (SAFMC, 1998a) to address EFH for shrimps; spiny lobster; golden crab; corals, coral reefs, and hard/live bottom; red drum; snapper-grouper management unit; and coastal pelagic fishes. In addition to the FMPs prepared by the SAFMC, highly migratory species are managed by the Highly Migratory Species Management Unit, Office of Sustainable Fisheries, NMFS. One FMP was recently prepared for highly migratory species that includes descriptions of EFH for sharks, tunas, and swordfish (NMFS, 1999a); a second FMP for Atlantic billfishes was amended to include EFH designations (NMFS, 1999b). Two additional highly migratory species, dolphin and wahoo, will soon be formally managed by the SAFMC, and an FMP is in progress. A separate FMP describing EFH for pelagic *Sargassum* in the South Atlantic was prepared in late 1998 (SAFMC, 2002).

Within the EFH designated for various species, particular areas termed Habitat Areas of Particular Concern (HAPCs) also are identified. HAPCs either play important roles in the life history (e.g., spawning areas) of Federally managed fish species or are especially vulnerable to degradation from fishing or other human activities. In many cases, HAPCs represent areas where detailed information is available on the structure and function within the larger EFH. Descriptions of EFH and HAPCs follow for the aforementioned FMPs and key managed species present in the study area. Some of these species also are “aquatic resources of national importance” under Section 906(e)(1) of the Water Resources

Development Act of 1986, and Part IV, Section 3(a) of the current Memorandum of Agreement between the Department of Commerce and USACE.

Penaeid and Rock Shrimps

EFH for penaeid shrimps includes inshore nursery areas such as tidal freshwater, estuarine, and marine wetlands (Table 7-1). Offshore sedimentary habitats where spawning and growth to maturity take place are important as EFH.

Table 7-1. Managed invertebrate and reef fish species for which Essential Fish Habitat has been identified off central east Florida (From: South Atlantic Fishery Management Council, 1998b). Organisms are listed in phylogenetic order.

Species	**Life Stages (Reproductive Activity)**	**Habitat**
Invertebrates		
Rock shrimp (*Syconia* spp.)	Adults; juveniles; larvae	Soft bottom (18 to 180 m); pelagic
Pink shrimp (*Farfantepenaeus duorarum*)	Adults; juveniles; larvae	Soft bottom, seagrass areas; pelagic
Spiny lobster (*Panulirus argus*)	Adults; juveniles; larvae	Hard bottom; seagrass areas, mangrove areas, sponges, macroalgae; pelagic
Golden crab (*Chaceon fenneri*)	Adults; larvae	Soft bottom (>200 m); pelagic
Reef Fishes		
Red grouper (*Epinephelus morio*)	Adults; juveniles; larvae; eggs	Hard bottom; seagrass areas; pelagic
Snowy grouper (*Epinephelus niveatus*)	Adults; juveniles; larvae; eggs	Hard bottom; pelagic
Black grouper (*Mycteroperca bonaci*)	Adults; juveniles; larvae; eggs	Hard bottom; seagrass areas; pelagic
Gag (*Mycteroperca microlepis*)	Adults; juveniles; larvae; eggs	Hard bottom; seagrass areas; pelagic
Scamp (*Mycteroperca phenax*)	Adults; juveniles; larvae; eggs	Hard bottom; pelagic
Mutton snapper (*Lutjanus analis*)	Adults; juveniles; larvae; eggs	Hard bottom; seagrass areas; pelagic
Gray snapper (*Lutjanus griseus*)	Adults; juveniles; larvae; eggs	Hard bottom; seagrass areas; mangrove areas; pelagic
Red snapper (*Lutjanus campechanus*)	Adults; juveniles; larvae; eggs	Hard and soft bottom shelf waters; pelagic
Lane snapper (*Lutjanus synagris*)	Adults; juveniles; larvae; eggs	Hard bottom; seagrass areas; pelagic
Vermilion snapper (*Rhomboplites aurorubens*)	Adults; juveniles; larvae; eggs	Hard bottom; pelagic
Yellowtail snapper (*Ocyurus chrysurus*)	Adults; juveniles; larvae; eggs	Hard bottom; seagrass areas; pelagic
Tilefish (*Lopholatilus chamaeleonticeps*)	Adults; juveniles; larvae; eggs	Soft bottom; pelagic
Greater amberjack (*Seriola dumerili*)	Adults; juveniles; larvae; eggs	Hard bottom; *Sargassum*; pelagic
Almaco jack (*Seriola rivoliana*)	Adults; juveniles; larvae; eggs	Hard bottom; *Sargassum*; pelagic
Gray triggerfish (*Balistes capriscus*)	Adults; juveniles; larvae; eggs	Hard bottom; *Sargassum*; pelagic

