SAFMC Fishery Ecosystem Plan II: Shallow Coral and Coral Reef Habitat

Shallow Coral and Coral Reef Habitat

Description and distribution

Shallow water coral reefs and coral communities exist within the southern geographical areas under Council authority. In this document these habitats are defined as occurring in depths generally less than 50 meters. Depending upon many variables, stony corals may dominate a habitat, be a significant component, or be individual colonies within a community characterized by other fauna (e.g., sponges or macroalgae). In some areas stony corals have grown in such profusion that their old skeletons accumulate and form reef structure (e.g., coral reefs). In other areas, corals grow as a less dominant component of benthic communities on geologically derived hard substrates (e.g., coral communities). This section focuses on those ecosystems under Council authority having Scleractinians as an important member of the community. Hardbottom communities that have little or no Scleractinians are treated in the Live/Hardbottom Habitat section of this document (Section XXXX).

Section 1: Reef biogeography, habitat, and community types:

North Florida to North Carolina

Coral assemblages from north Florida north to North Carolina, are dominated by ahermatypic stony coral species and gorgonians, although some hermatypic species do occur off North Carolina (MacIntyre and Pilkey 1969) and Georgia (Hunt 1974). The very limited coral assemblages within this area are found on shallow-water hardbottom habitats ((Johnston 1976); off Georgia and South Carolina (Stetson et al. 1962; Porter 1978 personal communication; Thomas 1978 personal communication); and North Carolina (Huntsman 1984; MacIntyre and Pilkey 1969)) and deep-water banks (*Oculina* spp.). These are further described in Section XXX of this document.

North Florida to St. Lucie Inlet

From St. John's Inlet to St. Lucie Inlet coral assemblages are relatively sparse and low in diversity as compared to reefs further south. Coral colonies are commonly located on non-coral-derived consolidated carbonate sediments (Avent et al. 1977). Corals are most common in the nearshore hardbottom and along two reef tracts (20 m, 30 m). The two major reef tracts consist of ledges of up to 3 m relief; while the outer 30 m shelf tract runs through the majority of this region, the 20 m shelf tract runs intermittently. Coral assemblages include octocorals (*Lophogorgia, Leptogorgia, Eunicea, Antillorgia* spp.), and scleractinian coral (*Oculina diffusa*, *Oculina varicosa*, and *Siderastrea* spp.). Both temperate and subtropical fish and invertebrate species are represented in this region. At the shelf-edge, high relief (up to 25 m) pinnacles begin at 50 m depth where *Oculina varicosa* form massive branching colonies (Reed 1980). For a more extensive review of the deep water *Oculina* reefs refer to Section XXX.

Comment [1]: Reed J.K. 1980. Distribution and structure of deep-water Oculina varicosa coral reefs off central eastern Florida. Bulletin of Marine Science 30(3): 667-677.

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SE Florida

The Florida Reef Tract (FRT) extends approximately 577 km from the St. Lucie Inlet (Martin County), southward to the Dry Tortugas banks. Off the mainland coast of southeast Florida, the northern extension of the FRT extends from Martin County approximately 170 km south into Miami-Dade County. From central Palm Beach County south to, in particular offshore Broward County, southern Miami-Dade County the reef system is described as a series of linear (Inner, Middle, and Outer) reef complexes (referred to as reefs, reef tracts, or reef terraces). These complexes run parallel to shore, generally at depths approximately 6m to 20m. In addition there are extensive nearshore ridges and colonized pavement areas nearer to shore (Moyer et al. 2003; Banks et al. 2007; Walker et al. 2008). Although these high latitude habitats are near the environmental threshold for significant coral reef growth, they are colonized by an extensive coral reef community which is quite similar within the linear reefs. This region has a similar diversity of key functional groups (stony corals, octocorals, sponges, and macroalgae) to that of the southern regions of the FRT (the Florida Keys and Dry Tortugas) but contributions of these groups to benthic cover may vary (Ruzicka et al. 2010; Ruzicka et al. 2012, Gilliam et al. 2015).

The nearshore ridges and colonized pavement areas occur within a km of shore in water depths generally less than 5 m and are most prominent off Palm Beach, Broward, and Miami-Dade counties. This habitat is defined as flat, low relief, solid carbonate rock with variable sand cover within the most nearshore areas (Walker et al. 2008). In Palm Beach and Martin Counties, the sessile community in less than 3 m is dominated primarily by turf and macroalgae. The dominant scleractinian at these depths are *Siderastrea* species (CSA 2009). In a number of these shallow water areas, the sabellariid polychaete *Phragmatopoma lapidosa* (know as worm rock) can be a dominant component of the habitat. South of these counties, these habitats have been documented to contain areas with the highest stony coral cover and the greatest abundance of larger (>2m) stony corals (dominated by *Montastrea cavernosa* and *Orbicella faveolata)* in the region (Gilliam et al. 2015; Gilliam et al. 2015, Walker...). In this area, this habitat also contains perhaps the most abundant population of staghorn coral, *Acropora cervicornis,* in the Council management area (Vargas-Angel et. al 2006, Walker et al. 2012, Gilliam et al. 2015; Gilliam et al. 2015, D'Antonio et al. 2016).

The Inner Reef occurs within 1 km of shore and crests in 3 to 7 m depths. The Middle Reef crests in 12 to 14 m depths, and Outer Reef crests in 15 to 21 m depths. A large sand area generally separates the Inner and Middle, and the Middle and Outer, reef complexes. The Inner and Middle Reefs extend from northern Broward County south into Miami-Dade County. The Outer Reef occurs within 3 km of shore and is the most continuous reef complex extending from central Palm Beach County south into Miami-Dade County. The community in these reefs includes over 30 species of stony corals and a diverse assemblage of gorgonians and sponges (Gilliam et al. 2015). The common stony coral species include: *Montastrea cavernosa*, *Siderastrea siderea*, *Porites astreoides*, *Solenastrea bournoni*, *Meandrina meandrites*, and *Dichocoenia stokesii*. Octocorals (gorgonians) and sponges generally have a greater density than stony corals. Some of the common octocoral genera include: *Eunicea*, *Antillogorgia*, *Muricea*, *Plexaurella*, *Pterogorgia* and *Icilogorgia* (Goldberg 1973). Very large (>1m wide) barrel sponges, *Xestospongia muta*, are conspicuous and quite abundant in certain areas of the Middle and Outer Reefs.

Florida Keys

The southernmost component of the Florida Reef Tract includes the area south of Soldier Key to the Dry Tortugas banks. Along the nearshore environs to the deep fore reef adjacent to the straits of Florida, coral-associated habitats consist of nearshore hardbottom communities, patch reefs, and a semi-continuous series of offshore bank-barrier reefs (reef flats, spur and groove) (summarized in Marszalek et al. 1977, Jaap 1984, and Chiappone 1996). These habitats boast a wide bathymetric distribution, from the intertidal to great depths, and are currently colonized by calcifying algae (e.g., *Halimeda*), sponges, octocorals, and a few species of stony corals. Local environmental conditions, driven by water exchange between Florida Bay and the Atlantic Ocean, dictate which species colonize the substrate.

Low relief hardbottom communities occur within 2 km of shore on the Florida Bay and Atlantic sides of the islands. These communities are highly diverse (as described in Chiappone and Sullivan 1994) and dominate the Florida Keys in terms of areal extent (Chiappone 1996).

The patch reef habitat is constructed by a few species of massive stony corals; most often the principal species is *Orbicella annularis*, boulder star coral. Other common foundation building species include *Colpophyllia natans* and *Siderastrea siderea*. Common octocoral genera found on patch reefs include: *Antillogorgia*, *Pseudoplexaura*, *Gorgonia*, *Muricea* and *Plexaurella*. Patch reefs are concentrated in the area off Elliott Key (Biscayne National Park), north Key Largo (John Pennekamp Coral Reef State Park, Florida Keys National Marine Sanctuary, FKNMS), and in the Hawk Channel area from Marathon to Key West (FKNMS).

The outer bank reefs are the seaward-most reefs in the Florida Keys coastal ecosystem. These reefs are most commonly visited by the diving and snorkeling charters. Their principal, unique feature is the spur and groove system (Shinn 1963). The system is a series of ridges and channels facilitating water transport from seaward to inshore. The coral most responsible for building the spurs was *Acropora palmata* (Shinn 1963), whose population has since experienced significant decline. The spur and groove systems occur in depths that range from a few centimeters to 10 meters. Beyond 10 meters, the spur and groove formation may or may not continue seaward as very low relief structures. Often, this habitat subunit is referred to as the fore-reef and may continue to about 30 m depth. Seaward, sediment beds separate the fore-reef from deeper reef formations in 40 m depth. Stony coral cover has significantly declined over time in this system

Comment [2]: Goldberg, W.M. 1973. The ecology of the coral-octocoral communities off the southeast Florida coast: geomorphology, species composition, and zonation. Bulletin of Marine Science 23(3): 465-488.

Comment [3]: Marszalek DS, Babashoff G, Noel MR, Worley DR (1977) Reef distribution in south Florida. Proc 3rd Int Coral Reef Symp 2: 223-229

Comment [4]: Jaap WC (1984) The Ecology of South Florida Reefs: A Community Profile. U.S. Fish and Wildlife Service, Office of Biological Sciences. Washington D.C. FWS/OBS-82/08. 138 pp.

Comment [5]: Chiappone M (1996). Marine Benthic Communities of the Florida Keys. In: Site Characterization for the Florida Keys National Marine Sanctuary and Environs, Vol. 4. The Preserver.

Comment [6]: Chiappone M, Sullivan KM (1994) Ecological structure and dynamics on nearshore hard-bottom communities of the Florida Keys. Bulletin of Marine Science 54(3): 747-756

Comment [7]: Shinn, EA (1963). Spur and groove formation on the Florida Reef Tract. Journal of Sedimentary Petrology. 33(2): 291-303.

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at both shallow and offshore fore-reefs, and a transition to octocoral dominance is most evident at shallow fore-reefs (Ruzicka et al. 2014). Octocorals of the genus *Antillogorgia*, *Gorgonia*, *Pseudoplexaura, Muricea*, *Eunicea* and *Plexaurella* are commonly found in these outer bank reefs.

Comment [8]: R. R. Ruzicka, M. A. Colella, J. W. Porter, J. M. Morrison, J. A. Kidney, V. Brinkhuis, K.S. Lunz, K. A. Macaulay, L. A. Bartlett, M. K. Meyers, J. Colee. (2014). Temporal changes in benthic assemblages on Florida Keys reefs 11 years after the 1997/1998 El Niño. Marine Ecology Progress Series. 489: 124-141.

The Tortugas Banks are a variation of the deeper reefs found in Dry Tortugas National Park. The depths are greater than 20 m and extend to 40 m. The foundation is Pleistocene karst limestone. The extensive banks host a major grouper and snapper fishery, including a critical 46 square mile spawning ground currently protected as a Research Natural Area. The banks have abundant coral of a few species. Black coral (Order Antipatharia) are common on the outer edge of the bank.

Section 2 - Ecological Functions

Coral reefs and hardbottom have many functional roles within the SAFMC jurisdiction. These functions include complex issues such as trophic relationships, shelter, and cross-shelf and large-scale population connectivity via reproduction. High diversity of reef residents support complex trophic relationships and novel routes of productivity, including significant bio-calcification which provides the architectural structure. The details of these relationships and functions have been examined in several recent large compilations such as Mora (2015) and Riegl and Dodge (2012) and Birkeland (2015).

Comment [9]: Birkeland, C (ed) 2015. Coral Reefs in the Anthropocene. 271 pp. Springer.

New emerging trends?

- Changing environment, including expected tropicalization resulting in shifting behavior and species interactions

Section 3 - Use

- Track down values for commercial and recreational fisheries, diving and tourism, etc. (Brett Roger)

Healthy coral reefs are among the most biologically diverse and economically valuable ecosystems on earth, providing valuable and vital economic goods and ecosystem services. Coral ecosystems are a source of food for millions; protect coastlines from storms and erosion; provide habitat, spawning, and nursery grounds for economically and recreationally important fish species; provide jobs and income to local economies from fishing, recreation, and tourism; are a source of new medicines, and are hotspots of marine biodiversity.

Section 4 - Current Habitat Management

FEDERAL

<u>Essential Fish Habitat</u>

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The 1996 federal reauthorization of the Magnuson Stevens Act (the Sustainable Fisheries Act) mandated that all eight federal fishery management councils identify Essential Fish Habitat (EFH) in their jurisdiction and amend all Fishery Management Plans (FMPs) as applicable. The SAFMC followed the enabling language and treated EFH as “those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity”. The SAFMC also identified EFH - Habitat Areas of Particular Concern (HAPCs), which are EFH areas that include one of these four attributes: provide important ecological functions; are sensitive to environmental degradation; include a habitat type that is/will be stressed by development; or include a habitat type that is rare (SAFMC, 1998a).

EFH applies to each life stage of managed species and different life stages of the same species often use different habitats. All coral and hardbottom habitats are designated as EFH-HAPC for the 60 reef species currently in the Snapper Grouper FMP as well as the Spiny Lobster. Additionally, other components of reef habitat such as sponges are EFH for Spiny Lobster. The habitat source document for these designations (SAFMC, 1998b) provided much rationale and content used also in first FEP document. Many administrative details on how EFH is used in coral conservation permitting among federal, state, and local agencies are reviewed in Lindeman and Ruppert (2011).

Place-based management:
The South Atlantic region includes a range of federally managed areas with coral reef habitats, most notably the Florida Keys National Marine Sanctuary (NOAA) and two units of the National Park Service (Dry Tortugas and Biscayne National Parks). Each of these areas has its own management plan, including some areas set aside as marine reserves.

The Florida Keys National Marine Sanctuary (FKNMS) was designated in 1990 for protection in response to concerns about the decline of the reef ecosystem in the area (FKNMS Protection Act 1990). Today, the FKNMS protects more than 9,946 km2 (2,900 nautical mi2) of Florida Keys coastal and ocean waters. With the designation, several protective measures were immediately put into place, such as prohibiting oil exploration, mining, or any type of activity that would alter the seafloor, and restricting large shipping traffic. Anchoring on, touching, and collecting coral are all restricted within sanctuary waters. The FKNMS is jointly administered by the State of Florida and the National Oceanic and Atmospheric Administration (NOAA). The FKNMS management plan was first established in 1998 and implemented a network of zones and protected areas as well as strategies including mooring buoys and a water quality protection program. Additional Ecological Reserves were implemented in the the Dry Tortugas region in 2001. NOAA is currently undertaking the first comprehensive review of the management plan, zoning plan and regulations. This review is a public process that will eventually culminate in an updated management plan and potential modifications to regulations, marine zones, and the sanctuary boundary.

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Two components of the National Park system manage coral reef habitats in the south Atlantic region, Biscayne National Park and Dry Tortugas National Park. Biscayne National Park was designated in 1980 (after prior status as Biscayne National Monument) and protects habitats adjacent to the south Florida urban area including Biscayne Bay, the barrier islands, and out to the reef tract. Biscayne NP recently released its first general management plan (June 2015) which includes a marine reserve zone incorporating both fore-reef and patch reef habitats. Dry Tortugas, administered under the management of Everglades National Park, was designated in 1992 and protects relatively remote marine habitats, 113 km southwest of Key West with visitation largely limited to ferry or sea plane. The general management plan for Dry Tortugas NP was amended in 2001 incorporating a zoning scheme including 46% of the park area in a Research Natural Area, the highest level of habitat protection where natural processes are protected from human impact (including fishing).

Endangered Species Act Critical Habitat:
Under the Endangered Species Act of 1973, critical habitat may be designated by NOAA Fisheries for the conservation of threatened and endangered species under its jurisdiction. Critical Habitat designations were made for ESA listed corals, *Acropora palmata* and *A. cervicornis*, in 2008 to include hardbottom habitats < 30m depth deemed suitable to support recruitment of these corals (namely, stable hard substrate free of algae and sediment). Under this designation, over 3,000 sq km of habitat in the south Atlantic region are protected from destruction or 'adverse modification' by actions undertaken, funded, or permitted by federal entities.

State of Florida

In 2009, the Florida Legislature passed the Coral Reef Protection Act (CRPA, s. 403.93345, Florida Statutes [F.S.]) to increase protection of coral reef resources on sovereign submerged lands off the coasts of Martin, Palm Beach, Broward, Miami-Dade, and Monroe counties. The CRPA is intended to provide protection for corals and habitat from injury or destruction caused by commercial and recreational vessels that are operated, anchored, or moored in a reckless or wanton manner.

The CRPA authorizes the Florida Department of Environmental Protection (FDEP), as the state's lead trustee for coral reef resources, to protect coral reefs through timely and efficient assessment and recovery of coral reefs from damage. To carry out the intent of the Act, the FDEP also has the authority to enter into delegation agreements with state and local government agencies with coral reefs in their jurisdictions. The CRPA is overseen by the FDEP Coral Reef Conservation Program which works with FDEP regulatory or legal entities to ensure the Act is enforced.

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The FWC's Marine Life Rule (Rule 68B-42.009, F.A.C.) prohibits the take, destruction, and sale of marine corals, sea fans, and encrusting octocorals. This rule, enforced by FWC officers, prohibits intentional take and harvest of coral species.

State Parks and Aquatic Preserves

Several Florida State Parks and Aquatic Preserves have jurisdiction within the nearshore marine environment (Chapter 258, Florida Statutes and Chapter 62D-2, Florida Administrative Code and Florida Aquatic Preserve Act of 1975 (Section 258.35, Florida Statutes)). State parks are charged to, "*promote the state park system for the use, enjoyment, and benefit of the people of Florida and visitors; to acquire typical portions of the original domain … as to emblemize the state's natural values; conserve these natural values for all time; ... to enable the people of Florida and visitors to enjoy these values without depleting them; ...; to contribute to the tourist appeal of Florida".*

While Aquatic preserves are charged to, "*be managed primarily for the maintenance of essentially natural conditions, the propagation of fish and wildlife, and public recreation, including hunting and fishing where deemed appropriate by the Board, and the managing agency. ..."*

Coastal State Parks such as John D. MacArthur Beach State Park in Palm Beach County, John U. Lloyd Beach State Park in Broward County, Bill Baggs Cape Florida State Park in Miami-Dade County, and Indian Key, Long Key, Curry Hammock, and Bahia Honda State, and Fort Zachary Taylor Historic state parks in Monroe County have purview to manage resources up to 400 ft from the Mean High Water Line (F.A.C **62D-2.014**.9b). Other coastal managed areas such as St. Lucie Inlet Preserve State Park in Martin County; Biscayne Bay-Cape Florida to Monroe County Line Aquatic Preserve in Miami-Dade County; John Pennekamp Coral Reef State Park, Lignumvitae Key Botanical State Park, San Pedro Underwater Archaeological Preserve State Park, and Lignumvitae Key and Coupon Bight Aquatic Preserves in Monroe County have jurisdiction over submerged resources up to a mile offshore. As the first undersea park in the U.S., John Pennekamp Coral Reef State Park, was dedicated in 1960 and encompasses approximately 70 nautical square miles.