Rock shrimp EFH is composed of offshore terrigeneous and biogenic sedimentary bottoms in water depths ranging from 18 to 182 m deep, with maximum occurrence and abundance of organisms between 34 and 55 m (Table 7-1). EFH includes the water current transport system near Cape Canaveral, Florida, which is important in the retention and inshore transport of larval rock shrimp. The Gulf Stream also is considered an important larval transport mechanism (SAFMC, 1998b).

Areas considered to be HAPCs for penaeid shrimps include all coastal inlets, all State-designated nursery habitats, and State-identified overwintering areas.

Because rock shrimps are found generally in waters deeper than the sand resource areas, impacts to EFH will be minimal. The EFH for penaeid shrimps could be affected by dredging projects; entrainment and turbidity may be factors in the vicinity of Areas A1 and A2. However, due to the small areal coverage of these sand resource areas, effects are expected to be minimal.

Spiny Lobster

Spiny lobster EFH consists of hard bottom, coral reefs, crevices, cracks, and other structured bottom in shelf waters (Table 7-1). Juvenile habitat is in nearshore waters and ranges in type from massive sponges, mangrove roots, and seagrass meadows to soft bottom with macroalgal clumps. The Gulf Stream provides an important mode of transport for early life history stages of the spiny lobster (SAFMC, 1998b).

All HAPCs for spiny lobster are located south of the sand resource areas and include the Dry Tortugas, Florida Keys, and hard bottom from Fowey Rocks near Miami to Jupiter Inlet.

Spiny lobster EFH exists in all hard bottom areas throughout the study area. Measures should be taken to protect hard bottom areas and avoid dredging impacts to spiny lobster.

Golden Crab

Table 7-1 indicates the EFH for golden crab in the central east Florida region. Golden crab EFH includes a variety of bottom types, including foraminiferan ooze, distinct mounds of dead corals, ripple bottom, dunes, black pebbles, low outcrop, and soft bioturbated bottom (SAFMC, 1998b). All of these habitats are in water depths exceeding 200 m. The Gulf Stream is considered to be important in dispersal of planktonic eggs and larvae.

There is not enough information available on the ecology of golden crab from which to identify HAPCs.

Golden crab EFH occurs in water depths much greater than the depths of the sand resource areas, and therefore no impacts are expected.

Corals, Coral Reefs, and Hard/Live Bottom

EFH for reef building stony corals is outside of the study area and extends from Palm Beach County, Florida south through the Florida reef tract bordering the Florida Keys. This area extends from nearshore (0 to 4 m) to 30 m water depths where salinity is consistently above 30 ppt and water temperatures range from 15ºC to 35ºC. Corals, coral reefs, and hard/live bottom habitats were not included in the EFH tables.

EFH for *Antipatharia* (black corals) includes hard, exposed, rough, stable substrate throughout the management area in high salinity (30 to 35 ppt) offshore waters and depths exceeding 18 m not restricted by light penetration.

EFH for octocorals, except the order Pennatulacea (sea pansies and sea pens), includes hard, exposed, rough, stable substrate throughout the management area in subtidal to outer shelf depths within a wide range of salinity and light penetration.

EFH for Pennatulacea (sea pansies and sea pens) includes muddy, silty bottoms in subtidal to outer shelf depths within a wide range of salinity and light penetration.

HAPCs for corals, coral reefs, and hard/live bottom habitats of central east Florida include 1) *Phragmatopoma* worm reefs in nearshore waters; 2) nearshore hard bottom in water depths of 0 to 4 m; 3) offshore hard bottom in water depths of 5 to 30 m; and 4) *Oculina* banks from Fort Pierce to Cape Canaveral in water depths >30 m. Only the third category occurs in the study area.

Measures should be taken to avoid hard bottom and associated dredging effects to corals, coral reefs, and hard/live bottom. Dredging operations causing mechanical damage or producing high turbidity and sedimentation could significantly affect corals attached to hard bottom areas within the study area. Hard bottom was found in Areas B1, D1, and D2 and near C2 during field surveys. Mechanical damage to hard bottom in all sand resource areas should be avoided. Areas D1 and D2 would be most susceptible to elevated turbidity and sediment deposition because organisms in these areas are less adapted to these stressors. See Section 7.5.1 for discussion of impacts to corals.

Red Drum

EFH for red drum includes artificial reefs, estuarine emergent vegetated wetlands (flooded brackish marsh, mangrove fringe, flooded salt marshes, and tidal creeks), high salinity coastal areas, oyster reefs, submerged rooted aquatic vegetation (seagrasses), tidal freshwater, and unconsolidated bottom (Table 7-2). These habitats occur from Virginia to the Florida Keys (SAFMC, 1998b).

HAPCs for red drum are all State-designated nursery habitats of particular importance to red drum, coastal inlets, documented sites of spawning aggregations, and habitats for submerged aquatic vegetation (SAFMC, 1998b).

EFH for red drum exists mostly in inshore waters well isolated from the sand resource areas. For this reason, effects to red drum EFH are expected to be minimal.

Table 7-2. Managed species (red drum and coastal pelagic fishes) for which Essential Fish Habitat has been identified off central east Florida (From: South Atlantic Fishery Management Council, 1998b). Fishes are listed in phylogenetic order.

Species	Life Stages (Reproductive Activity)	Habitat
Red Drum		
Red drum (*Sciaenops ocellatus*)	Adults; larvae and eggs (spawning area)	Soft bottom; seagrass areas; oyster reefs; mangrove areas; wetlands; pelagic
Coastal Pelagic Fishes		
Cobia (*Rachycentron canadum*)	Adults; juveniles/subadults; larvae; eggs	Pelagic; hard bottom areas
Dolphin (*Coryphaena hippurus*)	Adults; juveniles/subadults; larvae and eggs (spawning area)	Pelagic; *Sargassum* mats
King mackerel (*Scomberomorus cavalla*)	Adults; juveniles/subadults; larvae and eggs (spawning area)	Pelagic; hard bottom areas
Spanish mackerel (*Scomberomorus maculatus*)	Adults; juveniles/subadults; larvae; eggs	Pelagic; hard bottom areas
Little tunny (*Euthynnus alletteratus*)	Adults; juveniles/subadults; larvae and eggs (spawning area)	Pelagic; hard bottom areas

Snapper-Grouper Management Unit

The snapper-grouper management unit is composed of 73 species from 10 families. Only the most important species of snappers, groupers, jacks, tilefishes, and triggerfishes are listed in Table 7-1. Families not listed in Table 7-1 are grunts, porgies, spadefishes, temperate basses, and wrasses. EFH for adults of this species group consists of hard bottom features such as artificial reefs, coral reefs, live bottom, and rocky outcrops (SAFMC, 1998b).

These features extend from nearshore out to at least 200 m water depths. Juveniles of many species utilize either hard bottom features or inshore habitats, including artificial structures (i.e., dock and bridge pilings), mangrove roots, oyster reefs, and seagrass meadows. Eggs and larvae of reef fishes are pelagic and reside in the upper water column for the first 20 to 50 days of life.

HAPCs described for the snapper-grouper management unit include high relief offshore areas where spawning occurs, localities of known spawning aggregations, and nearshore hard bottom areas. The SAFMC has proposed HAPCs in the study area including “The Pines” area off Sebastian (near Areas B1 and B2) and the “Hobe Sound Bar” off Hobe Sound (near Areas C2 and D1).

Snapper-grouper EFH exists on all hard bottom areas throughout the study area; therefore, effects of attraction, entrainment, and turbidity are possible. Measures should be taken to protect hard bottom areas and avoid dredging effects to snapper-grouper.