Local Action Strategies

While the reefs in the lower two-thirds of the FRT have had coordinated management for many years, the northern one-third in southeast Florida has lacked a comprehensive management plan and, until recently, an understanding of the local impacts and use in that region. Current

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initiatives in southeast Florida are bringing together the science and stakeholders to recommend management actions that preserve these reef resources while balancing resource use and protection.

In 1998, the United States Coral Reef Task Force (USCRTF) was established by Presidential Executive Order #13089 to lead U.S. efforts to preserve and protect coral reef ecosystems. During the eighth meeting of the USCRTF, held in Puerto Rico in 2002, the Task Force adopted the Puerto Rico Resolution, which called for the development of Local Action Strategies (LAS) by each of the seven member U.S. states, territories and commonwealths. These LAS are locally-driven roadmaps for collaborative and cooperative action among federal, state, territory and non-governmental partners which identify and implement priority actions needed to reduce key threats to coral reef resources. The goals and objectives of the LAS are closely linked to those found in the U.S. National Action Plan to Conserve Coral Reefs, adopted by the U.S. Coral Reef Task Force in 2000.

With guidance from the U.S. Coral Reef Task Force, the Florida Department of Environmental Protection (FDEP) and the Florida Fish and Wildlife Conservation Commission (FWC) coordinated the formation of a team of interagency marine resource professionals (state, regional, local, and federal), scientists, and reef resource stakeholders to form the Southeast Florida Coral Reef Initiative (SEFCRI) in 2004.

The SEFCRI which is coordinated and chaired by FDEP, while not a regulatory body, identified the reefs from St. Lucie Inlet in Martin County to the northern boundary of Biscayne National Park as their area of focus due to the lack of understanding of these reefs as well as the lack of a coordinated management plan. The SEFCRI identified the priority threats to southeast Florida's reef resources and developed 140 projects to better understand and reduce those threats. One of the original projects was to: "*Develop and* [sic] *effective, balanced, and comprehensive management strategy for improved resource protection...Organize and hold public workshops to obtain input on the condition and usage trends, possible resource goals, and the potential (i.e. rationale, effectiveness, alternative approaches, etc.) of traditional fishery management and special management zones to achieve targets*."

That original project is currently underway as the *OUR FLORIDA REEFS* community planning process for southeast Florida's coral reefs. Hosted by the SEFCRI, this planning process brings together the community of local residents, reef users, business owners, visitors, and the broader public in Miami-Dade, Broward, Palm Beach, and Martin counties to discuss the future of coral reefs in this region. This process is designed to increase public involvement in the future management of southeast Florida's coral reefs by seeking input from community members on the development of recommendations that can become part of a comprehensive management strategy to ensure healthy coral reefs in the future. The recommended management actions developed through the process will be taken to the various local, state, federal, or non-agency

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entities that could implement them. These recommendations address issues from land based sources of pollution; maritime industry and coastal construction; fishing, diving, boating, and other uses; enforcement; education and outreach; and place-based management.

Section 5 - Ongoing Threats (Margaret, Lauren, whole team needed to fill in.)

Many local actions create or exacerbate detrimental impacts to shallow coral reef ecosystems. Coastal construction and infrastructure development are particularly common near the urban centers from Palm Beach to Miami-Dade counties (Shivlani et al. 2011, Walker et al 2012). Dredge and fill activities such as beach nourishment and port maintenance and expansion result in direct loss of habitat and cumulative as well as acute effects to coral communities through increased turbidity and sedimentation (Wanless and Maier, 2007; Jordan et al. 2010). Beach nourishment activities are on-going especially within Palm Beach, Broward, and Miami-Dade counties. Recent (2015) port dredging at Port Miami greatly exceeded planned impacts by sedimentation to coral reef habitat, with another large dredging project upcoming at Port Everglades (Fort Lauderdale).

Overfishing has been suggested to result in a global decline of piscine predators with subsequent significant changes in the numbers of herbivores (Mumby et al. 2006). In the Caribbean, parrotfish overfishing has been hypothesized to be pivotal in adversely affecting corals in this region (Jackson et al. 2014). Decreases in parrotfish could result in increased macroalgae which directly outcompetes corals for space or inhibits coral recruitment. However,in the Florida Keys, herbivore-targeting fishing efforts have been relatively nonexistent (Bohnsack et al. 1994). Fishing activities such as that of trap fisheries more clearly create disturbance to reef benthic communities. Although trap fishers report generally avoiding coral reef habitats, ocean dynamics result in an accumulation of trap debris in coral-associated habitats (Uhrin et al 2014). These authors estimate the presence of almost two million items of lobster trap debris in the Florida Keys National Marine Sanctuary. The cover of benthic sessile fauna is reduced by ~ 10 % in areas affected by trap movement, events occurring over a wind threshold of 2 days duration at 15 kt (Lewis et al. 2009).

Water quality degradation from regional water management activities, sewage, coastal runoff, and local use likely have detrimental impacts (reviewed by Gregg 2013) with documented detriments to coral health (see Section XXX corals;). However, reef-scale impacts of water quality are difficult to partition from the myriad stressors which co-occur on reefs in the region. It is highly likely that both coastal hardening/construction and coastal water quality degradation will be exacerbated in the near future by rapid sea level rise from global climate change (Koch et al. 2015).

Comment [10]: Shivlani, M., M. Estevanez, L. McManus, C. Kruer, and T. Murray. 2011. Reference
Document and Guide to the Evaluation of Permitted Coastal Construction Activities
That Affect Coral Reef and Coastal Resources in Southeast Florida. Contract Report to SEFCRI and Florida Dept of Env Protection. Pg. iv and 159. Available from:http://www.dep.state.fl.us/coastal/programs/coral/reports/MICCI/MICCI_07_11_Reference_Document.pdf

Comment [11]: need ref

Comment [12]: need refs

Comment [13]: Uhrin AV, Matthews TR, Lewis C (2014) Lobster trap debris in the Florida Keys National Marine Sanctuary: distribution, abundance, density, and patterns of accumulation. Marine and Coastal Fisheries 6:20-32

Comment [14]: Lewis CF, Slade SL, Maxwell KE, Matthews TR (2009) Lobster trap impact on coral reefs: Effects of wind□driven trap movement. N Z J Mar Freshwater Res 43:271-282

Comment [15]: Gregg, K. 2013. Literature Review and Synthesis of Land-Based Sources of Pollution
Affecting Essential Fish Habitats in Southeast Florida. Report prepared for NMFS Southeast Regional Office. Available from http://www.dep.state.fl.us/coastal/programs/coral/reports/LBSP/LBSP_EFH_Lit_Review_and_Synth_Final.pdf

Comment [16]: Koch MS, Coronado C, Miller MW, Rudnick DT, Stabenau E, Halley RB, Sklar FH (2015) Climate change projected effects on coastal foundation communities of the Greater Everglades using a 2060 scenario: Need for a new management paradigm. Environ Manage 55:857-875

Invasive lionfish (*Pterois* spp.) likely continue to alter the structure of coral reef fish and invertebrate communities (Albins and Hixon 2008; Albins 2013, 2015; Green et al. 2014), and thus potentially alter coral reef ecosystem function. Lionfish impacts arise predominantly via direct predation (lionfish are voracious generalist predators - Morris and Akins 2009, Muñoz et al. 2011), but also likely occur through competition - e.g., for habitat or prey. Assessing the community- and broader-level impacts of lionfish is a critical need (see related text in Section 6).

With nearly 6 million residents in Martin (146,000), Palm Beach (1.3 million), Broward (1.75 million), Miami-Dade (2.5 million); and millions of visitors every year, awareness and appreciation of reef resources in the northern portion of the reef tract is severely lacking (US Census Bureau 2010).

Section 6 - Recommendations

Comment [17]: From http://www.coralreef.gov/ecosystem/:
•Understand coral reef community dynamics and the impacts of human-caused and natural stressors;
•Identify possible management strategies to mitigate negative impacts; and
•Evaluate the effectiveness of these management actions after they are implemented.

Knowledge Gaps:

-Tropicalization: effects of anticipated shifting species assemblages with warming temperatures.

- Where needed, expand knowledge of the distribution and benthic community attributes of coral reef ecosystems (e.g., via expanded mapping efforts in intermediate depths)

- Lionfish:

- While there have been multiple studies documenting local-scale effects of invasive lionfish (*Pterois* spp.) predation on native fish species (Albins and Hixon 2008; Albins 2013, 2015; Green et al. 2014), none of those studies have occurred in SAFMC-managed waters (a majority of the studies were performed in Bahamian waters), and no studies in any area have assessed the effect of lionfish predation over relatively broad scales. Research is needed to assess the realized effects of lionfish, via predation and potentially competition, on coral reef fish community structure at broader spatial scales (e.g., sub-regional, regional, ecosystem).
- There is considerable interest in controlling, reducing or depleting local lionfish populations through culling efforts (e.g., via spearfishers). Research is needed into (1) the effectiveness of culling efforts, in terms of the frequency and intensity of culling needed to maintain lionfish below targeted densities, (2) what target densities are most appropriate (e.g., near-zero, low, moderate…) in terms of reducing probable ecosystem impacts, and (3) assessing the trade-offs between the costs of culling efforts and the benefits (ecological and fishery-related) derived from those efforts.

- Assess and monitor spatial and temporal patterns in use of coral reef ecosystems in terms of fishing, snorkeling / diving and other uses

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- Assess efficacy (direct and indirect results) of management actions such as MPAs
- Identify fish and invertebrate spawning habitats or locations, and the degree to which spawning aggregations are targeted by fishers

- Due to repeated reef impacts from large dredging and beach projects in the area, from direct disturbance and ongoing turbidity, (Wanless & Maier 2007), there is a need for better understanding chronic and acute turbidity and/or sedimentation on coral reef habitats.

Potential Management recommendations:

-Develop and implement numeric water quality standards, including for turbidity, that are protective of coral reef habitats (Gregg 2013)

- "Focused removal of submerged trap debris from especially vulnerable habitats such as reefs and hardbottom, where trap debris density is high, would mitigate key habitat issues but would not address ghost fishing or the cost of lost gear." (Uhrin et al 2014)

-Diadema restoration (Acropora Recovery Plan, and Florida Pillar Coral Action Plan)

- The Our Florida Reefs community planning process is developing and vetting Recommended Management Actions to improve conservation of southeast Florida reef ecosystems. While these recommendations are not yet final, several key recommendations include

1) seasonal protection of spawning aggregations;
2) reducing spearfishing activity on SCUBA;
3) adjusting Spiny Lobster recreational catch limits within State waters;
4) enforcing better protection of key herbivorous species from fishing (e.g. Parrotfish);
5) the recognition of southeast Florida reefs need for a coordinated management plan which includes a network of areas-of-interest for place-based management. This network of place-based management could include coordination with existing managed areas (e.g. see State Parks and Aquatic Preserves), creation of areas for seasonal protection, marine reserves, and areas identified for restoration.

- Coral reef habitats are impacted by ongoing and repeated damage from dredging and coastal construction projects in the region. Much of this damage should be preventable under existing regulations, but improvements in permitting, monitoring, implementation, compliance and enforcement are needed. Specific recommendations for such improvement are provided in Lindeman and Ruppert (2011) include

Comment [18]: Wanless, H. R., & Maier, K. L. 2007. An evaluation of beach renourishment sands adjacent to reefal settings, southeast Florida. Southeastern Geology 45(1):25-42.

Comment [19]: Gregg, K. 2013.Literature Review and Synthesis of Land-Based Sources of Pollution
Affecting Essential Fish Habitats in Southeast Florida.Report Prepared for: NOAA Fisheries Southeast Region Habitat Conservation Division, West Palm Beach. http://www.dep.state.fl.us/coastal/programs/coral/reports/LBSP/LBSP_EFH_Lit_Review_and_Synth_Final.pdf

Comment [20]: Uhrin AV, Matthews TR, and Lewis C. 2014. Lobster trap debris in the Florida Keys National Marine Sanctuary: distribution, abundance, density, and patterns of accumulation. Marine and Coastal Fisheries 6:20-32.

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Comment [22]: Lindeman, K.C. and T. Ruppert. 2011. Policy recommendations and training to improve agency permitting, compliance, and enforcement for coral resource conservation in southeast Florida. Florida Dept. of Environmental Protection, Southeast Florida Coral Reef Initiative, 207 pp.

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· Development of a template by permitting agencies with standard language for 'special conditions' to avoid, minimize, and monitor coral impacts
· Development by NMFS of regulatory criteria to identify 'destruction or adverse modification' of ESA Critical Habitat, replacing the current working definition.

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CHAPTER: SHALLOW WATER CORALS -

Section 1 - Taxonomy and Life History

Stony corals are marine invertebrates that secrete a calcium carbonate skeleton. Stony corals include members of both the Class Hydrozoa (fire corals and lace corals) and Order Scleractinia (true stony corals). Most reef-building corals are zooxanthellate, hosting symbiotic algae from the genus *Symbiodinium* in their gastrodermal cells. These symbionts provide a phototrophic contribution to the coral's energy budget, enhance calcification, and give the coral most of its color. The largest colonial members of the Scleractinia help produce the carbonate structures known as coral reefs in shallow tropical and subtropical seas around the world.

For the purpose of this plan, Octocorals include species belonging to the Class Octocorallia , Order Alcyonacea (soft corals and gorgonians). Similar to stony coral corals, octocorals are colonial animals with a polyp as the individual building unit and may contain endosymbiotic algae (zooxanthellae). Unlike stony coral, octocorals do not secrete a calcium carbonate skeleton but have an axial skeleton mainly composed of collagen fibers in a proteinaceous matrix. Although octocorals do not contribute to reef framework, they do contribute greatly to reef complexity and diversity.

Table XXX Classification of corals included under the Council's Coral, Coral reefs and Live/ Hard Bottom Fishery Management Plan.

- Phylum Cnidaria
  - Subphylum Medusozoa
    - Class Hydrozoa
      - Order Anthoathecata
        - Suborder Capitata
          - Family Milleporidae (fire, stinging corals)
        - Suborder Filifera
          - Family Stylasteridae (lace corals)
  - Subphylum Anthozoa
    - Class Anthozoa
      - Subclass Hexacorallia (or Zoantharia)
        - Order Scleractinia
      - Subclass Octocorallia
        - Order Alcyonacea (soft corals)
          - Suborder Alcyoniidae (soft corals)
          - Suborder Scleraxonia (gorgonians)

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Suborder Holaxonia (gorgonians)
Suborder Calcaxonia (gorgonians)

Corals can reproduce asexually when fragments break off and reattach to the reef. However, corals also have a complex life cycle including pelagic (sexual) larval and sessile, usually colonial, adult phases. There are a multitude of breeding systems described among scleractinian corals (Baird et al. 2009) with the primary categories being brooding vs. broadcast spawning, and hermaphroditic vs. gonochroic. The primary reef-building species in the region, including *Acropora* spp. and *Orbicella* spp. are hermaphroditic (colonies produce both eggs adn sperm), broadcast spawners (gametes are shed into the water column where they undergo fertilization and development). Dilution, advection, and other environmental stressors in the open ocean environment yield lower rates of fertilization, higher rates of larval mortality, and greater average dispersal distance by broadcasted, compared with brooded larvae. Brooded larvae are released with symbionts inherited from the parent colony enabling them to renew energy reserves via photosynthesis and are generally able to settle soon after they are released from the parent colony. In contrast, broadcast larvae must rely on lipid reserves from its egg and remain in the water column from a few days to weeks to complete larval development prior to settlement competence. Hence, broadcasting species (with few exceptions, predominantly *Siderastraea siderea*) generally display much lower rates of larval recruitment than brooding species, in some cases vanishingly low. It is likely that both low larval production and declining habitat quality (due to sediments, turf and macroalgae) contribute to low recruitment in broadcast-spawning, reef-building corals in the region.

After metamorphosis onto appropriate hard substrata, metabolic energy is diverted to colony growth and maintenance. Because newly settled corals barely protrude above the substratum, juveniles need to reach a certain size to reduce damage or mortality from impacts such as grazing, sediment burial, and algal overgrowth (Bak and Elgershuizen 1976; Birkeland 1977; Sammarco 1985). Generally, mounding corals grow slowly; most growth rates (linear extension) for *Montastraea*, *Porites*, and *Diploria* are less than 1 cm per year. Hubbard and Scaturo (1985) report average extension rates of 0.12-0.45 cm/yr for several species including *Stephanocoenia intersepta*, *Agaricia agaricites*, *Diploria labyrinthiformis*, *Colpophyllia natans*, *Montastraea cavernosa*, *Porites astreoides*, and *Siderastrea siderea*. Growth rates for branching species are generally higher, with branch extension rates over 10 cm per year commonly reported for *Acropora cervicornis* in the Florida Keys, and even higher rates of total productivity in local in situ *A.cervicornis* nurseries (Lirman et al. 2014). However, long term reductions in coral growth rates are expected under near term future scenarios of climate warming/temperature extremes and acidification (refs) as these stressors reduce the efficiency of calcification.

Octocorals have not been studied as extensively as scleractinian corals and their reproductive biology is poorly known for most species. In 2009, Simpson performed a review of published literature on octocoral reproduction and all known reproduction systems of octocorals are described therein. Like scleractinian corals, both sexual and asexual reproduction have been

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documented in octocorals. Types of asexual reproduction include fragmentation, fission (commonly observed in encrusting species), and development of new colonies from stolons or runners. Asexual reproduction is known to be more common in true “soft corals” and is limited to only a few octocoral species found in Florida. The vast majority of gorgonian octocorals reproduce sexually by broadcast spawning or brooding (either internally or externally). The reproductive strategy of external or surface brooding has been documented in octocorals, where eggs are released passively onto the surface of the colony (Benayahu and Loya 1983, Brazeau and Lasker 1990, Gutiérrez-Rodriguez and Lasker 2004). While sampling female colonies of *Antillogorgia* (*Pseudopterogorgia*) *elisabethae*, Gutiérrez-Rodriguez and Lasker (2004) did not find developing embryos or planulae inside the polyps, and they suggested that fertilization occurred either internally immediately before the eggs were released or externally on the surface of the maternal colony.

Comment [26]: Benayahu Y. and Y. Loya. 1983. Surface brooding in the Red Sea soft coral Parerythropodium fulvum fulvum (Forskaal, 1775). Biological Bulletin 165: 353-369.