Coastal Pelagic Fishes

All members of the coastal pelagic management unit occur in central east Florida waters. Species most important to regional fisheries are cobia, dolphin, king and Spanish mackerels, and little tunny. Coastal pelagic species are migratory water column dwellers; however, most species have some affinity for manmade or natural structures. Hard bottom features, sandy bottoms, and shoal areas occurring from the surf zone to the shelf break encompass EFH for coastal pelagic fishes. Coastal inlets, high-salinity bays, and *Sargassum* rafts also are important for various life stages of coastal pelagic fishes. A species account of EFH for these species in central east Florida is given in Table 7-2.

EFH for coastal pelagic fishes could be affected by turbidity that could alter migratory routes or temporarily disrupt feeding activity in shelf or nearshore waters. Coastal pelagic species such as cobia, jacks, king and Spanish mackerels, round scad, and Spanish sardine could be attracted to a dredge and its attendant structures. Although these effects could occur, the small spatial and temporal scales of individual projects make these effects negligible.

Highly Migratory Species

Many highly migratory species are caught in the fisheries of central east Florida because of the proximity of the Gulf Stream to shore. Table 7-3 lists the billfishes, dolphin, sharks, swordfish, tunas, and wahoo with EFH in the central east Florida study area. For many of these fishes, species-specific information is limited. Blue and white marlins occur off central east Florida. Several shark species also frequent Gulf Stream, shelf, and in the case of the bull shark, estuarine waters of the region. *Sargassum* is important habitat for various life stages of swordfish and tunas. Swordfish and bluefin tuna migrate through the Florida Straits and into the eastern Gulf of Mexico to spawn (NMFS, 1999a). From an analysis of oceanic longline catch records, Worm et al. (2003) found the oceanic waters off east Florida to be “diversity hotspots” for highly migratory species.

HAPCs have not been designated by NMFS (1999ab) for members of the highly migratory species groups.

As with coastal pelagic fishes, highly migratory species could be affected by turbidity generated during a dredging project. Turbidity plumes could alter normal migratory and feeding patterns, but these effects would be of short duration. Some highly migratory species could be attracted to a dredge or related structures. These effects would be most important in the southern portion of the study area where the Gulf Stream current flows closer to shore.

Table 7-3. Managed highly migratory species for which Essential Fish Habitat has been identified off central east Florida (National Marine Fisheries Service, 1999a, b). Fishes are listed in phylogenetic order.

Species	Life Stages (Reproductive Activity)	Habitat
Sharks		
Nurse shark (*Ginglymostoma cirratum*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic; hard bottom areas
Longfin mako shark (*Isurus paucus*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Oceanic whitetip shark (*Carcharhinus longimanus*)	Late juvenile/subadult	
Spinner shark (*Carcharhinus brevipinna*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Silky shark (*Carcharhinus falciformis*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Bull shark (*Carcharhinus leucas*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic; bays and estuaries
Night shark (*Carcharhinus signatus*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Dusky shark (*Carcharhinus obscurus*)	Neonates/early juveniles	Pelagic
Caribbean reef shark (*Carcharhinus perezi*)	Adult; late juveniles/subadults	Pelagic
Sandbar shark (*Carcharhinus plumbeus*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Tiger shark (*Galeocerdo cuvier*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Lemon shark (*Negaprion brevirostris*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Scalloped hammerhead (*Sphyrna lewini*)	Adults; late juvenile/subadults	Pelagic
Great hammerhead (*Sphyrna mokarran*)	Adults; late juvenile/subadults	Pelagic
Bonnethead (*Sphyrna tiburo*)	Adults; late juvenile/subadult; neonates/early juveniles	Pelagic
Tunas and Mackerels		
Wahoo (*Acanthocybium solanderi**)	Adults; juveniles and subadults; larvae and eggs (spawning area)	Pelagic
Skipjack tuna (*Katsuwonus pelamis*)	Adults; larvae and eggs (spawning area)	Pelagic; *Sargassum*
Yellowfin tuna (*Thunnus albacares*)	Adults; juveniles/subadults; larvae and eggs (spawning area)	Pelagic; *Sargassum*
Bluefin tuna (*Thunnus thynnus*)	Adults; larvae and eggs (spawning area)	Pelagic; *Sargassum*
Swordfish		
Swordfish (*Xiphias gladius*)	Adults; larvae and eggs (spawning area)	Pelagic
Billfishes		
Blue marlin (*Makaira nigricans*)	Adults; juveniles and subadults; larvae and eggs	Pelagic
White marlin (*Tetrapterus albidus*)	Adults; juveniles and subadults	Pelagic
Longbill spearfish (*Tetrapterus pfluegeri*)	Adults	Pelagic
Atlantic sailfish (*Istiophorus platypterus*)	Adults; juveniles and subadults; larvae and eggs (spawning area)	Pelagic