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As with stony corals, octocoral planulae settle onto an appropriate substratum and undergo metamorphosis into a feeding polyp. Octocorals are known to settle in shaded microhabitats, such as the underside of settlement plates, small cavities in the substratum or under clumps of macroalgae. Studies suggest that this settlement behavior may be influenced by turbulent eddies that facilitate larval settlement and an avoidance response to unfavorable conditions such as high light intensity, low tides, predator grazing pressure, and sedimentation (Simpson 2009, Benayahu and Loya 1987). Studies have indicated that successful settlement and recruitment into a population occurs at a low rate (Lasker et al. 1998, Simpson 2009). Lasker et al. (1998) suggested that extremely high post-settlement mortality of new recruits indicates that successful settlement may be more related to water column and post-settlement survival than to gamete production and fertilization rates. Despite low recruitment rates, octocorals are excellent spacial competitors and are known to have much higher growth rates in general as compared to most species of scleractinian corals. Cary (1914) discussed the obvious advantage of young octocorals over stony coral recruits in that their most rapid growth is perpendicular to the substratum, keeping the most active growing part of the colony in a favorable position for resource allocation.

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Comment [30]: Lasker, H.R., K. Kim, and M.A. Coffroth. 1998. Production, settlement, and survival of plexaurid gorgonian recruits. Marine Ecology Progress Series 162: 111-123.

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Section 2 - Abundance and Trends of coral populations (Dave, Rob, Mark,)

Scleractinians

SEFL

The reefs offshore the mainland coast of southeast Florida, the northern extension of the Florida Reef Tract (FRT), have a similar stony coral diversity to that of the southern regions of the FRT (the Florida Keys and Dry Tortugas) and much of the Caribbean, but benthic cover, 2-5%, is generally lower and colony size, average less than 20 cm diameter, is generally smaller (Gilliam et al 2014, Gilliam et al 2015). Nearly 30 species of stony corals have been identified, but six species (*Montastraea cavernosa*, *Siderastrea siderea*, *Porites astreoides*, *Stephanocoenia intersepta*, *Agaricia agaricites*, and *Meandrina meandrites*) contribute greatly to benthic cover and colony density (Gilliam et al 2014, Gilliam et al 2015). Three of these species (*M. cavernosa*, *S. siderea*, and *P. astreoides*) were also identified as being three of the most common species in the Florida Keys (Ruzicka et al. 2013) and in the Dry Tortugas (Ruzicka et al. 2012). Two long-term monitoring programs have been operating since at least 2003 and neither has documented a significant trend in stony coral benthic cover (Gilliam et al 2014, Gilliam et al 2015). This is in contrast to much of the Caribbean (Gardner et al. 2003, Jackson et al. 2014) and the southern regions of the FRT (Ruzicka et al. 2014).

FLK

Octocorals

SEFL

Octocorals are a significant component of the reef community along the FRT. Offshore southeast Florida octocoral colony density and species diversity tend to be greater than those of stony corals. Octocoral benthic cover, 3-20%, is also generally higher than stony coral. Octocoral cover has shown a significant decreasing trend in parts of the region (Gilliam et al 2015) which is in contrast to significantly increasing trend identified in the Florida keys (Ruzicka et al. 2014).

Section 3- Threats

Mounting threats of myriad sorts have resulted in drastic declines in scleractinian corals, both in the South Atlantic region and throughout the Caribbean, over the past few decades. Recent analyses of extinction risk for seven coral species concluded that global changes (including warming and changing ocean chemistry) along with disease pose the greatest threat to coral extinction (Brainard et al 2011). These global threats are superimposed and interaction with additional stressors at the local level (also reviewed in Brainard et al. 2011). The relative importance of these local stressors vary somewhat across the South Atlantic region, related to the local human population density and use along the coast.

Global climate change has already caused significant coral declines in the region, with notable increases in year-round local reef sea surface temperature documented over the past century and is estimated at an annual rate of 0.9^{o}C over the past 3 decades (Kuffner et al 2014). As a result, the occurrence of warm temperature stress above bleaching thresholds is projected to occur annually within the next decade, much sooner than global climate models predict (Manzello 2015). Mass coral bleaching events have resulted from warm temperature extremes in 1997-8, 2005, 2014 and 2015. Many corals die directly from bleaching and also from subsequent coral disease outbreaks following the physiological stress of bleaching (Brandt & McManus 2009). Due to high latitude, episodic cold water events also affect South Atlantic corals, particularly in 2010 when cold water caused mass coral mortality, especially in nearshore patch reef habitats (Lirman et al. 2011).

Comment [32]: Gardner TA, Cote IM, Gill JA, Grant A, and Watkinson AR. 2003. Long-term region-wide declines in Caribbean corals. Science 301:958-960.

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Coastal water quality in the region is affected by broad scale regional water management actions, sewage via both offshore outfalls and seepage from septic tanks, runoff and stormwater. The effect of these combined constituents, including endocrine disruptors, pesticides, nutrients, freshwater, etc. are poorly characterized (but see Downs et al. 2005, Edge et al 2013, Ross et al. 2015) but most certainly detrimental to health of corals in the region, consequently reducing their physiological scope to deal with global stressors.

Fishing is another factor which has imposed significant reduction in fish biomass in reef ecosystems in the South Atlantic region. Meanwhile, fishes are engaged in important positive feedbacks with corals including grazing to maintain benthic habitat quality and nutrient delivery (Shantz et al. 2015). Although parrotfishes are not highly targeted in local fisheries as in other Caribbean regions allowing persistence of high grazing (Paddack et al. 2006), this is a factor which should be monitored as fisheries preferences may change over time.

While the effects of many stressors causing direct coral mortality are relatively easy to observe, many sublethal stressors such as sedimentation, water-born toxicants, acidification, chronic temperature stress, and non-lethal diseases impair the replenishment capacity of coral populations both by impairing larval output and by impairing larval survival and/or recruitment (e.g., Jones et al. 2015, Albright et al 2010).

The effects of ocean acidification (i.e. changes in the carbonate chemistry of ocean waters), water quality, and trophic disruption threats are less well characterized for octocorals, though warm temperature bleaching and disease have both been documented, particularly in sea fans (refs). Unlike scleractinians, some octocorals are also subject to harvest (Miller et al. 2014).

Section 4 - Management

Scleractinian corals are currently managed under a zero-take FMP and are protected as Essential Fish Habitat - Habitat Areas of Particular Concern. Seven species in the region are also protected as threatened species under the US Endangered Species Act, with one of these (Dendrogyra cylindrus) previously designated as a Threatened species by the state of Florida. Hence, an ESA Recovery Plan (for Acropora palmata and A.cervicornis) and Florida Species Action Plan (for D. cylindrus) both provide relevant actions for coral conservation and restoration in the region.

Octocorals are currently managed by the State of Florida under chapter 68B of the Florida Administrative Code (FAC). The State of Florida defines octocorals as "any erect, non-encrusting species of the Subclass Octocorallia, except the species *Gorgonia flabellum* and *G. ventalina*" which are prohibited (FAC 68B-42.002). Up to six octocoral colonies per day may be collected recreationally with a Florida Recreational Saltwater Fishing License (FAC 68B-42.005). There are no limits on the harvest of octocorals for commercial purposes. However, the annual quota for octocorals harvested in State of Florida and adjacent Federal waters is 70,000 colonies (FAC 68-42.006). No power tools may be used to harvest colonies and only one

Comment [39]: Downs C, Fauth J, Robinson C, Curry R, Lanzendorf B, Halas J, Halas J, Woodley C (2005) Cellular diagnostics and coral health: Declining coral health in the Florida Keys. Mar Pollut Bull 51:558-569

Comment [40]: Edge SE, Shearer TL, Morgan MB, Snell TW (2013) Sub-lethal coral stress: Detecting molecular responses of coral populations to environmental conditions over space and time. Aquatic Toxicology 128–129:135-146

Comment [41]: Ross C, Olsen K, Henry M, Pierce R (2015) Mosquito control pesticides and sea surface temperatures have differential effects on the survival and oxidative stress response of coral larvae. Ecotoxicology 24:540-552

Comment [42]: Shantz AA, Ladd MC, Shrack E, Burkepile DE (2015) Fish-derived nutrient hotspots shape coral reef benthic communities. Ecol Appl

Comment [43]: Paddack M, Cowen R, Sponaugle S (2006) Grazing pressure of herbivorous coral reef fishes on low coral-cover reefs. Coral Reefs 25:461-472

Comment [44]: Jones R, Ricardo GF, Negri AP (2015) Effects of sediments on the reproductive cycle of corals. Mar Pollut Bull 100:13-33

Comment [45]: Albright R, Mason B, Miller M, Langdon C (2010) Ocean acidification compromises recruitment success of the threatened Caribbean coral Acropora palmata. Proceedings of the National Academy of Sciences 107:20400-20404 inch of attached substrate around the perimeter of the base of the octocoral holdfast may be removed (FAC 68B-42.006, 68B-42.007, FAC 68B-42.008). Octocorals must be collected and landed live and stored in a re-circulating live-well or oxygenated system aboard the collection vessel (FAC 68B-42.0035).

Areas that are closed to octocoral collection include Atlantic Federal waters north of Cape Canaveral, Biscayne National Park, and in the Stetson-Miami Terrace Deep Water Coral Habitat Area of Particular Concern, as well as the Pourtales Terrace Deep Water Habitat Area of Particular Concern adjacent to Florida state waters (FAC 68B-42.0036). Additional area closures for marine life collection exist in southeastern Florida, including National Parks (Everglades, Biscayne, Dry Tortugas), John Pennekamp Coral Reef State Park, and the Florida Keys National Marine Sanctuary, including the Key Largo Management Area (formerly Key Largo National Marine Sanctuary), the Looe Key Management Area (formerly Looe Key National Marine Sanctuary), and various smaller no-take zones including sanctuary preservation areas, special-use/research-only areas, and ecological reserves (Miller et al. 2014). For further information, Miller et al. (2014) prepared an in-depth description of the U.S South Atlantic Octocoral Fishery.

Comment [46]: Miller SL, Espitia P, Chiappone M, Rutten LM (2014) Description of the U.S. South Atlantic Octocoral Fishery: 2014 Final Report to the South Atlantic Fishery Management Council. National Coral Reef Institute, Nova Southeastern University Oceanographic Center, Dania Beach, FL. 184 pp

Section 5-Recommendations

Coral Knowledge Gaps:

-Efficacy and improvement of coral (proactive) restoration strategies (Hunt & Sharp 2014)
- Efficacy of coral predator removal or other mitigation (Acropora Recovery Plan)
- Carrying capacity of coral disease, predation, (Acropora Recovery Plan)
- Impact threshold levels for nutrients, sedimentation, toxicants (Acropora Recovery Plan)
- Determine causal factors in coral disease impacts, especially regarding interactions with temperature and local anthropogenic stressors. (Acropora Recovery Plan)
- Due to repeated reef impacts from large dredging and beach projects in the area, from direct disturbance and ongoing turbidity, there is a need for better understanding of chronic and acute turbidity and/or sedimentation on all life phases of shallow corals, including recruitment.

Comment [47]: Hunt J and Sharp W. 2014. Developing a Comprehensive Strategy for Coral Restoration for Florida.State Wildlife Grant Award T-32-R 1169 Final Report. http://myfwc.com/media/2982098/CoralRestoration.pdf

Comment [48]: Wanless, H. R., & Maier, K. L. 2007. An evaluation of beach renourishment sands adjacent to reefal settings, southeast Florida. Southeastern Geology 45(1):25-42.

Coral Potential Management Recommendations

-Coral population enhancement (Acropora Recovery Plan and Our Florida Reefs)
- Enhanced mooring balls in sensitive areas (Florida Pillar Coral Aciton Plan; OFR)
- Enhanced legal enforcement of Florida Coral Reef Protection Act (Florida Pillar Coral Action Plan)
- Improve coastal construction project permitting/compliance/mitigation to achieve 'no net loss' of coral

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Live/Hardbottom Habitat

The continental shelf off the southeastern United States, commonly called the South Atlantic Bight (SAB), extends from Cape Hatteras, North Carolina, to Cape Canaveral, Florida (or according to some researchers, to West Palm Beach, Florida). The northern part of the SAB is known as the Carolina Capes Region, while the middle and southern areas are called the Georgia Embayment, or Georgia Bight. The Carolina Capes Region is characterized by complex topography. The prominent shoals there extend to the shelf break and are effective in trapping Gulf Stream eddies, whereas the Georgia Embayment to the south is smoother.
Shelf widths of the South Atlantic Bight vary from just a few kilometers off West Palm Beach, FL, to a maximum of 120 km off Brunswick and Savannah, Georgia. Gently sloping shelves (about 1m/km) can be divided into the following zones based on depth. The shallowest is the inner shelf zone (5-20 m, 16-66 ft.) which is dominated by tidal currents, river runoff, local wind forcing and seasonal atmospheric changes (Table 1). The mid-shelf zone (20-40 m, 66-98 ft.)waters are dominated by winds but influenced by the Gulf Stream. Stratification of the water column changes seasonally; mixed conditions, in general, characterize fall and winter while vertical stratification prevails during spring and summer. Strong stratification allows offshore upwelled waters to advect farther onshore near the bottom and, at the same time, it facilitates offshore spreading of lower salinity water in surface layer. Further offshore, the outer-shelf zone (30-50 m, 98-164 ft.) is dominated by the Gulf Stream.

Generally, the shelf edge or break occurs between 50-100 m depth (164-328 ft.) but occurs shallower to the south of Cape Canaveral in into the Florida Keys. The shelf edge is the transition from a gradually sloping shelf area to relatively steeper slopes compared to the shelf area. Offshore of the shelf edge, the upper slope occurs in 100 to 300 m (328 to 984 ft.), and the mid slope is slightly deeper at 300-400 m (984-1,312 ft.). The slope areas include habitats such as the Big Rock, Blake Plateau, Charleston Bump, and Pourtales Terrace. Deep offshore and deep areas occur in depths greater than 400 m (1,312 ft.).

Table 1. Approximate depth distribution of hardbottom habitat zones in the southeastern U.S.

Better chart in Word Doc

Hardbottom Habitat Zones	Depth (m)	Depth (ft)
Nearshore	0-5	0-16
Inner Shelf	5-20	16-66
Mid Shelf	20-30	66-98

Outer Shelf	30-50	98-164
Shelf Edge	50-100	164-328
Upper Slope	100-300	328-984
Mid Slope	300-400	984-1,312
Deep Offshore	400-5,000	1,312-16,404
Deep	>5,000	>16,404

Ecological role and function

Hardbottom structure provides protective cover for numerous fish and invertebrate species and increases the surface area available for colonization by sessile invertebrates and macroalgae through increased relief and irregularity of the structure. The variability in abundance and diversity of fish on hardbottom and artificial reefs is related to the amount and type of structural complexity of the reef (Carr and Hixon 1997, Schobernd and Sedberry 2008) and likely explains invertebrate diversity and abundance similarly. Because of the structural complexity, natural reefs can sustain greater 10 times the fish biomass compared to non-reef open shelf bottom (Huntsman 1979, Wenner 1983). In addition, areas with small patches of hardbottom surrounded by sand bottom supported greater fish abundance and diversity than one large area of equal material, suggesting the importance of habitat edge and diversity to ecosystem productivity (Bohnsack et al. 1994, Auster and Langton 1999).

Nearshore and inner shelf hardbottom areas serve as important settlement and nursery habitat for early life history stages of many important fisheries species (Lindeman and Snyder, 1999). Nearshore hardbottom also serves as intermediate nursery habitat for late juveniles emigrating out of estuaries (CSA 2009).

In addition to being an important settlement and nursery areas in shallow waters, deeper hardbottom areas are important spawning areas for some reef fishes (Heyman et al. 2005, Sedberry et al. 2006, Coleman et al. 2011). Spawning occurs on nearshore hardbottom for Black Sea Bass (*Centropristis striata*), Sand Perch (*Diplectrum formosum*), Sheepshead (*Archosargus probatocephalus*), Atlantic Spadefish (*Chaetodipterus faber*) and some additional non-fishery reef species (Powell and Robins 1998, F. Rohde, DMF, pers. com., 2001). Spawning for most managed reef fish occurs on mid- and outer-shelf reefs. Riley's Hump is an example of mid-shelf spawning location identified as a spawning aggregation area for Mutton Snapper (*Lutjanus analis*) and may serve in a similar fashion for other snapper/grouper species (Locascio and Burton 2016). Similarly, many deep-water reef species spawn on the upper slope and Blake

Plateau (Sedberry et al. 2006, Locascio and Burton 2016, Farmer et al. in prep.). Other potential hardbottom spawning areas were included in Amendment 14 for MPA protection (Figure xx), and additional sites have been identified in Snapper-Grouper Amendment 36 as Spawning Special Management Zones to further protect spawning reef fishes (Figure xx). In the Amendment 14 MPAs and Spawning SMZs, fish in spawning condition have been observed in the area or have been reported through anecdotal reports (SAFMC 2016a, SAFMC 2016b, Farmer et al. in prep). Although approved by the SAFMC, regulations for the SMZs have not been enacted as of July 2016.

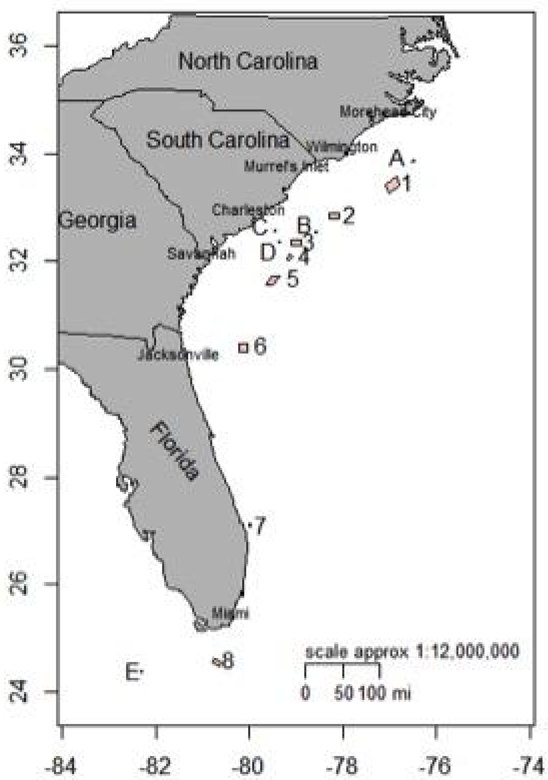

Figure xx. Map of the South Atlantic Region's Deepwater MPAs (Numerals) and Spawning Special Management Zones (Characters). 1=Snowy Wreck MPA, 2=Northern South Carolina MPA, 3=Edisto MPA, 4=Charleston Deep Artificial Reef MPA, 5=Georgia MPA, 6=North Florida MPA, 7=St. Lucie Hump MPA, 8=East Hump MPA, A=South Cape Lookout SMZ, B=Devil's Hole SMZ, C=Area 51 SMZ, D=Area 53 SMZ, E=Warsaw Hole SMZ.

Nearshore

Nearshore hardbottom habitats in the South Atlantic Region are predominantly found on the east of Florida in depths of 0-5 m. These are primarily accretionary ridges of coquina shells, sand, and shell marl that lithified parallel to ancient shorelines during Pleistocene interglacial periods (*Insert Gram, 1965,* Duane and Meisburger 1969) and are patchily distributed among large expanses of barren coarse sediments. These hardbottoms commonly possess tubeworm reefs, macro and turf algae, and low coral diversities. The habitat complexity of nearshore hardbottom is expanded by mounds of tube-building polychaete worms (Kirtley and Tanner 1968; McCarthy 2001), other invertebrates and macroalgae (Goldberg 1973; Nelson and Demetriades 1992). Hard corals are rare due to high turbidities and wave energy. However, hard corals that are found off mainland of Florida from St. Lucie to Broward include *Acropora cervicornis*, *Oculina diffusa, Oculina varicosa,* and *Siderastrea spp* (CSA 2009). A large array of literature and many new species records are summarized for algae (277 species total), invertebrates (523 species), fishes (257) and sea turtles from nearshore hardbottom of mainland east Florida in CSA (2009).

Based on visual censusing of fishes in three mainland southeast Florida sites over two years, 86 species from 36 families were recorded (Lindeman and Snyder 1999). Pooled early life stages (newly settled, early juvenile, and juvenile) represented over 80% of the individuals at all sites. Nearshore hardbottom fish assemblages of this subregion are characterized by diverse tropical faunas which are dominated by early life stages (Lindeman and Snyder 1999).

Over 190 fish species within 62 families have now been recorded in association with nearshore hardbottom habitats of mainland southeast Florida from Palm Beach County to Ft. Lauderdale (Futch and Dwinell 1977, Gilmore 1977, Gilmore et al. 1983, Vare 1991, Gilmore 1992, Lindeman and Snyder 1999, Baron et al. 2004). At least 90 species are utilized in recreational, commercial, bait, or aquaria fisheries. Some of the important taxa identified included Haemulon (grunts), clupeids (herrings and sardines), carangids (jacks), and engraulids (anchovies). Nearshore hardbottom habitats typically had over thirty times the individuals per transect as natural sand habitats (Lindeman and Snyder 1999) and newly settled individuals were not recorded during any surveys of natural sand habitats.
- KL will update and add Jordan papers

Significant differences ($p<0.05$) in total abundance, species richness and biomass were noted among the three reef tracts off Broward County, FL (Ferro et al. 2005). In general, greater species richness and fish abundance was found on the offshore reef tract than on the middle or inshore reef tracts. The juvenile grunts, an important forage base, were significantly higher on the inshore and middle reefs, which did not differ significantly from each other, than on the offshore reef. Of management interest, the results of this census highlight a scarcity of legal size groupers (2) and snappers (198) over the entire survey.

Off mainland east Florida, nearshore hardbottom is often colonized by sabellariid worm reefs (*Phragmatopoma lapidosa*) that go through predictable patterns of annual change which include high recruitment in early autumn through winter, rapid reef growth (~0.5 cm/day) resulting in maximum structure in spring and summer, and decay by early autumn (McCarthy 2001; McCarthy 2003). As recruits grow, the structure of their reef changes and these changes are important in determining the resiliency of the reefs when disturbed. Juveniles form low-lying mounds and reefs that often survive winter wave and sand disturbance (McCarthy 2001). As individuals continue to grow and accrete sand, they form large reefs that reach maximum size during the summer. Many of the intertidal colonies grow into somewhat unstable mushroom-shaped mounds whereas subtidal *P. lapidosa* mounds generally remain carpet-like in shape (McCarthy 2001).

Mortality of *P. lapidosa* colonies increases during the summer as a result of the effects of several disturbance agents (McCarthy 2001). In the early summer, some individuals at the tops of intertidal mounds perish, leaving the tops susceptible to decay. It is likely that this mortality is caused by desiccation and/or heat stress from extreme summer temperatures. By the late summer and early autumn, wave activity from hurricanes results in maximum physical disturbance to sabellariid reefs. A large percentage of both intertidal and subtidal reefs are severely damaged at this time. Intertidal worms are more susceptible to physical destruction of their colonies, whereas subtidal worms get smothered by sand but the sand reef remains intact.

Almost simultaneously with peaks in lethal disturbance, however, larvae of *P. lapidosa* arrive in large numbers to renew the colonies by massive recruitment in cracks or atop mounds of adults (McCarthy 2001). This process results in low lying reefs that are highly resilient and will eventually restore the structure of the reefs. Consequently, as disturbance lowers adult abundance and creates new settlement space, new individuals arrive in sufficient numbers to restore the populations. Therefore, local metapopulations may remain at fairly high abundances year after year while experiencing moderately high mortality from various agents of disturbance. When these seasonal data are integrated with those of other researchers (Gilmore 1977; Gilmore et al. 1981; Lindeman and Snyder 1999), they reveal important links between the seasonal cycle of sabellariid reef expansion and degradation, and the occupation of those reefs by juvenile and adult organisms.

Nearshore hardbottom habitats of the Florida Keys can differ both geologically and biologically from mainland areas. Within the Keys, nearshore hardbottom is widely distributed and shows compositional differences based on proximity to tidal passes (Chiappone and Sullivan 1994). Near tidal passes, these habitats can be dominated by algae, gorgonians and sponges. In the absences of strong circulation, such habitats are characterized by fleshy algae, such as *Laurencia* (Chiappone and Sullivan 1994). Hard corals are relatively uncommon in nearshore areas of mainland east Florida, presumably due to greater environmental variability in key parameters (temperature, turbidity, salinity).

Inner Shelf

The hard-bottom areas of the inner shelf are typically found in depths between 5 and 20 meters. In more temperate regions, the inshore areas at depths less than 18 m have seasonally-variable temperatures, less diverse populations of invertebrates, and are inhabited primarily by Black Sea Bass, Scup and associated warm-temperate species (Sedberry and Van Dolah 1984).

Most of the substrate on the inner shelf of the SAB is covered by a vast plain of sand and mud (Newton et al. 1971) underlain at depths of less than a meter by surface sand (Riggs et al. 1996; Riggs et al. 1998). The fish biomass of this sand- and mud-covered plain is relatively low. Scattered irregularly over the shelf, however, are zones of highly concentrated invertebrate and algal growth, usually in association with marked deviations in relief that support substantial fish assemblages (Struhsaker 1969; Huntsman and Mcintyre 1971; Wenner et al. 1983; Chester et al. 1984; Sedberry and Van Dolah 1984; Sedberry et al. 1998; Sedberry et al. 2001; . Studies that have examined fish assemblages on natural and artificial reef habitats include Huntsman and Manooch (1978), Miller and Richards (1980), Grimes et al. (1982), Lindquist et al. (1989), Potts and Hulbert (1994), Parker and Dixon (1998), SAFMC-SA 2001; Ojeda et al. 2004; Whitfield et al. (2011).

The federal waters of the inner shelf off Georgia includes an MPA, Gray's Reef National Marine Sanctuary. The Sanctuary contains excellent examples of high- and moderate-relief ledges, low relief hardbottom (often covered with a veneer of sand) and sand plains. Roughly one third of the Sanctuary (eight square miles) is a no-fishing zone; the remainder is a popular recreational fishing site.

South of Ft. Pierce Inlet, Florida, the shelf becomes increasingly tropical through the Florida Keys. This is reflected in an increase in corals and associated organisms. See the Coral Chapter of the Fishery Ecosystem Plan (in prep.) and Reigl and Dodge (2008) for greater detail. In southeast Florida, several parallel ridges of hardbottom reefs, derived from Pleistocene and Holocene reefs, begin in depths usually exceeding 8 m (Goldberg 1973; Lighty 1977). The geologic origins and biotic characteristics of these inner shelf reef systems are different from the nearshore hardbottom reefs (Lighty 1977), although reefs of both strata are lower in relief than reefs of the Florida Reef tract. Using various collecting gears and literature reviews, Herrema (1974) recognized the occurrence of 206 fishes off the mainland southeast coast of Florida. Lutjanids, haemulids and many other families were represented in both subregions on almost a species by species basis (Herrema 1974). This information was not contradicted by the faunal characterizations in Courtenay et al. (1974, 1980). Based primarily on offshore records, Perkins et al. (1997) identified 264 fish taxa from the shelf of mainland Florida as hardbottom obligate taxa.

Mid Shelf

Off the temperate southeastern United States, most hard/live bottom habitats occur at depths greater than 27 m (90 ft), but many also are found at depths of from 20 to 30 m (66 to 98 ft), especially off the coasts of North Carolina and South Carolina, and within Gray's Reef National Marine Sanctuary off Georgia. Studies of live bottom areas from North Carolina to northern Florida (Continental Shelf Associates, 1979; Wenner *et al*., 1983) revealed three habitat types: 1) emergent hardbottom dominated by sponges and gorgonian corals; 2) sand bottom underlain by hard substrate dominated by anthozoans, sponges and polychaetes, with hydroids, bryozoans, and ascidians frequently observed; and 3) softer bottom areas not underlain with hardbottom. See the Coral Chapter of the Fishery Ecosystem Plan (in prep.) and Reigl and Dodge (2008) for greater detail on mid-shelf hardbottom and coral associated fauna.

Shelf Edge

At the first break on the edge of the continental shelf, there are outcroppings of sedimentary rock and steep dropoffs (10 m or more) in the zone from 50 to 100 meters. High-relief rock outcrops are especially evident at the shelf break, a zone from about 55-200m where the continental shelf ends and the upper slope begins; this area is often characterized by steep cliffs and ledges (Huntsman and Manooch 1978; Sedberry et al. 2001; Wenner and Barans 2001; Fraser and Sedberry 2008; Schobernd and Sedberry 2009). At the shelf break, the topography is a discontinuous series of terraces before sloping or dropping off into steep slopes dominated by unconsolidated sediments, with submarine canyons, the relatively flat Blake Plateau, or deep Straits of Florida, depending on latitude.

The shelf edge habitat extends more or less continuously along the edge of the continental shelf at depths of 50 to 100 m (164 to 328 ft). The sediment types vary from smooth mud to areas that are characterized by great relief and heavy encrustations of coral, sponge, and other subtropical and tropical invertebrate fauna. Some of these broken bottom areas (e.g., in Onslow Bay, North Carolina) may represent the remnants of ancient reefs that existed when the sea level was lowered during the last glacial period. Fishes that generally inhabit the shelf edge zone are more tropical, such as wrasses, snappers, groupers, and porgies. Fish distribution is often patchy in this zone, with fishes aggregating over broken bottom relief in associations similar to those formed at inshore live bottom sites and are important spawning grounds for many species of managed reef fish (Sedberry et al. 2006; Schobernd and Sedberry 2009; Farmer *et al*. in prep.).

Slope

The upper slope habitat has a predominantly smooth mud bottom, but is interspersed with rocky and very coarse gravel substrates. In addition to rocky outcrops and manganese-phosphorite pavements, there are areas of rough bottom formed by iceberg scours. From North Carolina to south Florida, the retreat of the Northern Hemisphere ice sheets during the last deglaciation (20 to 6 thousand years ago) was accompanied by the discharge of meltwater and icebergs to the southeastern waters of North America, where they encountered then-shallow waters and created plow marks, rock piles and rough bottom (Hill et al. 2008, Hill and Condron 2014). Subsequent sea-level rise have submerged these features on the upper continental slope. These various rocky and mixed bottom types are where Snowy Grouper (*Hyporthodus niveatus*), Yellowedge Grouper (*H. flavolimbatum*) and tilefishes (Malacanthidae) are found (Sedberry and Schobernd 2008, Yeckley, in prep.). This habitat and its association of fishes roughly mark the transition between the faunas of the continental shelf and the slope. Depths represented by this habitat zone range from 100 to 400 m (328 to 1,312 ft), where bottom water temperatures vary from approximately 11° to 14°C (51° to 57°F). Some species inhabiting the deeper live- or hard-bottom areas may be particularly susceptible to heavy fishing pressure due to limited habitat and life history characteristics.

The continental slope off North Carolina, Georgia and Northern Florida is interrupted by the relatively flat Blake Plateau, which divides the slope into the Florida-Hatteras Slope and the Blake Escarpment. On the northern Blake Plateau are important fish habitats, including coral mounds and the Charleston Bump, an important habitat for Wreckfish.

Deep Offshore

Blake Plateau

Discontinuous large mounds of deep-sea coral reefs occur between the 360-500 m (1,181 to 1640 ft) depth contours on the Blake Plateau. While this deep coral habitat was previously described (Squires 1959; Stetson et al. 1962; Rowe and Menzies 1968), submersible dives have documented more information on their location and species composition (Popenoe and Manheim 2001; Ross 2004; Partyka *et al.* 2007 See Section 3.3.1.3). The mounds consist primarily of dense thickets of the branching ahermatypic coral *Lophelia pertusa*, although other coral species have also been identified. As coral colonies die, others form on top of the mound, and extensive coral rubble accumulates to the sides of the mound. In North Carolina, two areas of mounds have been documented off Cape Lookout and one area off Cape Fear. The vertical height of the mounds was estimated to range from 50 to 80 m over 0.4 to 1.0 km distance. Over 43 benthic or benthopelagic fish species have been identified on these coral mounds (Ross *et al*. 2004).

The Charleston Bump is a deep-water rocky bottom feature on the Blake Plateau southeast of Charleston, South Carolina (Sedberry et al. 2001). It includes a shoaling ramp and ridge/trough features on which the seafloor rises from 700 m to shallower than 400 m within a relatively short distance and at a transverse angle to both the general isobath pattern of the upper slope, and to Gulf Stream currents (Brooks and Bane, 1978). The Charleston Bump includes areas of nearly vertical, 100-200-m high rocky scarps with carbonate outcrops and overhangs; other complex bottom such as coral mounds (mostly dead coral); and flat hardbottom consisting of phosphorite-manganese pavement (Popenoe and Manheim 2001; Sedberry *et al.* 2001). The bottom relief is important to deep reef species and supports the Wreckfish (*Polyprion americanus*) (Sedberry *et al*. 1999) and pelagic longlining fisheries (Cramer 1996; Sedberry *et a*l. 2001; Cramer 2001).

The feature was first described by Brooks and Bane (1978), who noted that it deflected the Gulf Stream offshore. This deflection and the subsequent downstream eddies, gyres and upwellings may increase productivity and concentrate fishes and other organisms along thermal fronts downstream from the Charleston Bump (McGowan and Richards 1985; Dewar and Bane 1985; Haney 1986; Collins and Stender 1987; Lee et al. 1991) including the Charleston Gyre. The cyclonic Charleston Gyre is a permanent but highly variable oceanographic feature of the South Atlantic Bight induced by the deflection of rapidly moving Gulf Stream waters by the Charleston Bump. The gyre produces a large area of upwelling of nutrients, which contributes significantly to primary and secondary production within the SAB region. It is also important in retention and cross-shelf transport of larvae of reef fishes that spawn at the shelf edge (Sedberry et al. 2001). The size of the deflection and physical response in terms of replacement of surface waters with nutrient rich bottom waters from depths of 450 meters to near surface (less than 50 meters) vary with seasonal position and velocity of the Gulf Stream currents (Bane et al. 2001). The nutritional contribution of the large upwelling area to productivity of the relatively nutrient poor SAB is significant. While a lot of emphasis has been placed on shallow habitats, the South Atlantic Fishery Management Council (SAFMC 1998) designated the Charleston Gyre as an essential nursery habitat for some offshore fish species with pelagic stages, such as reef fishes, because of increased productivity that is important to ichthyoplankton (Govoni and Hare 2001; Sedberry et al. 2001).

Artificial Reefs

In addition to the natural hard or live bottom reef habitats, wrecks and other manmade structures (e.g. artificial reefs) also provide suitable substrate for the proliferation of live bottom. However, the combined area of artificial substrates will always be dwarfed compared with the total area of natural, exposed live/hardbottom. The effectiveness of artificial reefs to enhance populations has been reviewed by many researchers. The rugosity of the material, density of the reef mounds, distance to other reefs, and other factors have been tested to determine the effectiveness of

artificial reefs to enhance fish populations (DMF 1998; Strelcheck et al. 2005; Lindberg et al. 2006; Simon et al 2013; Syc and Szedlmayer 2012). In some studies, the faunal species composition on artificial reefs is similar to that identified on natural hardbottom habitat at the same depth and in the same general area (Stone et al. 1979; Stephan and Lindquist 1989; Potts and Hulbert 1994; DMF 1998). However in some studies, species richness has been reported to be higher on natural reefs (Rook et al. 1994), CPUE on natural reefs was 71-85% greater than on nearby artificial reefs (DMF 1998), and fish were in better condition or grew faster on natural reefs (Lindberg et al. 2006).
KL will update using CSA (2014) and Jordan papers.

Essential Fish Habitat

The live bottom areas constitute essential habitat for a high number of species of warm-temperate and tropical species of snappers, groupers, and associated fishes. Fautin et al. (2011) reported 1200 species of fish reported from the entire South Atlantic region, including the Florida Keys. Distinct faunal assemblages have been associated with at least four hard-bottom habitats: live/hardbottom on the open shelf; the shelf edge reef; upper slope reef; and Blake Plateau/Charleston Bump. Exploratory surveys for reef fishes has yielded 119 species representing 47 families of predominantely tropical and subtropical fishes off the coasts of North Carolina and South Carolina (Grimes et al., 1982; Lindquist et al 1989; Table 3.3-2). Parker and Dixon (1998, 2002) identified 119 species of reef fish representing 46 families during underwater surveys 44 km off Beaufort, North Carolina (Table 2.18). Off South Carolina and Georgia, 54 families, 98 genera and 128 species were taken in 83 trawl collections during winter and summer, in depths from 16-67 m (Sedberry and Van Dolah 1983). Sedberry and Schobernd (2009) reported 25 families and 54 species seen during nine shelf-edge submersible dives off Florida, Georgia and South Carolina. Three upper-slope dives yielded seven families, and seven species.

A total of 181 fish species has been reported from Gray‘s Reef National Marine Sanctuary, an inner-shelf (18-20 m) live bottom reef off Georgia (Fautin et al. 2010; J. Hare, unpublished data). A study of South Atlantic Bight reef fish communities by Chester et al. (1984) confirmed that specific reef fish communities could be identified based on the type of habitat. Bottom topography and bottom water temperatures are the two most important factors which create habitats suitable for warm-temperate and tropical species. Hardbottom habitats off mainland southeast Florida and areas off the Carolinas are often centrally placed between mid-shelf reefs to the east and estuarine habitats within inlets to the west. Therefore, they may serve as settlement habitats for immigrating larvae or as intermediate nursery habitats for juveniles emigrating out of inlets (Vare 1991; Lindeman and Snyder 1999). This cross-shelf positioning, coupled with their role as the only natural structures in these areas, suggests nearshore hardbottom can represent important Essential Fish Habitat.