* Fishery Management Plan in progress.

Sargassum

Sargassum floats at the sea surface, often forming large mats. These accumulations attract numerous small fishes and invertebrates that become mobile epipelagic assemblages. Larger fishes, particularly billfishes, dolphin, tunas, and wahoo, associate with *Sargassum* mats in search of prey and possibly shelter (SAFMC, 2002). EFH for *Sargassum* is simply the shelf waters and Gulf Stream.

The Gulf Stream is considered an HAPC for drifting *Sargassum*.

Sargassum EFH encompasses much of the study area, particularly the south portion where the Gulf Stream is closest to shore. Effects on the drifting *Sargassum* assemblage are expected to be minimal.

7.6.2 Sea Turtles

Potential impact producing factors from dredging operations in the sand resource areas that may affect sea turtles offshore of central east Florida include physical injury, habitat loss or modification, turbidity, hypoxia/anoxia, and underwater noise. These factors along with potential impacts are described in following subsections. Project scheduling considerations also are discussed.

7.6.2.1 Physical Injury

The main potential effect of dredging on sea turtles is physical injury or death caused by entrainment. Numerous sea turtle injuries and mortalities have been documented during dredging projects along Florida's east coast (Studt, 1987; Dickerson et al., 1992; Slay, 1995; NMFS, 1996, 1997). Physical impact can occur when a turtle feeding or resting on the seafloor is contacted by the dredge head. Two types of dredges may be used. Cutterhead suction dredges are considered unlikely to kill or injure turtles, perhaps because the cutterhead encounters a smaller area of seafloor per unit time, allowing more opportunity for turtles to escape (Palermo, 1990). Hopper dredges are believed to pose the greatest risk to sea turtles (Dickerson, 1990; NMFS, 1997). There has been considerable research into designing modified hopper dredges with turtle deflectors that reduce the likelihood of entraining sea turtles (Studt, 1987; Berry, 1990; Dickerson et al., 1992; Banks and Alexander, 1994; USACE, 1999b).

Of the five turtle species that may occur off Florida, three (loggerhead, Kemp's ridley, and green) are considered to be at risk from dredging activities because of their benthic feeding habits (Dickerson et al., 1992). Chelonid sea turtles (i.e., those other than leatherbacks) feed primarily in depths of 15 m or less (NMFS, 1996). The risk of physical impacts to turtles would appear to be greatest in the shallowest depths of the sand resource areas. However, there also is risk in deeper water because when turtles feed there, they tend to stay on the bottom longer (NMFS, 1996).

Loggerheads are the most abundant turtles in the study area and historically have been the species most frequently entrained during hopper dredging, possibly accounting for up to 86% of the total (Reine and Clarke, 1998). Kemp's ridley and green turtles historically have accounted for much smaller portions of the total. Leatherbacks, which also occur off Florida, are unlikely to be affected by dredging because they feed in the water column rather than on the bottom (NMFS, 1996). Hawksbills are unlikely to be affected because they are the least common turtles in the study area and tend to occur in the vicinity of hard bottom habitats. If a hopper dredge is used during the loggerhead turtle nesting season of April through September, the NMFS may require turtle monitoring and use of a turtle-deflecting draghead.

Based on the opinion of the NMFS (1996), the level of "take" (defined in this case as death or injury) resulting from physical injuries from the dredging operations is not likely to jeopardize the continued existence of any sea turtle species along Florida's east coast.