Section 600.815 (a) (9) of the final rule on essential fish habitat determinations recognizes that subunits of EFH can be of particular concern. Such areas, termed Essential Fish Habitat-Habitat Areas of Particular Concern (EFH-HAPCs), can be identified using four criteria from the rule: a) importance of ecological functions; b) sensitivity to human degradation; c) probability and extent of effects from development activities; and d) rarity of the habitat. Applications of EFH and EFH-HAPCs in the management of the SAFMC snapper-grouper complex was examined in Lindeman et al (2000), with a focus on developmental variation and MPAs.

Hardbottom habitat types which have been identified as EFH-HAPCs include the following areas.

Charleston Bump and Gyre

The South Atlantic Bight, the Charleston Bump and Gyre are described in greater detail in several research and review papers (e.g., Bane et al. 2001; Sedberry et al. 2001; Govoni and Hare 2001 and papers cited therein). The following synopsis is based on the review by Sedberry et al. (2001), Fautin et al. (2010) and O. Pashuk (unpublished MS).

In general, the Gulf Stream flows along the shelf break, with very little meandering, from Florida to about 32° N latitude where it encounters the Charleston Bump and is deflected seaward forming a large offshore meander. The cyclonic Charleston Gyre is formed, with a large upwelling of nutrient-rich deep water in its cold core. The Charleston Bump is the underwater ridge/trough feature located southeast of Charleston, South Carolina, where seafloor rises from 700 to 300 m within a relatively short distance and at a transverse angle to both the general isobaths pattern of the upper slope, and to Gulf Stream currents. Downstream of the Charleston Bump, enlarged wavelike meanders can displace the Gulf Stream front up to 150 km from the shelf break. These meanders can be easily seen in satellite images.

Although two to three large meanders and eddies can form downstream of the Bump, the Charleston Gyre is the largest and the most prominent feature. The consistent upwelling of nutrient-rich deep water from the depths over 450 m to the near-surface layer (less than 50 m) is the main steady source of nutrients near the shelf break within the entire South Atlantic Bight, and it contributes significantly to primary and secondary production in the region. The Charleston Gyre is considered an essential nursery habitat for some offshore fish species with pelagic stages. It is also implicated in retention of fish eggs and larvae and their transport onshore.

The Charleston Bump and the Gyre can also create suitable habitats for adult fish. For example, the highest relief of the Bump is the only known spawning location of the Wreckfish. The Charleston Gyre may be also beneficial to other demersal species of the Snapper-Grouper complex, as well as to pelagic migratory fishes, due to food availability and unique patterns of the currents in this area.

Ten Fathom Ledge and Big Rock

The Ten Fathom Ledge and Big Rock areas are hard-bottom habitats located south of Cape Lookout, North Carolina. The Ten Fathom Ledge is located at 34° 11‘ N. and 76° 07‘ W. in 95 to 120 meter depth on the Continental Shelf in Onslow Bay, North Carolina, beginning along the southern edge of Cape Lookout Shoals. This area encompasses numerous patch reefs of coral-algal-sponge growth on rock outcroppings distributed over 136 square miles of ocean floor. The substrate consists of oolithic calcarenites and coquina forming a thin veneer over the underlying Yorktown formation of silty sands, clays, and calcareous quartz sandstones.

The Big Rock area encompasses 36 square miles of deep drowned reef around the 50-100 meter isobath on the outer shelf and upper slope approximately 36 miles south of Cape Lookout. Hard substrates at the Big Rock area are predominately algal limestone and calcareous sandstone. Unique bottom topography at both sites produces oases of productive bottom relief with diverse and productive epifaunal and algal communities surrounded by a generally monotonous and relatively unproductive sand bottom. Approximately 150 species of reef-associated species have been documented from the two sites (R. Parker, unpublished data.).

Shelf Break Area from Florida to North Carolina

Although the area of bottom between 100 and 300 meters depths from Cape Hatteras to Cape Canaveral is small relative to the more inshore live bottom shelf habitat as a whole, it constitutes essential fish habitat for deep-water reef fish. A series of troughs and terraces are composed of bioeroded limestone and carbonate sandstone (Newton et al. 1971), and exhibit vertical relief ranging from less than half a meter to more than 10 meters. Ledge systems formed by rock outcrops and piles of irregularly sized boulders are common.

Overall, the deep-water reef fish community likely consists of fewer than 60 species; however, many fishery species spawn there (Sedberry et al. 2006). Parker and Ross (1986) observed 34 species of deep-water reef fishes representing 17 families from submersible operations off North Carolina in waters 98 to 152 meters deep. In another submersible operation in the Charleston Bump area off South Carolina, Gutherz et al. (1995) describe sightings of 27 species of deep-water reef fish in waters 185 to 220 meters in depth. Schobernd and Sedberry (2009) reported 25 families and 54 species seen during nine shelf-edge submersible dives off Florida, Georgia and South Carolina. Three upper-slope dives yielded seven families, and seven species.

Gray's Reef National Marine Sanctuary

Gray's Reef National Marine Sanctuary (GRNMS) is located 17.5 nautical miles east of Sapelo Island, Georgia, and 35 nautical miles northeast of Brunswick, Georgia. Gray's Reef encompasses nearly 32 km^2 at a depth of about 22 meters (Parker et al. 1994). The Sanctuary contains extensive, but patchy hardbottoms of moderate relief (up to 2 meters). Rock outcrops, in the form of ledges, are often separated by wide expanses of sand, and are subject to weathering, shifting sediments, and slumping, which create a complex habitat including caves, burrows, troughs, and overhangs (Hunt 1974). Parker et al. (1994) described the habitat preference of 66 species of reef fish distributed over five different habitat types. Numbers of species and fish densities were highest on the ledge habitat, intermediate on live bottom, and lowest over sand. Kendall et al. (2008) found that presence of dominant groupers, Gag and Scamp, was most strongly related to height of ledge undercut, whereas abundance of Black Sea Bass was best explained by percent cover of sessile biota. A designated research area was created within the sanctuary boundary in 2010 to potentially evaluate the effects of fishing, natural events and cycles, and climate change.

Nearshore Hardbottom of Mainland East Florida

Extending semi-continuously from at least St. Augustine Cape Canaveral to the Florida Keys, nearshore hardbottom was evaluated in terms of the four HAPC criteria in Section 600.815 of the final EFH interim rule. In terms of ecological function, several lines of evidence suggest that nearshore hardbottom reefs may serve as nursery habitat. The most recent summary information on NHB in mainland east Florida is within CSA (2009). The following is based on the quantitative information available for the southeast Florida mainland (Lindeman 1997a and b; Lindeman and Snyder 1999; Baron et al. 2004), which also included life stage-specific abundance data. First, pooled early life stages consistently represented over 80% of the total individuals at all sites censused. Second, eight of the top ten most abundant species were consistently represented by early stages. Third, use of hardbottom habitats was recorded for newly settled stages of more than 20 species.

The mere presence of more juvenile stages than adults does not guarantee a habitat is a valuable nursery. Rapid decays in the benthic or planktonic survival of early stages of marine fishes are common demographic patterns (Shulman and Ogden 1987; Richards and Lindeman 1987), ensuring that if distributions are homogeneous, all habitats will have more early stages than adults. The high numbers of early stages on nearshore reefs appear to reflect more than just larger initial numbers of young individuals. Newly settled stages of most species of grunts and eight of nine species of snappers of the southeast mainland Florida shelf have been recorded primarily in depths less than five meters, despite substantial sampling efforts in deeper waters. Adults are infrequent or absent from the same shallow habitats. There is habitat segregation among life stages, with the earliest stages using the shallowest habitats in many species of grunts and snappers (Starck 1970; Dennis 1992; Lindeman et al. 1998). Similar ontogenetic differences in both distribution and abundance exist for many other taxa which utilize nearshore hardbottom habitats. Based on this and other evidence, Lindeman and Snyder (1999) concluded that at least 35 species utilize nearshore hardbottom as a primary or secondary nursery area. At least ten of these species are managed under the Snapper/Grouper FMP.

Because nearshore areas are relatively featureless expanses of sand in the absence of hardbottom, such structures may also have substantial value as reference points for spawning activities of inshore fishes. Many fishes require three-dimensional structure as a reference point for coarse-scale aggregation and fine-scale behavior during spawning (Thresher 1984). Using information from the literature, personal observations, and discussions with commercial fishermen, 15 species were estimated to spawn on nearshore reefs (Lindeman 1997a). An additional 20 species may also spawn on or near these reefs. Some are of substantial economic value; these include snook, pompano, and several herring species. At least 90 species known to associate with nearshore hardbottom structures occur in south Florida. The majority of these species are represented primarily by early life stages. Approximately 51 species are of recreational value and thirty species are of commercial value. Twenty-two species are utilized for bait and 21 species are marketed within the aquaria industry.

Based on the demonstrated or potential value of these areas as nurseries and spawning sites for many economically valuable species, nearshore hardbottom habitats were estimated to support highly important ecological functions, the first HAPC criterion. The second and third HAPC criteria, sensitivity and probability of anthropogenic stressors, are interrelated in terms of nearshore hardbottom. They are treated collectively here. Various stretches of nearshore hardbottom have been completely buried by dredging projects associated with beach management activities in this subregion (Section 7.4.2.2). They may also be subjected to indirect stressors over both short and long time scales from such projects. For example, between 1995 and 1998, up to 19 acres of nearshore hardbottom reefs were buried by beach dredging projects at two sites in Palm Beach County. Such activities occur within other counties of this subregion as well. The 50-year planning document for beach management in southeast mainland Florida (ACOE 1996), includes beach dredge-fill projects for over fifteen areas, with renourishment intervals averaging 6-8 years. Given the past and projected future, it is concluded that both the sensitivity of these habitats and the probability of anthropogenic stressors is high.

In terms of the final EFH-HAPC criterion, rarity, nearshore hardbottom also ranks high. In southeast mainland Florida, most shorelines between Dade and Broward Counties (25°30'-26°20' N) lack natural nearshore hardbottom with substantial three-dimensional structure (ACOE 1996). Although substantial stretches of nearshore hardbottom exist in portions of Palm Beach, Martin, St. Lucie, and Indian River Counties (Perkins et al. 1997) (26°20'-27°15' N) these reefs are often separated by kilometers of barren stretches of sand. Offshore, most mid-shelf areas (5-20 m) are also dominated by expanses of sand despite the variable occurrence of several mid-shelf reef lines. Therefore, there are no natural habitats in the same or adjacent nearshore areas that can support equivalent abundances of early life stages. Absences of nursery structure can logically result in increased predation and lowered growth. In newly settled and juvenile stages, such conditions could create demographic bottlenecks that ultimately result in lowered local population sizes.

Nursery usage of nearshore hardbottom reefs may be a bi-directional phenomenon. Many species utilize these habitats during both newly settled and older juvenile life stages. This suggests that nearshore hardbottom can facilitate both inshore and offshore migrations during differing ontogenetic stages of some species. Their limited availability does not necessarily decrease their value. When present, they may serve a primary nursery role as shelter for incoming early life stages which would undergo increased predation mortality without substantial habitat structure. In addition, some species use these structures as resident nurseries; settling, growing-out, and maturing sexually as permanent residents (e. g., pomacentrids, labrisomids). A secondary nursery role may result from increased growth because of higher food availabilities in structure-rich environments. Nearshore hardbottom may also serve as secondary nursery habitat for juveniles that emigrate out of inlets towards offshore reefs. This pattern is seen in gray snapper and blue striped grunt which typically settle inside inlets and primarily use nearshore hardbottom as older juveniles (Lindeman et al. 1998; CSA 2009).

In summary, nearshore hardbottom habitats of southeast Florida ranked high in terms of ecological function, sensitivity, probability of stressor introduction, and rarity. Based on the criteria in Section 600.815 (a) (9), it is concluded that they represent Essential Fish Habitat-Habitat Areas of Particular Concern for species managed under the Snapper/Grouper Fishery Management Plan and dozens of other species which co-occur with many species in this management unit. Many of these other species, not currently managed under the SAFMC are important prey items (Randall, 1968) for those species under management.

End of New Edits

SAFMC Fishery Ecosystem Plan: Estuarine Emergent Habitat
3.2 Estuarine/inshore systems
3.2.1 Estuarine Emergent (salt marsh and brackish marsh)

Description and Distribution
One of the dominant features of the Coastal Plain of the southeastern U.S. is its extensive saltmarshes. Saltmarshes are transitional areas between land and water, occurring along the intertidal estuarine shorelines where salinity ranges from near ocean strength to near fresh in upriver marshes. The saltmarsh is a type of wetland. Wetlands are classified on the basis of their hydrology, vegetation, and substrate. The most widely used classification system, that proposed by Cowardin et al. (1979), classifies wetlands into five ecological systems, one of which is the Estuarine System. The Estuarine System is further divided into the Subtidal and Intertidal subsystems. Emergent Wetland is one of eight classes of wetlands within the Estuarine Intertidal Subsystem. Estuarine emergent wetlands are characterized by the presence of erect, rooted, herbaceous hydrophytes dominated by salt-tolerant perennial plants. In the southeastern U.S., saltmarsh cordgrass (*Spartina alterniflora*), saltmeadow cordgrass (*S. patens*), big cordgrass (*S. cynosuroides*), needlerush (*Juncus roemerianus*), salt grass (*Distichlis spicata*) and narrow-leaved cattail (*Typha angustifolia*) are major components of the estuarine emergent plant community.

In this section, the term saltmarsh encompasses brackish marsh, as well. Although there is no clear distinction between the commonly used terms saltmarsh and brackish marsh, the latter typically refers to estuarine emergent wetlands with salinities near the lower end of the mixohaline range, which includes oligohaline (0.5-5.0 ppt), mesohaline (5.0-18.0 ppt), and polyhaline (18.0-30.0 ppt) salinity regimes. By contrast, saltmarshes can also occur in salinity regimes that are fully marine or euhaline (30.0-40.0 ppt), as well as in hyperhaline (>40 ppt) environments. Characteristic plant species vary along a continuum from high salinity saltmarshes, which are typically dominated by *S. alterniflora* in the southeast, to lower salinity brackish marshes,‖ where species such as *S. cynosuroides* and *J. romerianus* achieve greater dominance. Because tidal brackish marshes are transitional areas between saltmarshes and tidal freshwater marshes, brackish marshes include species from both habitats, and, therefore, have relatively high plant diversity.

Saltmarshes occur in each of the states in the South Atlantic Region. The total area of saltmarshes in this region is approximately 894,200 acres (Field et al. 1991). It is estimated that saltmarshes in the South Atlantic account for 21% of the nation‘s total salt marshes (Field et al. 1992). Unlike the Gulf Coast states, particularly Louisiana, which have lost thousands of acres of estuarine emergent marsh due to a variety of causes including erosion, saltwater intrusion, subsidence sea-level rise, sediment deprivation and physical alteration, the acreage of estuarine emergent marsh throughout the remainder of the southeastern U.S. has remained relatively stable from the mid-1970s to mid-1980s (Hefner et al. 1994).

Comment [1]: is this relevant? Seems like an outdated reference. If there is an updated reference to this, I believe is is applicable.

In the southeastern U.S., South Carolina has the greatest saltmarsh acreage (365,900 acres), followed by North Carolina (212,800 acres) and Georgia (351,236 acres). Florida (east coast) has the least saltmarsh acreage (106,000 acres). The Albemarle-Pamlico Sound (NC) and the St. Andrews-Simons Sounds are the estuarine drainage areas (EDA) with the greatest marsh habitat.

Note--There needs to be a better assessment of emergent marsh totals for the different states within the SAFMC area. It is hard to manage something you do not have relatively accurate data on how much it is you have and what kind it is. There is also a need to better assess the managed species that rely on these habitats at some point in their life-histories and the forage species they rely upon, better linking salt marshes to EFH. Here and in other sections it is necessary to partner with Federal Agencies, each state, with non-government organizations (NGO's), and NOAA Sea Grant to obtain funds to acquire this fundamental information. This should be a priority issue for the Council.

Table 3.2-1 presents baseline estimates of coastal wetland acreage by estuarine drainage area in the South Atlantic region compiled through a cooperative effort of NOAA and USFWS (NOAA, 1991a). Figure 3.2-1 shows the estuarine drainage areas in the South Atlantic Region for which the estimates have been compiled.

Table 3.2-1. Coastal wetlands by estuarine drainage area in the South Atlantic (Source: NOAA 1991a).

(Acres X 100)

Estuarine Drainage Salt Marshb Fresh Marshb Forested and Scrubb Tidal Flatsb Totalb

Areaa

1 Albemarle/Pamlico Sounds (8) 1,576 (14) 365 (3) 9,062 (80) 311 (3) 11,314

2 Bogue Sound (65) 211 (22) 11 (1) 616 (64) 118 (12) 956

3 New River (46) 41 (16) 5 (2) 203 (81) 45 (1) 252

4 Cape Fear River (13) 90 (6) 97 (6) 1,291 (86) 20(1) 1,498

5 Winyah Bay (30) 124 (2) 308 (5) 5,472 (93) 6 (0) 5,910

6 North and

South Santee Rivers (88) 129 (7) 174 (9) 1,613 (84) 1 (0) 1,916

7 Charleston Harbor (10) 268 (14) 169 (9) 1,540 (78) 8 (0) 1,985

8 St. Helena Sound (100) 916 (21) 321 (7) 3,036 (71) 25 (1) 4,299

10 Savannah Sound (100) 322 (11) 141 (5) 2,428 (84) 9 (0) 2,900

11 Ossabaw Sound (82) 245 (10) 40 (2) 2,282 (89) 4 (0) 2,571

12 St. Catherine's/

Sapelo Sounds (29) 352 (40) 46 (5) 461 (53) 13 (2) 872
13 Altamaha River (35) 79 (7) 81 (7) 976 (86) 2 (0) 1,138
14 St. Andrews/
Simmons Sounds (66) 1,134 (20) 157 (3) 4,420 (77) 59 (1) 5,771
15 St. Marys R./Cumberland Sound N/A N/A N/A N/A N/A
16 St. Johns River (96) 168 (2) 2,646 (25) 7,665 (73) 2 (0) 10,481
17 Indian River (95) 24 (2) 591 (57) 368 (36) 45 (4) 1,028
18 Biscayne Bay (79) 104 (3) 1,556 (41) 2,059 (55) 49 (1) 3,769
South Atlantic Total 6,666 (11) 6,743 (11) 44,615 (76) 747 (1) 58,770
a. Values in parentheses represent the percent of county grid sampled by NOAA. Areas with less than 100 percent coverage may not be completely mapped by the U. S. Fish and Wildlife Service.
b. Values in parentheses represent the percent of total Estuarine Drainage Area wetlands grid sampled by NOAA.

Saltmarshes occur in the intertidal zone in coastal and estuarine waters. The coastal physiography of the northern and southern part of the South Atlantic Bight (e.g. North Carolina and Florida) is dominated by shallow water lagoons behind sand coastal barrier shoreline. In the central portion (e.g. South Carolina and Georgia) there are depositional marsh-filled lagoons. In both of these systems, marshes may occur in vast expanses, in narrow fringing bands, or as small pocket marshes‖ interspersed among higher elevation areas. Although marshes may develop in sandy sediments, especially in high-energy areas, marsh development typically leads to sediments with fine particle-size (mud) and high organic matter content. In most physical settings, marshes can accrete sediments, and thus maintain their elevation in relation to the rising sea level . Salt marshes persist longest in low-energy protected areas where the rate of sediment accretion is greater than or equal to the rate of subsidence (Mitsch and Gosselink 1986).

Figure 3.2-1. Estuarine drainage areas in the South Atlantic Region (Source: NOAA 1991a).

Ecological Role and Function

Structure and function of a saltmarsh are influenced by marsh habitat size, landscape setting, tide, salinity, nutrients and temperature. The saltmarsh can be a stressful environment to plants and animals, with rapid changes occurring in these abiotic variables (Gosselink 1980; Gosselink et al. 1974). Although species diversity may vary widely among salt marshes , the saltmarsh is one of the most biologically productive ecosystems in the world (Teal 1962; Teal and Teal 1969). The high primary productivity that occurs in the marsh, and the transfer of detritus throughout the estuary from the marsh, provides the base of the food chain supporting many marine organisms.

Comment [2]: Focus more on valuation of the ecosystem services provided by key characteristics of emergent wetlands – characteristics that can be replicated by restoration/mitigation projects, assuming the FEP is about more than definite EFH and HAPC. Conservation recommendations resulting from EFH consultation sometime include mitigation based on valuation of habitat functions.

Few aquatic species feed directly on living plant tissue in salt/brackish marsh (i.e., periwinkle), and their productivity is very low compared to that of detritivores and consumers of microalgae (Wiegert and Freeman 1990; Steel 1991; SAFMC 1998a). However, biotic interactions with primary consumers can result in degradation or loss of wetlands. Recent study results from the southeastern United States suggest that blue crab predation on snails may prevent the snail from overgrazing the marsh grass (Silliman and Bertness 2002).

Detrital and bacterial production from salt/brackish marsh exhibits some of the highest recorded values per unit area of any ecosystem in the world (Wiegert and Evans 1967). Slow-moving or sessile species residing in salt/brackish marsh and contributing to secondary production include fiddler crabs, mud snails, amphipods, oysters, clams, and ribbed mussels (Wiegert and Freeman 1990). Based on data from Georgia marshes, biomass of these resident species exceeded 15 g carbon/m2, and consisted of 80-200 fiddler crabs, 400-700 periwinkle snails or mud snails, and 7-8 mussels (Wiegert and Freeman 1990). The resident estuarine fishes (i.e., killifish, grass shrimp, sheepshead minnow) are an important link between estuarine production and transient predatory fish populations (Wiegert and Freeman 1990; Kneib 1997). Salt-brackish marsh edge also provides important feeding areas for blue crabs, red drum, flounder, seatrout and other large predators searching the edge of complex structure near deeper water, as illustrated by greater predation on grass shrimp with increasing depth in shallow-estuarine water (Clark et al. 2003).

It has been estimated that 45% of salt marsh production is exported to the estuarine system in the form of detritus, dissolved organic matter, and transient nekton (i.e., grass shrimp and killifish; Teal 1962). The biomass of secondary production going in and out with the tide (fish, shrimp) is less well known than resident species biomass (Kneib and Wagner 1994). The exported production of brown and white shrimp is probably the best known and most significant to coastal fisheries (Turner 1977; Wiegert and Freeman 1990). The estimated yield of shrimp from North Carolina was 107 lb per acre of intertidal vegetated bottom (Turner 1977), where intertidal vegetation included salt marsh macrophytes, *Spartina* spp. [and] *Juncus* spp. However, recent research suggests that wetlands vary greatly in their role as exporters or importers of organic matter (Wiegert and Freeman 1990). This variation could be the result of variable erosion or deposition rates among seasons or wetland areas.

Primary production in salt/brackish marshes is converted into fish production through several pathways. Using sulfur, carbon, and nitrogen isotopes to trace organic matter flow in the salt marsh estuaries of Sapelo Island, Georgia, Peterson and Howarth (1987) found two major sources of organic matter used in fish production: *Spartina* (detritus) and algae. The relative importance of each source is determined by the feeding mode, size, location, and trophic position of the marsh and estuarine consumers (Peterson and Howarth 1987). For example, benthic microalgae probably support herbivorous snails, whereas detritus supports sheepshead minnows, mummichogs, and their prey. Attached algae can be found on the marsh grass itself, the intertidal mudflats, and the shallow subtidal bottom near the marsh. Pinckney and Zingmark (1993) compared production rates of benthic microalgae in various bottom types in an estuarine system (North Inlet, South Carolina). Short *Spartina* marsh accounted for the greatest amount of microalgal productivity (44.6%) in the system, followed by intertidal mudflats (22%), tall *Spartina* marsh (18%), and shallow subtidal bottom (<1 m mean low water) (13%). Sand flats accounted for only 3% of the total annual microalgal production (Pinckney and Zingmark 1993). Many saltmarshes are drained by an intricate network of tidal creeks. These creeks and the adjacent marsh function as nursery areas for larval and juvenile finfish, crustaceans, and mollusks, and as a critical fisheries habitat to adult species. Greater than 95% of the commercial species in the United States are estuarine dependent species (Feierabend and Zelazny 1987 as cited in Mitsch and Gosselink 1993). Most of the juveniles of fishery species found in salt/brackish marsh nurseries were spawned offshore during winter. The larvae were transported through inlets and into estuarine waters where they settled in the upper (low salinity) or lowermost (high salinity) reaches of estuarine creek systems (Ross 2003). The peak of juvenile settlement generally occurs in spring through early summer, although the peak is correlated more with water temperature (Ross and Epperly 1985). Settlement in upper reaches is particularly beneficial to spot and croaker, where growth and survivorship are enhanced compared to lower reaches (Ross 2003). If movement to general regions of the estuary is largely passive (Pietrafesa et al. 1986; Pietrafesa and Janowitz 1988), the viability of spot and croaker stocks could be reduced by hydrodynamic conditions resulting in more settlement to lower regions of the estuary (Ross 2003). This settlement pattern could also occur in other estuarine-dependent species.

The marsh not only provides food, structure, and refuge from predators to fishery organisms, but also regulates the amount of freshwater, nutrient and sediment inputs into the estuary. In addition to its function as an essential fisheries habitat, the marsh plays a vital role in the health and water quality of the estuary. The position of saltmarshes along the margins of estuaries and their dense stands of persistent plants make them valuable for stabilizing the shoreline and for storing floodwaters during coastal storms.

Species composition and community structure

Flora

There are more than one hundred species of vascular flora and algae that compose the various intertidal macrophytic communities that are common to the estuaries of the South Atlantic Bight (SAB) (Beccasio et al. 1980). Most of those communities are tidally influenced marshes and, to a lesser degree, tidally influenced shrub and forest communities. South of the St. John River estuary in northern Florida the wetland communities of the lagoonal estuaries of the lower Florida peninsula gradually change from a marsh dominated landscape to a shrub community dominated by mangroves.

Comment [3]: Climate change is enabling mangroves to expand north and causing a gradual transition of saltmarsh to mangroves.

The macrophytes identified in this section are all influenced in their growth characteristics by salinity in the water. Salinities in south Atlantic estuaries generally range from 30.0 ppt or above (essentially sea strength) at the mouths of coastal inlets to less than 0.5 ppt at the upper reaches of the estuaries under the influence of freshwater outflow from coastal plain streams and rivers (Odum et al. 1984).

The tolerance of salinity in the water column and in the soils that serve as substrate directly influence the composition of the plant community. Salinity in combination with the periodicity of inundation due to tidal action and downstream discharge, soil chemistry, soil type, shading and erosion all result in a predictable model of the zonation of individual species and, at times, discrete plant communities. Because salt marshes in the southeastern U.S. are influenced by the twice daily rise and fall of tides, they can be subject to rapid changes in salinity, temperature and water depth. Salinity, flood frequency and extent, marsh size and landscape setting can influence the types and densities of flora and fauna occuring in the salt marshes. The low marsh zone typically floods twice daily, while the high marsh floods only during storms and unusually high tides. One plant species, *S. alterniflora*, dominates the regularly flooded low marsh. *S. alterniflora* is the most abundant plant in southeastern marshes and is responsible for much of the marsh's productivity. *S. alterniflora* is able to tolerate salinities from sea strength to freshwater, as well as the saturated soils that are characteristic of twice-daily tidal inundation. *S. alterniflora*, a true grass, commonly occurs in vast stands growing on the fine grained soils that have been deposited in the low energy coastal lagoons and drowned river valleys behind the barrier islands that fringe the oceanic shoreline. Within the vertical zonation of the tidal amplitude *S. alterniflora* occurs from an elevation that generally equates to mean tide level up to mean high water. *S. alterniflora* exhibits three growth forms, tall, medium and short. The tall form dominates the immediate shorelines of the tidal stream banks at an elevation from mean tide level up to slightly below the mean high tide level and to a horizontal depth shoreward of about two meters. The stem height commonly attains one to one and a half meters. The medium form is found from the stream side levee horizontally into the interior of the marsh. Stem density is less dense that the tall form and stem height averages up to about one meter. The short form grows in the interior portion of the marsh where sediments are finer and less well-drained. Stem density can be higher than the medium growth form and stem height averages about 0.2 to 0.3 meters or shorter. This growth pattern is attributed to a combination of periodicity of tidal inundation, soil salinity, soil saturation, nutrient availability and other less predictable factors. The zonation and stem density, however, play a key role in the use of *Spartina* marshes by consumer organisms.

The second most common marsh plant that occurs in the region is *J. roemerianus*, like *S. alterniflora*, is found in all of the estuaries of the ? South Atlantic Basin?. Less salt tolerant and

Comment [4]: in

Comment [5]: than in

Comment [6]: needs to be finalized

not as well adapted to longer periods of inundation as *S. alterniflora*, *J. roemerianus* is found in the higher elevations of tidal coastal marshes. In salinity regimes higher that 15 ppt *J. roemerianus* is found in dense monospecific stands often in a zone between the *Spartina* and high ground. Stem height averages one meter but may approach two meters.

Comment [7]: At least in Florida, the transition zone from Spartina to Juncus is also accompanied by Cladium jamaicense (Jamaica Swamp Sawgrass), Acrostichum danaefolium (Giant Leatherfern), Typha spp. cattails) and Phragmites (

Diversity of the vascular plant community increases at higher tide elevations and at lower salinities. In the outer portions of the estuary, *S. patens* or saltmeadow cordgrass, occurs between mean high water and spring high water. Other plants characteristic of the high marsh are *Salicornia virginica* and *Distichlis spicata*. In more brackish portions of the estuary, *S. alterniflora* is replaced by *S. cynosuroides* and *Scrirpus olneyii*.

Several species of macroalgae may become abundant within salt marsh tidal creeks and on the marsh surface, particularly in early spring. These include *Ulva*, *Codium*, *Gracilaria* and *Enteromorpha*. These macroalgal communities, although ephemeral, can provide both refuge and food resources to marsh consumer organisms. Additionally, a diverse community of benthic and epiphytic microalgae inhabits the marsh surface and the stems of marsh plants. This community is composed of diatoms, cyanobacteria, and photosynthetic bacteria, and may represent a significant portion of marsh primary production. The primary production of this algal community also plays an important role in supporting fisheries production in salt marsh habitats.

Fauna

Estuarine intertidal marshes provide habitat for Council-managed species, other fish, shellfish, and invertebrates, as well as endangered and threatened species, furbearers and other mammals, waterfowl, wading birds, shorebirds and other birds, and reptiles and amphibians. Beyond the estuaries, exported marsh nutrients, detritus, and prey species contained in the food web ultimately add to the ecosystems supporting additional managed species such as coastal migratory pelagics (i.e., mackerels) and species in the snapper grouper complex.

In contrast to freshwater marshes, salt marshes typically have low species diversity of the higher vertebrates, higher species diversity of invertebrates. The invertebrate community in salt marshes is composed of various macrofaunal and mesofaunal species. The macrofaunal community is dominated by various species of crabs (e.g., fiddler and blue crabs), gastropod molluscs (such as *Littorina irrorata*), polychaetes, and amphipods. The protection afforded by marsh grass stem structure and the abundant food supply of salt marshes make them important nursery habitats for larval and juvenile stages of decapod species such as blue crab (*Callinectes sapidus*), white shrimp (*Penaeus setiferus*), and grass shrimp (*Palaemonetes* spp.). Subadult stages move into intertidal marshes along the creek edge on incoming tides and penetrate the interior marshes during flood tide (Kneib and Wagner 1994). Resident species such as fiddler crabs (*Uca* spp.)

burrow preferentially in sediments with intermediate densities of *Spartina* root mats (Bertness and Miller 1984). *Uca* spp. and *Palaemonetes* spp. are important prey of piscine, avian, and mammalian marsh inhabitants. These are the primary foragers of marsh vegetation, detritus, and mesofauna. The mesofaunal community consists of protozoa, nematodes, copepods, annelids, and rotifers. These organisms primarily feed on the microbial population, which chiefly consists of various species of bacteria and fungi. *S. alterniflora* supports a large number of epiphytic fungi, which not only contribute carbon and nutrients, but also participate in decomposition of standing biomass.

Table 3.2-2 reviews examples of fishes and crustaceans common to southeastern U.S. marshes. These organisms utilize the marsh structure (including the stems of emergent vascular plants, attached macroalgae, substrate materials such as shells and sediments, attached living oysters and mussels, residual tidal pools, and accumulated woody flotsam). Some feed directly on the vegetation, especially decapods and gastropods. Some species, are not found within the marsh, but derive substantial food resources from marsh plants as detritus.

Table 3.2-2. List of select macrofaunal species observed in collections from some marsh habitats located in the southeastern United States (Source: NMFS, 1998).

Species Common Name Resident Status Macrophyte Genera Fisheries Value

FISH

Anchoa spp. anchovy M Sp, Sc, Ty P

Anguilla rostrata American eel M Sp, Ju C/P

Archosargus probatocephalus sheepshead M Sp R/C/P

Bairdiella chrysoura silver perch M Sp, Sc, Ty, Ju R/P

Brevootia tyrannus Atlantic menhaden M Sp, Sc, Ty R/C/P

Cynoscion nebulosus spotted seatrout M Sp, Ju R/C/P

Cyprinodon variegatus sheepshead minnow R Sp, Ju P

Dorosoma cepedianum gizzard shad F Sc, Ty C/P

Eucinostomus sp. mojarra M Sp, Sc, Ty, Ju P

Fundulus spp. killifish R Sp, Sc, Ty, Ju R/P

Gambusia affinus mosquito fish R Sc, Ty, Ju P

Gobiidae gobies R Sp, Sc, Ty, Ju P

Ictalurus catus white catfish F Sc, Ty R/C/P

Lagodon rhomboides pinfish M Sp, Sc, Ty, Ju R/P

Leiostomus xanthurus spot M Sp, Sc, Ty, Ju R/C/P
Lepomis gibbosus pumpkinseed F Sc, Ty R/P
Lutjanus griseus gray snapper M Sp R/C/P
Lutjanus synagris lane snapper M Sp R/C/P
Lucainia parva rainwater killifish R Sp, Ju P
Menidia spp. silversides R Sp, Sc, Ty, Ju P
Micropogonias undulatus Atlantic croaker M Sc, Ty R/C/P
Micropterus salmoides largemouth bass F Sc, Ty R/C/P
Morone saxatilis striped bass F Sp, Sc,Ty R/C/P
Mugil spp. mullet M Sp, Sc, Ty, Ju R/P
Orthopristis chrysoptera pigfish M Sp R/P
Paralichthys spp. flounder M Sp, Sc, Ty, Ju R/C/P
Pogonias cromis black drum M Sp R/C/P
Pomatomus saltatrix bluefish M Sp, Sc, Ty R/C/P
Pomoxis nigromaculatus black crappie F Sc, Ty R/C/P
Sciaenops ocellatus red drum M Sp R/C/P
Sphyraena barracuda great barracuda M Sp R/P
Symphurus plagiusa black cheek tonguefish M Sp P
Urophycis spp. hake M Sp R/C/P
DECAPODS ---
Callinectes sapidus blue crab M Sp, Sc, Ty, Ju R/C/P
Menippe mercenaria stone crab R Sp R/C/P
Palaemonetes spp. grass shrimp R Sp, Sc, Ty, Ju P
Penaeus spp. penaeid shrimp M Sp, Sc, Ty, Ju R/C/P
Uca spp. fiddler crabs R Sp, Ju R/C/P
Letter codes for the Resident Status heading are R = resident, M = transient (marine spawner), F = transient (freshwater spawner); for the Macrophyte Genera heading are Sp = *Spartina* spp., Sc = *Scirpus* sp., Ty = *Typha* spp., Ju = *Juncus* spp.; and for the Fisheries Value heading are R = recreational, C = commercial, P = prey species.

The protection afforded by the stem structure and intertidal water levels provides spawning habitat for some fish species, such as killifish, atherinids and gobiids, but most fishes associated with the marsh are recruited as larvae or early juveniles (Boesch and Turner 1984). Taxa spawning in or near the marsh are considered residents, but the most of the fish species (but not necessarily most of the biomass) are seasonally transient (Weinstein 1979). Transients spawn elsewhere, either upstream in freshwater (e.g., striped bass), or downstream in the coastal waters (e.g., flounders) (Schreiber and Gill 1995), and occupy the marsh habitat primarily as juveniles in the warmer months. Some of these species do not penetrate into the marsh, but are strongly linked to it in the adjacent fringing water.

Marshes as Essential Fish Habitat

It is estimated that over 95% of the finfish and shellfish species harvested commercially in the United States are wetland-dependent (Feierabend and Zelanzy 1987). Coastal wetlands are implicated when you consider that a large majority of commercial fishing occurs in estuarine and marine systems. Within the coastal wetlands category, there are a relative small number of anadromous species that are dependent on riverine forested wetlands for spawning and nursery habitat rather than estuarine marsh. But they only account for a small fraction of species in the commercial catch. The vast majority of finfish and shellfish could thus be considered dependent on estuarine wetlands.