7.6.2.2 Habitat Loss or Modification

Juvenile and subadult loggerhead, Kemp's ridley, and green turtles use central east Florida inner shelf waters as developmental habitat, foraging on benthic organisms primarily on inner-shelf hard bottom habitats. Therefore, when borrow sites have significant concentrations of benthic resources, dredging can reduce food availability both by removing potential food items and altering the benthic habitat (NMFS, 1996). These effects would be temporary, as benthic populations within these soft bottom habitats would be expected to recover over a period of months to years, depending on the grain size and stability of subsurface sediments exposed after dredging (see Section 7.5.3). In addition, borrow sites represent only a small portion of this type of shallow benthic habitat available off east Florida.

7.6.2.3 Turbidity

Sea turtles in and near the study area may encounter turbid water that could temporarily interfere with feeding. However, due to the limited areal extent and transient occurrence of the sediment plume, turbidity is considered unlikely to significantly affect turtle behavior or survival.

7.6.2.4 Noise

Dredging is one of many human activities in the marine environment that produce underwater noise. Sea turtles have limited hearing ability (Ridgway et al., 1969; Lenhardt, 1994; Bartol et al., 1999), and its role in their life cycle and behavior is poorly known. It is believed that sea turtles do not rely on sound to any significant degree for communication or food location, although it has been suggested that low frequency sound may be involved in natal beach homing behavior (Dodd, 1988). The latter could be a consideration during the nesting season.

There are indications that underwater noise is unlikely to significantly affect sea turtles. First, studies in the Gulf of Mexico have shown some evidence for positive association of sea turtles with petroleum platforms (Rosman et al., 1987; Lohoefener et al., 1990) despite the industrial noise associated with these structures. Second, experiments testing the use of seismic airguns to repel turtles from dredging activities indicate that even loud noises cause avoidance only at very close range (e.g., 100 m or less) (Moein et al., 1994; Zawila, 1994). If noise does have any impact on turtles, it would most likely be positive by encouraging avoidance of the dredge.

7.6.2.5 Project Scheduling Considerations

Project scheduling, such as the implementation of environmental windows, is one way to avoid or reduce sea turtle impacts during dredging operations (Studt, 1987; Arnold, 1992; Dickerson et al., 1998; Reine et al., 1998; NRC, 2001). If a hopper dredge is used, then it would be best to avoid the loggerhead nesting season, which has been reported as April through September (Ryder et al., 1994). This same period would generally have higher risk of encountering juvenile and subadult green, Kemp's ridley, hawksbill, and leatherback turtles. If use of a hopper dredge during this season cannot be avoided, then other mitigation and monitoring requirements are likely to be imposed, such as turtle monitoring (requiring onboard observers), use of a turtle-deflecting draghead (NMFS, 1996), or relocation trawling. If a cutterhead suction dredge is used, seasonal or other restrictions are considered unnecessary because this procedure is considered not likely to adversely affect sea turtles by the NMFS (B. Hoffman, 2002, pers. comm., NMFS, St. Petersburg, FL).

7.6.3 Marine Mammals

Potential impact producing factors from dredging operations in the sand resource areas that may affect marine mammals offshore of central east Florida include physical injury, turbidity, and noise. These factors along with potential impacts are described in following subsections. Project scheduling considerations also are discussed.

7.6.3.1 Physical Injury

Marine mammals are unlikely to be physically injured by dredging *per se* because they generally do not rest on the bottom, and most can avoid contact with dredging vessels and equipment. The odontocete (toothed) marine mammals most likely to be found in inner shelf waters off central east Florida, such as bottlenose dolphin and Atlantic spotted dolphin, are agile swimmers that are presumed capable of avoiding physical injury during dredging.