The detritus and attached microalgae made available to secondary consumers by the presence of marsh grass forms the contribution of estuarine marsh production to commercial fisheries production. However, the environment creating individual salt marshes can differ such that more or less production is exported and available for consumption. Species associated with adjacent mud flats and channels benefit more from the presence of marsh plants as more production is exported. There are also species that use marsh grass more directly as refuge and/or foraging areas. Of all the SAFMC managed species, red drum and shrimp are considered most dependent on salt marsh habitat (SAFMC 1998).

Turner (1977) demonstrated the association between shrimp and intertidal habitat (defined as salt marsh or mangroves) at a regional scale. The study compared the commercial harvest of shrimp in various locations with areal estimates of salt/brackish marsh coverage. The results indicated a strong correlation between shrimp yield and area of estuarine vegetation, with little correlation between yield and estuarine area, average depth, and volume. The relationship between shrimp harvest (y) and area of estuarine marsh (e) was quantified in the following equation (where x is degrees latitude):

Y = 159e-0.070(x)

However, it should be noted that annual shrimp abundance is highly dependent on weather conditions, in addition to fishing mortality and habitat changes (??? shrimp management plan – draft 2005).

The relationship between red drum production and estuarine marsh areas has not been quantified to the same level as that of shrimp. Juvenile red drum are found year-round over a wide array of salinity and habitats, although they seem to prefer sheltered, nearshore areas of coastal rivers and submerged aquatic vegetation (SAV) growing near marsh grass behind barrier islands (Ross and

Stevens 1992). However, there is substantial evidence for the association of red drum with salt marsh habitat from diet studies. A summary of study results in DMF (2000) found the diet of juvenile red drum was comprised of predominantly mud crabs and fiddler crabs, the latter being closely associated with marsh habitat (Weigert and Freeman 1990).

Suggested by D. Meyer. It seems to me we need some new sections related to:

Linkages to adjacent habitats-- In this section we could discuss the important linkage of salt marsh to terrestrial habitat: the importance of oyster reefs to maintain the integrity of the salt marsh and reduce erosion of it, the importance of oyster reef and sub-aquatic vegetation (SAV) as low tide refuge habitats and the importance of the combination with salt marsh to increase habitat heterogeneity and maintain a shallow water wedge restrict predation on forage and juvenile game fishes. The linkage to upland habitats related to marshes acting as transition zones for freshwater inputs and how uplands are important a sediment sources to maintain marsh elevation relative to sea-level changes, a refuge for marsh migration during sea-level rise periods, and increasing the salinity regime profile (especially in creek feed salt marshes), which increases the heterogeneity of the marsh and increases rare species accumulation…. Etc.

Threats to salt marsh habitats—Too much fresh water input as freshets from storm water, road building causing impoundment of existing salt marsh causing salinity changes and poor hydrologic circulation patterns [this can enhance the eventual change in vegetation dominance and potentially lead to nuisance plant species, including the non-native form of the common reed (Phragmites australis) to invade disturbed areas], coastal development (bulk-heads, sea walls, revetments, etc.) causing no retreat for the salt marsh as sea-level changes occur, eutrophication, etc.

Policy for management—We need to have a coherent policy as to how the SAFMC will participate in the management of these habitats. Help direct research that is conducted at the federal, state and local level to obtain the information necessary for managing the habitats outlined in the Habitat Management Plan. There needs to be a link with federal agencies and state and local governments to proceed with best management practices to put these ideas into action and not just come out with documents to distribute and read.

Comment [8]: A Strategic Habitat Area assessment in North Carolina identified "low-elevation uplands" as an important habitat for marsh migration. Monitoring this "habitat" could be a very important aspect of emergent wetland management. Reference below.

Weaver, J., Chappell, W.S., Deaton, A.S., West, K., Hart, K., and Buckel, J. (2011). Strategic Habitat Area Nominations for Pamlico Sound System, North Carolina (Region 2). Reviewed and endorsed by Strategic Habitat Area Regional Advisory Committee and approved by the N.C. Marine Fisheries Comission. Morehead City, NC. 135 pages.

Comment [9]: timely to include effects from vertical structures on fisheries communities, hence the movement of living shorelines to stabilize tidal creek banks.

SAFMC Fishery Ecosystem Plan II: Soft Bottom Habitat

2.7 Soft Bottom/Subtidal

Note: NC Coastal Habitat Protection Plan Document was a basis for this section. We need to enhance this to include other State's input into this section. That is a major goal.

Description and Distribution

(Excerpted from NC Coastal Habitat Protection Plan)
Soft bottom habitat is unconsolidated, unvegetated sediment that occurs in freshwater, estuarine, and marine systems. Soft bottom has only one habitat requirement – sediment supply. Environmental characteristics, such as sediment grain size and distribution, salinity, dissolved oxygen, and flow conditions, will affect the condition of the soft bottom habitat and the type of organisms that utilize it. Nevertheless, the habitat itself will persist regardless of its condition unless it becomes starved for sediment or is colonized by other organisms, transforming it into another habitat such as SAV or shell bottom. Refer to FEP Volume IV for more information on ecological impacts of alterations to soft bottom habitat.

Although soft bottom habitat is defined as unvegetated and lacks visible structural habitat, the surface sediments support an abundance of microscopic plants; numerous burrowing animals are hidden below the surface (Peterson and Peterson 1979). The characteristic common to all soft bottom types is the mobility of unconsolidated, uncemented soft sediment (Peterson and Peterson 1979). Soft bottom habitat can be characterized by geomorphology (the shape and size of the system), sediment type, water depth, hydrography (riverine, intertidal, or subtidal), and/or salinity regime (DENR 2000a). It is important to understand the physical and chemical properties of soft bottom habitat since these affect the benthic organisms that inhabit these areas and, in turn, their value as fish habitat. The physical and chemical character of all soft bottom is determined by the underlying geology, basin morphology, and associated physical processes (Riggs 1996).

Estuaries and sounds - intertidal flats, unvegetated shoreline and subtidal bottom
Sediment composition of soft bottoms in estuaries and sounds varies with geomorphology and position within the estuary. In North Carolina, the basin morphology of most northern estuaries is similar to a shallow, flat-bottomed dish with a small lip around the perimeter (Pilkey et al. 1998). The estuarine shoreline is a cut bank with a narrow and shallow perimeter platform (the lip) that slopes gradually away from the shoreline to approximately 3-7 ft (1.5-2 m) deep, and then more abruptly to the floor of the central basin. The central basins deepen gradually from the inner estuary to the outer estuary from about 12 – 23 ft deep (4 to 7 m). The central basins become shallow near the mouths of the estuaries due to formation of sandy bars, and behind the barrier islands due to storm overwash and transport of sand from the inlets. Coarse sands are concentrated on the shallow perimeter platforms, shoals, and inlet mouths, while fine sediments

such as organic rich mud (ORM) are concentrated in the deeper central basins and downstream channels (Wells 1989; Riggs 1996; Pilkey et al. 1998). The width and thickness of ORM increase as the estuary widens and deepens in the downstream direction, since the fine sediments are easily suspended and transported away from high energy waters (Riggs 1996).

Unvegetated shorelines occur where wave energy prevents colonization by plants and there is a gently sloping area for sand to build upon (Riggs 2001). The shoreline provides an area to absorb the physical energy from waves, tides, and currents, protecting upland areas. Although unvegetated nontidal shorelines are ordinarily exposed from water, and therefore not used by fish, the dynamic processes of erosion and sediment deposition affect the composition and supply of sediment in adjacent shallow water habitats. This in turn affects the type and productivity of the benthic invertebrate community. For example, unvegetated sediment bank shorelines are generally eroding and sandy, providing a source of sand to adjacent waters (Riggs 2001). Sand deposits from inlet flood tide deltas and overwash events on back barrier islands form shallow sand flats behind the islands. In contrast, marsh or swamp forest shorelines are generally not eroding and have a high organic content, thus providing fine organic sediments to adjacent waters. Peats, sediments with more than 50% organic matter, form in the swamp forests of riverine floodplains or in coastal marshes (Riggs and Ames 2003). Several shoreline erosion studies have been conducted along North Carolina's coast that provide information on the character and condition of intertidal, shoreline, and shallow subtidal soft bottom and were compiled and summarized in Riggs (2001) and updated in 2011.

The inlets separating North Carolina's barrier islands are part of a sand-sharing system among the islands, estuaries, and nearshore ocean. Intertidal flats or deltas form on the ebb and flood sides of inlets as sediments shift with tides and waves. Sediments in the vicinity of inlets are typically composed of coarse sands and shell fragments (Peterson and Peterson 1979). Ebb-tidal and flood-tidal deltas (i.e., the seaward and estuarine shoals of an inlet, respectively) are formed by waves and currents, and may contain large volumes of sand. Intense wave and current energy cause the flats to continually change, erode, and reform. The high instability of the ebb and flood tide deltas makes colonization by benthic invertebrates difficult (Peterson and Peterson 1979). Inlets are classified as stable, migrating, or ebb-tidal delta breaching (Fitzgerald et al. 1978). Unstable inlets may form extensive spits, tidal deltas, and sandbars, creating bathymetric complexity (or differences in water depth) in nearshore waters that attract certain fish species. The process of channel realignment and abandonment provides a mechanism for large sandbar complexes to move onto the adjacent barrier islands, supporting productive intertidal beach communities (Cleary and Marden 1999).

Ecological Role and Function
(from CHPP)

Soft bottom plays a very important role in the ecology of estuarine ecosystems as a storage reservoir of chemicals and microbes. Intense biogeochemical processing and recycling establish a filter to trap and reprocess watershed-derived natural and human-induced nutrients and toxic substances. These materials may pass through an estuary (Matoura and Woodward 1983), become trapped in the organic rich oligohaline (low salinity) zone (Sigels et al. 1982; Imberger et al. 1983), or migrate within the estuary over seasonal cycles (Uncles et al. 1988). The fate of the materials depends upon salinity gradients, which are driven by freshwater discharges, density stratification, and formation of salt wedges (Matson and Brinson 1985, 1990; Paerl et al. 1998). Density gradients (stratification) hamper mixing and oxygen exchange of sediments and water in bottom waters with overlying oxygenated waters, leading to depletion of dissolved oxygen in bottom water (Malone et al. 1988).

In North Carolina‘s slow-moving, expansive estuaries, nutrients and organic matter from the watershed runoff and phytoplankton production are stored in the soft bottoms. Depending upon freshwater discharge and density stratification, these materials are recycled within the sediments via microbial activities and from the sediments into the overlying waters. Increased inflows of Fishery nutrients exacerbate the process, leading to more rapid and expanding dissolved oxygen depletion. In organic enriched oligohaline zones (e.g., Pamlico and Neuse River estuaries), nutrient-induced recycling results in higher microbial activity and oxygen depletion (Buzzelli et al. 2002; MacPherson et al. 2007)(B.J. Copeland, NCSU, pers. com., 2004).

Although soft bottom habitat is composed of unconsolidated shifting sediments, colonization by benthic microalgae reduces the extent to which sediment is resuspended at low velocities, stabilizing bottom sediments and reducing turbidity in the water column (Holland et al. 1974; Underwood and Paterson 1993; Yallop et al. 1994; Miller et al. 1996). In spite of this, microalgae cannot stabilize sediments under intense or prolonged disturbance conditions, such as during large storm events or in the surf zone (Miller 1989). Structure from tube dwelling invertebrates also helps to bind the sediment (Peterson and Peterson 1979), while filtering activity of dense aggregations of suspension feeders (hard clams) clears significant amounts of plankton and sediment from the water column and improves water clarity (Miller et al. 1996). Yet, because of the absence of large, extensive structure, soft bottom provides relatively less stabilization benefits than other estuarine habitats.

Intertidal shorelines, flats, tidal deltas, and sandbars along the ocean shoreline buffer and modify wave energy, reducing shoreline erosion. Alterations to the ebb and flood tide deltas can result in significant changes in the adjacent barrier island shorelines. Flood-tidal deltas are an important source of sand, which allows barrier island migration to respond to sea level rise (Cleary and Marden 1999). The soft bottom associated with inlets has a great influence on overall barrier island dynamics.

Fish utilization

Like the water column, soft bottom is used to some extent by almost all native coastal fish species in North Carolina. However, certain species are better adapted to, characteristic of, or dependent on shallow unvegetated bottom. Flatfish, rays, and skates are well suited for utilization of soft bottom. Juvenile and adult fish species that forage on the rich abundance of microalgae, detritus, and small invertebrates are highly dependent on the condition of soft bottom. Table 3.2-11 summarizes important fishery and nonfishery species that are dependent on subtidal bottom for some portion of their life history and the ecological function of the soft bottom habitat.

Foraging

One of the most important functions of soft bottom habitat is as a foraging area. Members of several trophic levels in the benthic community benefit directly or indirectly from a) the high concentrations of organic matter transported to and produced on soft bottom and b) the numerically abundant, diverse invertebrate fauna associated with soft bottom – including herbivores (e.g., planktonic and benthic algal feeders), detritivores, predators of benthic invertebrates and fish (secondary consumers), and predators of those predators (tertiary consumers) (Peterson and Peterson 1979). On shallow intertidal flats, planktonic and benthic feeding herbivorous fish, (e.g., anchovies, killifish, menhaden) consume phyto- and zooplankton in the water column, as well as suspended benthic algae, microfauna, and meiofauna (Peterson and Peterson 1979). In North Carolina, largemouth bass (*Micropterus salmoides*) and white catfish (*Ameiurus catus*) have been reported to forage on both benthic-associated crustaceans and fishes in oligohaline, intertidal rivulets of the upper Cape Fear River Estuary (Rozas and Hackney 1984). Most fish that forage on estuarine soft bottom are predators of benthic invertebrates. These fish include juvenile and adult rays, skates, flatfish, drums, pigfish, sea robins, lizardfish, gobies, and sturgeons (Bain 1997; Peterson and Peterson 1979). Larger piscivorous fishes typically move onto estuarine flats during high water to feed on baitfish. These predators include sharks (sandbar, dusky, smooth dogfish, spiny dogfish, Atlantic sharpnose, and scalloped hammerhead), red drum, weakfish, bluefish, spotted seatrout, striped bass, and estuary-dependent reef fish (black sea bass, gag grouper, sand perch) (Peterson and Peterson 1979; Thorpe et al. 2003). Flatfish, rays, and skates are particularly adapted to forage on shallow flats with their compressed body forms (Peterson and Peterson 1979). Small flatfish (e.g., bay whiff, fringed flounder, hogchoker, and tonguefish) feed mostly on copepods, amphipods, mysids, polychaetes, mollusks, and small fish. Summer and southern flounder and larger flatfish primarily consume fish such as silversides and anchovies as well as shrimp and crabs, small mollusks, annelids, and amphipods (Burke 1995; Peterson and Peterson 1979). Ocean soft bottom, particularly the surf zone and along shoals and inlets, serves as an important feeding ground for numerous fishes foraging on benthic invertebrates (Peterson and Peterson

1979). These predators can have high recreational and commercial value, and include Florida pompano, red drum, kingfish, spot, Atlantic croaker, weakfish, Spanish mackerel, and striped bass. Reef species known to forage over sand bottom away from the reef include tomtate (*Haemulon aurolineatum*), whitebone porgy (*Calamus leucosteus*), cubbyu (*Equetus umbrosus*), black sea bass (*Centropristis striata*), and scup (*Stenotomus chrysops*) (Lindquist et al. 1994b).

Spawning

Many demersal fish spawn over various areas of soft bottom habitat in North Carolina‘s coastal waters (Table 3.2-11). In freshwater, resident species such as largemouth bass (*Micropterus salmoides*) and bluegill (*Lepomis macrochirus*) spawn on shallow flats where they lay eggs in bowl-shaped nests. Eggs may be dependent on the small structure available on the unvegetated bottom, such as emerging worm tubes or woody debris, to hold them in position. Since all life stages of freshwater resident fish (spawning adults, eggs, larvae, juveniles) remain near the same area of soft bottom habitat, they are relatively more vulnerable to degraded soft bottom habitat conditions than migratory species. Anadromous species, such as Atlantic and shortnose sturgeon (*Acipenser oxyrinchus oxyrinchus* and *A. brevirostrum*, respectively), spawn in upper freshwater portions of coastal rivers (Moser and Ross 1995).

Estuarine spawners include resident fish and invertebrates, as well as migratory fish that are summer estuarine spawners. Estuarine resident species include common invertebrates that occupy the intertidal flats, like hard clams, whelks, snapping shrimp, and hermit crabs. Small schooling baitfish such as mummichogs and striped killifish spawn in the marsh edges near soft bottom (Hildebrand and Schroeder 1972; Manooch 1984). Species of flatfish, including the windowpane, and hogfish have been reported to spawn on estuarine soft bottom (Hildebrand and Schroeder 1972; Manooch 1984).

Summer estuarine spawners include several species of drum. Weakfish and silver perch were documented spawning in deep estuarine channels near Pamlico Sound inlets (Ocracoke and Hatteras inlets) and in deep areas of Pamlico Sound from May to September, peaking in May and June (Luczkovich et al. 1999a). Spotted seatrout are year-round residents of estuaries along the South Atlantic coast and spawning takes place inshore and in coastal areas (McMichael and Peters, 1989). In North Carolina, spotted sea trout spawn on the east and west sides of Pamlico Sound during a similar time period, with peak activity observed around July. Specific spawning areas for spotted sea trout identified on the west side of Pamlico Sound were Rose Bay, Jones Bay, Fisherman‘s Bay, and Bay River (Luczkovich et al. 1999a). In South Carolina, spotted seatrout spawn in similar habitats from April through September (Roumillat and Brouwer, 2004). Red drum were documented spawning in the mouth of the Bay River on the west side of Pamlico

Sound, and in estuarine channels near Ocracoke Inlet (Luczkovich et al. 1999a). Blue crabs also spawn near inlets in summer (DMF 2000d).

Nursery

Shallow soft bottom habitat, usually adjacent to wetlands, is utilized as a nursery for many species of juvenile fish. The shallow unvegetated bottom provides an abundance of food and is inaccessible to larger predators. Shallow unvegetated flats have been documented as being particularly important nursery habitats for juvenile summer and southern flounder (Burke et al. 1991; Walsh et al. 1999). A partial list of species that use soft bottom habitat as a nursery area is included in Table 3.2-11. Studies and ongoing juvenile fish monitoring conducted by the North Carolina Division of Marine Fisheries have found that shallow unvegetated bottom supports high abundances of juvenile fish, composed of relatively few species but which have similar life histories and feeding patterns (Ross and Epperly 1985).