However, physical injury from vessel strikes is a serious concern for two endangered species of mysticete (baleen) whales: the North Atlantic right whale and humpback whale. Recovery plans for these species identify vessel strikes as a contributing factor impeding their recovery (NMFS, 1991a,b; Reeves et al., 1998). Vessel strikes are an especially serious concern for North Atlantic right whales. NMFS published regulations in February 1997 restricting vessel approaches of North Atlantic right whales. These regulations prohibit all approaches within 460 m of any North Atlantic right whale, whether by boat, aircraft, or other means (NMFS, 1998). Manatees are uncommon to rare within offshore waters of the inner shelf. However, they are extremely vulnerable to vessel strikes within inshore waters from transiting vessels. Measures to minimize the potential for vessel strikes of endangered whales and manatees could be part of any Biological Opinion issued by the NMFS and USFWS for dredging off east Florida.

7.6.3.2 Turbidity

Marine mammals in and near the study area may encounter turbid water during dredging. This turbidity could temporarily interfere with feeding or other activities, but the animals could easily swim to avoid turbid areas. Due to the limited areal extent and transient occurrence of the sediment plume, turbidity is considered unlikely to significantly affect marine mammal behavior or survival.

7.6.3.3 Noise

Dredging can be a significant source of continuous underwater noise in nearshore areas, particularly in low frequencies (<1,000 Hz) (Richardson et al., 1995). This noise typically diminishes to background levels within about 20 to 25 km of the source (Richardson et al., 1995). Noise levels are not sufficient to cause hearing loss or other auditory damage to marine mammals (Richardson et al., 1995). However, some observations in the vicinity of dredging operations and other industrial activities have documented avoidance behavior, while in other cases, animals seem to develop a tolerance for the industrial noise (Malme et al., 1983; Richardson et al., 1995). Due to the frequency range of their hearing, mysticete (baleen) whales and especially manatees are more likely to be affected by low frequency noise than are odontocete marine mammals (Richardson et al., 1995; Gerstein et al., 1999). The main concern would be that dredging noise could cause avoidance of the dredging area during humpback whale and (especially) North Atlantic right whale migrations. It is presumed that any manatees in offshore waters near the dredging operation would avoid the source of noise.

7.6.3.4 Project Scheduling Considerations

Northern right whales occur as seasonal (winter and early spring) residents. Humpback whales could occur as occasional transients (strays), primarily during winter. Generally, the probability of encountering these species in the study area would be lowest during summer. The months of December through March would be least favorable because North Atlantic right whales typically reside in waters of the study area, particularly the northern part (Kraus et al., 1993; Slay et al., 1998). Whether or not environmental windows (seasonal restrictions on dredging) are implemented, measures to minimize possible vessel interactions with endangered whales are likely to be required by the NMFS. Common shelf species such as bottlenose dolphin and Atlantic spotted dolphin may be present year-round and, as noted above, are unlikely to be adversely affected by dredging.

7.7 POTENTIAL CUMULATIVE EFFECTS

Cumulative impacts resulting from multiple sand mining operations within a sand resource area are a concern when evaluating potential long-term effects on benthic and pelagic assemblages. The most likely mechanism that could result in adverse cumulative effects is the extraction of sand from the same shoal site more than once, resulting in a relatively deep pit feature where development of natural benthic assemblages is impeded. For the purpose of this analysis, it is assumed that a different area of the targeted sand shoal, or a different shoal, would be dredged each replenishment interval.

Cumulative physical environmental impacts from multiple sand extraction scenarios at one or all sand borrow sites within the study area were evaluated to assess long-term effects at potential borrow sites and along the coastline. Results presented above for wave and sediment transport processes reflect the impact of large extraction scenarios from one or multiple offshore sites that are expected to be within the cumulative sand resource needs of the State for the next 10 years. It was determined that no significant changes to longshore sediment transport will result from the modeled borrow site configurations for Areas A, B, and D. However, the proposed sites in Area C do have significant impacts to transport potential along the shoreline. Therefore, Area C sites should be redesigned so impacts are within acceptable limits, most likely by reducing the maximum depth of excavation at the sites.

Given that the expected beach replenishment interval is on the order of 5 to 10 years, and that the expected recovery time of the affected benthic community after sand removal is anticipated to be much less than that (within 2 years), the potential for significant cumulative benthic impacts is remote. No cumulative impacts to the pelagic environment, including zooplankton, squids, fishes, sea turtles, and marine mammals, are expected from multiple sand mining operations within a sand resource area.