The dominant juvenile species utilizing shallow soft bottom estuarine nursery areas are estuarine dependent winter spawners. Most of the species spawn offshore during the winter. The larvae are transported through inlets into estuarine waters. For many species, the uppermost area of shallow creek systems corresponds to where larval settlement of winter spawned species occurs – the primary nursery areas (Weinstein 1979; Ross and Epperly 1985). However, in tributaries on the western side of Pamlico Sound, such as Neuse, Pamlico, Bay and Pungo rivers, larval settlement tends to occur in lower portions of the creeks. Unlike larval settlement in areas south of Pamlico Sound, salinity is low in the upper reaches of the Sound's tributaries and this may deter larval settlement in those areas. Abundance of juvenile species in estuarine nursery areas peaks between April and July and is correlated with water temperatures (Ross and Epperly 1985). As fish grow, they move to deeper waters and areas lower in the estuary.

In North Carolina, many areas used as nurseries by estuarine dependent fish have been designated as Primary or Secondary Nursery Areas by the Marine Fisheries Commission. However, there are other areas of soft bottom that function as nurseries but are undesignated. Benthic anadromous fish, such as Atlantic and shortnose sturgeon, use freshwater soft bottom as a nursery.

Refuge

Soft bottom habitat can provide refuge to some organisms in some locations through predator exclusion. Shallow, intertidal flats may be inaccessible to large fish predators and therefore protect small and juvenile fish and invertebrates (Peterson and Peterson 1979; Ross and Epperly 1985). Consequently, juvenile fish recruit into the shallowest portions of the estuary first. Many invertebrates, including hard clams, can avoid predation by burrowing into the sediment (Luettich et al. 1999). Flatfish, such as flounder and rays, and other small cryptic fish, like gobies, can bury slightly into the sediment, camouflaging themselves from predators (Peterson and Peterson 1979). Nonetheless, soft bottom habitat in deepwater is a vulnerable place for small fish and invertebrates that cannot burrow. For example, flounders also camouflage themselves in the sediment to ambush prey (Walsh et al. 1999). Because of this, many fish in subtidal water will venture out to feed on the open bottom only at night (Summerson and Peterson 1984).

Corridor and connectivity
Freshwater and estuarine soft bottom channels are the highways for migrating adult demersal fish species to and from other estuarine habitats and the ocean. Demersal feeding anadromous fish, such as sturgeon and striped bass, require a corridor of soft bottom to reach upstream spawning areas. Inlets act as conduits for exchange of sediment, water, and marine organisms between the estuaries and the ocean. Because large fish are less likely to be consumed as prey, they can travel relatively safely over less turbid sand flats and in channels of the middle and lower estuaries (Walsh et al. 1999). Smaller flatfish tend to be more abundant in the shallower uppermost portion of the estuary, where salinities are low, turbidity high, and sediments muddy with high detritus content (Walsh et al. 1999).

While connectivity among structured habitat patches, such as SAV, wetlands, and shell bottom, facilitates movement of blue crabs and other mobile predators through an estuary, a few meters of unvegetated bottom can act as a barrier to movement (Micheli and Peterson 1999). Such barriers can be beneficial to small invertebrates by potentially obstructing predator dispersal and reducing predation risk. Small crabs, gastropods, and infaunal bivalves, such as hard clams, were more abundant, denser, and had higher survival rates on isolated oyster beds (at least 10-15 m of unvegetated bottom between habitats) than on oyster beds adjacent to salt marsh or SAV (Micheli and Peterson 1999). Blue crab predation on infaunal bivalves was greater along vegetated edges of salt marshes and seagrass beds than in unvegetated intertidal flats (Micheli and Peterson 1999). Although structural habitat separations by unvegetated soft bottom may benefit the survival or viability of infaunal populations, fish and crustacean productivity may be enhanced by connectivity of structured estuarine habitats (Micheli and Peterson 1999). These habitat-mediated predator/prey interactions point out the importance of maintaining the integrity of an entire estuarine system.

Species composition and community structure
Benthic microalgae are a key part of the food chain in estuarine soft bottom habitat. Benthic microalgae are microscopic photosynthetic algae that live in the top few millimeters of the surface of soft bottom (Miller et al. 1996). Because the unvegetated bottom appears barren, but is actually rich in photosynthetic algae, MacIntyre et al. (1996) referred to benthic microalgae as ―The Secret Garden.‖ Benthic microalgae on sand, mud flats, and subtidal bottom are composed primarily of benthic diatoms and blue green algae, with benthic dinoflagellates and filamentous

green algae also present (Peterson and Peterson 1979). Dense mats of blue green algae sometimes form in protected higher portions of intertidal flats, giving the sediment surface a dark brown or blue-green appearance, which can form a crusty mat when dry at low tides (Peterson and Peterson 1979). Diatom mats are more abundant in the lower intertidal zone (Peterson and Peterson 1979). Benthic microalgae can either be attached to sediment particles or be mobile, migrating vertically through the sediment. Productivity depends on photosynthesis by these microalgae, which can only occur in sediments having adequate light penetration (MacIntyre et al. 1996). Photosynthetically active light generally penetrates only about 2-3 mm into the sediment, but can reach 5-20 mm in sandy, high energy environments.

Most benthic invertebrates inhabiting soft bottom live in the sediment (infauna), as opposed to the bottom surface (epifauna), because of the high mobility of sediments (Peterson and Peterson 1979). These animals are classified by size and feeding mode. Microfauna are the very small protozoans (< 0.06 mm). Meiofauna are about 0.06 – 0.40 mm in size (the size of a sand grain), and include nematodes and copepods. Both microfauna and meiofauna are important grazers on benthic microalgae and bacteria. Macrofauna (>0.5 mm) contribute the most to infaunal biomass and include organisms such as amphipods, polychaetes, mollusks, echinoderms, and crustaceans (Peterson and Peterson 1979). These macrofauna may be deposit feeders or suspension feeders (Peterson and Peterson 1979; Miller et al. 1996). Deposit feeders ingest sediment and detrital deposits and assimilate bacteria, fungi, and microalgae from them. Compared to detritus and larger plants, microalgae may be a nutritionally richer food source for benthic invertebrates (Miller et al. 1996). Deposit feeders include mud snails, many polychaete worms, and certain bivalve clams and crustaceans.

Table 3.2-11. Partial list of common or important fish species occurring on soft bottom habitat in riverine, estuarine, and ocean waters, and ecological functions provided to those species. Bolded species indicate relatively higher association on soft bottom habitat (Source: Street et al. 2005). Suspension feeders capture particles suspended in the water column. Common suspension feeders are bivalves such as the hard clam (*Mercenaria mercenaria*) and razor clam (*Tagelus plebeius*), and some polychaete worms (Miller et al. 1996). When sediment is resuspended, the benthic microalgae become available to the suspension feeders (Miller et al. 1996). A large proportion of intertidal bivalves‘ diet has been shown to consist of suspended benthic microalgae, particularly when chlorophyll concentrations in the water column are low (Page and Lastra 2003). While resuspended benthic microalgae can be beneficial to the invertebrate community as an additional food source, excessive suspended sediment and associated algae have been found to reduce growth rates and survival of macrofauna, such as hard clams (Bock and Miller 1995). Although the abundance of food sources affects invertebrate populations, benthic predators (such as spot and pinfish) were found to have a larger influence on soft bottom community composition and biomass relative to that of nutrient availability (Posey et al. 1995).

On submerged flats and shallow bottom, blue crab (*Callinectes sapidus*) is an important predator. Other mobile invertebrates include horseshoe crab (*Limulus polyphemus*), whelks (*Busycon* spp.), tulip snails (*Fasciolaria* spp.), moon snails (*Polinices duplicatus*), penaeid shrimp (*Farfantepenaeus* spp. and *Litopenaeus* spp.), hermit crabs (*Pagurus* spp., *Petrochirus* spp., and *Clibanarius vittatus*), sand dollars (*Mellita quinquiesperforata*), and spider crabs (*Libinia* spp.). Overall, estuarine soft bottom supports a high diversity of benthic invertebrates, with over 300 species documented in the southern portion of North Carolina (Hackney et al. 1996). Soft bottom is the most abundant submerged coastal fish habitat, and estuarine acreage of soft bottom has undoubtedly increased over time as shell bottom, SAV, and wetland habitats have declined.

Soft Bottom/Subtidal as Essential Fish Habitat

SAFMC Fishery Ecosystem Plan II: Oyster Reefs and Shell Banks

3.2.4 Oyster Reefs and Shell Banks
Description and Distribution

Introductory Comments from SurveyMonkey

What major South Atlantic actions, plans, and activities are you aware of that belong in this section update?

- *Use of oyster reefs in living shorelines projects? Restoration projects (do restored reefs provide same functions?)*
- *The Nature Conservancy's work on the South Atlantic Bight Marine Assessment (https://www.conservationgateway.org/ConservationByGeography/NorthAmerica/UnitedStates/edc/reportsdata/marine/sabma/Pages/default.aspx) would lend itself well to this section.*
- *Though the work isn't on the website yet, Mary Conley (mconley@tnc.org) might be able to share the maps associated with the work (parts of the assessment are currently available).*
- *Also the SALCC Conservation Blueprint (http://www.southatlanticlcc.org/page/conservation-blueprint) has been updated since this section was written and might be helpful.*
- *Create a new section to deal with socioeconomics of oyster reefs. Environmental variability, climate change and vulnerability of oyster reefs.*

What are the most critical updates/edits needed in this section?
This document does a very good job on covering the description of the habitat and its importance related to the coastal habitats including linkages to adjacent habitats.

- *It seems to me there needs to be a better assessment of reef, aggregations and accumulation habitat related to areal totals for oyster reef and shell bank habitats in the different states within the SAFMC area.*
- *Make clear distinctions between intertidal and subtidal reefs as well as fringing and patch reefs because these can serve different functions. Along this line what about the reefs that are produced on sea wall. These too are important and provide important habitat in areas that might be space limited or significantly impacted by development. Should these too be included? It is hard to manage something you do not have relatively accurate data on how much it is you have and what kind it is.*
- *Better assess the managed species that rely on these habitats at some point in their life-histories and the forage species they rely upon, better linking oyster reef and shell bank to EFH. Here and in other sections it is necessary to partner with Federal Agencies, each state, with non-government organizations (NGOs), and NOAA Sea Grant to obtain*

funds to acquire this fundamental information. This should be a priority issue for the Council.

- *More focus on oysters (after all that is the name of the section) with less material on other reef forming species not really found in South Atlantic*
- *More up-to-date peer reviewed literature.*
- *Regional mapping of oyster reefs. State agency information of current oyster reef mapping efforts underway.*
- *Current mapping of oyster habitat distribution, living oyster distribution/densities, and comprehensive restoration project locations - show the gaps in SAFMC online Atlas.*

What do you see as the most critical edits/amendments to this section needs?

- *New section: Linkages to adjacent habitats-- In this section we could discuss the important linkage of fringing oyster reef to terrestrial habitat: the importance of oyster reefs to maintain the integrity of the salt marsh and reduce erosion of it, and essentially the uplands adjacent to the marshes.*
- *The importance of oyster reef and sub-aquatic vegetation (SAV) as low tide refuge habitats and the importance of their combination with salt marsh to increase habitat heterogeneity and maintain a shallow water wedge restrict predation on forage and juvenile game fishes. Perhaps we need to better describe the linkage of oyster reefs to the production of an initial stable habitat into which other habitats might develop through plant invasion, including salt marsh and SAV.*
- *Importance for migration of fringing oyster reefs upland as sea level changes occur during sea-level rise periods and the need to reduce impediments for this migration.*
- *Threats to oyster reef and shell bank habitats. In this section we could outline some of the threats to these habitats that need to be managed including: over harvest, habitat destruction via removal of biogenic substrate; coastal development issues such as fresh water input as freshets from storm water, road building causing impoundment of existing estuarine areas causing salinity changes and poor hydrologic circulation patterns; land based practices causing oyster recruitment limitations and reef degradation through sedimentation, chemical toxification, eutrophication, pronounced salinity variation and physical destruction of the habitat; threats from oyster disease, parasites and predators, invasive and introduced oyster and predator species;, etc.*
- *Policy for management. We should have a coherent policy as to how the SAFMC will participate in the management of these habitats. Help direct research that is conducted at the federal, state and local level to obtain the information necessary for managing the habitats outlined in the Habitat Management Plan. There needs to be a link with federal agencies and state and local governments to proceed with best*

What connectivity modeling issues must be considered?

- *Modeling of fisheries habitat utilization data and how it relates to oyster reefs. Connectivity of fish species found on oyster reefs and how that relates to the nearshore and offshore environment.*

Please share with your edit team lead additional thoughts and perspectives on this section you feel would be beneficial to creating a useful product.

- *This level of detail is beyond my expertise so I can't comment on the current accuracy of the details, but there have been some new mapping analyses that might be of benefit for this section.*
- *Information on techniques for oyster reef monitoring*

Guiding Principles

Description and Distribution - condense - too long
Couple of paragraphs - condense sections considerably
There needs to be consistency with how scientific and common species names are used.
Update research references, where possible

Short Intro
Description and distribution
Role and Function
Species Composition
EFH Aspect - why it's important for Fed Managed species

(Dave) Section - Threats to oyster reefs and shell bank hab --It seems to me there needs to be a better assessment of reef, aggregations and accumulation habitat related to areal totals for oyster reef and shell bank habitats in the different states within the SAFMC area. I think we need to make clear distinctions between intertidal and subtidal reefs as well as fringing and patch reefs because these can serve different functions. Along this line what about the reefs that are produced on sea wall. These too are important and provide important habitat in areas that might be space limited or significantly impacted by development. Should these too be included? It is hard to manage something you do not have relatively accurate data on how much it is you have and what kind it is. There is also a need to better assess the managed species that rely on these habitats at some point in their life-histories and the forage species they rely upon, better linking oyster reef and shell bank to EFH. Here and in other sections it is necessary to partner with Federal Agencies, each state, with non-government organizations (NGO's), and NOAA Sea Grant to obtain funds to acquire this fundamental information. **This should be a priority issue for the Council**.

Each State and Fed drop in Jurisdiction (Lisa) - Section Policy Management - map - utilize partners (State, Fed NGO) - for research
VA: is in partnership with Chesapeake Bay Program for habitat restoration for oysters. They do not have any guiding documents for their work.

NC: uses multiple documents for oysters and living shorelines (that use oysters and oyster shells for stabilizing the shorelines). These include NC's Oyster FMP (**http://portal.ncdenr.org/c/document_library/get_file?uuid=75abdf16-9291-4d9e-9bf5-cdabafab9814&groupId=38337**), Coastal Management's guidelines for living shorelines (**http://portal.ncdenr.org/c/document_library/get_file?uuid=7a9230cb-ed99-4324-b9fe-3243a9b78c95&groupId=38319**), the North Carolina Coastal Federation's recommendations on their website (**http://www.nccoast.org/protect-the-coast/restore/oyster-habitat/**), and APNEP's CCMP (**http://portal.ncdenr.org/web/apnep/ccmp** **Specifically Objectives B3 and C5**).

SC: SC has some of the most extensive shellfish beds of any state in the region due to our extensive coastline and network of salt marsh systems. Unlike some other states further north, almost all of our oyster beds are intertidal. This makes them easier to assess compared to subtidal shellfish beds, but the enormous extent of our beds still makes mapping and monitoring this resource complicated and time consuming. The most recent comprehensive assessment of South Carolina's shellfish beds occurred from 2003-2006 when the state obtained and analyzed high resolution digital orthophotos to create new GIS maps of all the shellfish beds, with extensive ground-truthing by field teams to verify the accuracy of those maps. The ground-truthing has continued and includes low altitude very high resolution digital photography of the beds using helicopter surveys. A summary of the program and maps of all the shellfish grounds can be found at the following link: http://www.dnr.sc.gov/GIS/descoysterbed.html.

Due to the extensive nature of our shellfish beds, there is no single procedure for monitoring and managing those beds, which include state shellfish grounds (SSGs) that can be harvested both commercially and recreationally, recreational shellfish grounds open only for recreational harvest, culture permit areas for commercial harvest only (unless a recreational harvester has permission from the permit holder), and areas that are not open for any harvesting due to various reasons (often due to water quality). Maps of the state shellfish grounds can be viewed here http://www.dnr.sc.gov/marine/shellfish/ground/index.html. These maps also identify public health status. A summary of the shellfish bed management program, and links to regulations can be found at: http://www.dnr.sc.gov/marine/shellfish/commregs.html.

Our agency has no standard written protocol for how our harvestable beds are monitored and managed, but in general terms, Division staff conduct annual on the water or helicopter assessments of the managed areas to determine if they appear to be overharvested or suitable for continued harvesting. A subset of the SSGs are generally closed each year to allow for recovery, with the areas staggered around the state to ensure that open grounds are available in all parts of the state. The culture permit areas (commercial harvest only) have the most extensive regulations regarding re-planting requirements (I have the document as a word doc if we want to include it).

The SCDNR considers all shellfish beds as EFH and therefore, to my knowledge, has no special designation of some beds over others with respect to EFH related management activities. The SCDNR also conducts extensive shellfish bed plantings that are both large-scale (i.e. using barges filled with shell that are blasted onto the shoreline to provide suitable substrate for spat settlement, and smaller scale plantings using shell bags that are placed along the shoreline by volunteer groups (SCORE).   The link to the SCORE planting program is http://score.dnr.sc.gov Information on the larger scale planting, if needed, could be provided by Nancy Hadley or her staff. Some of large scale and SCORE plantings are done in areas closed for shellfish harvesting as part of mitigation or restoration efforts since it is well documented that shellfish beds are critical habitat for many of our fish and crustacean species. SCDNR staff also continue to have extensive monitoring and research programs on shellfish grounds. Information on much of these activities can be found at http://www.dnr.sc.gov/marine/shellfish/index.html.

The ability to harvest shellfish is also co-managed by the SC Dept. of Health and Environmental Control (SCDHEC), which has primary regulatory authority on whether beds are suitable for safe harvest. During heavy rain events, areas along the coast that are normally open for shellfish harvesting are sometimes closed if bacterial levels indicate unsafe harvesting conditions. SCDHEC has a very extensive shellfish monitoring program for bacterial levels throughout the state. Finally, shellfish harvesting is generally limited to the colder months ("r" months) rather than year round (although I'm not sure that applies to some mariculture operations). The specific season is defined by the SCDNR.

GA: Currently GADNR does not use any plan or guiding documents in regards to creating oyster reefs and shellfish banks. Oyster restoration sites are selected by the Habitat Restoration and Enhancement Unit, or by political motives, or through partnerships with organizations like the Coastal Conservation Association of Georgia, etc. Restoration sites occur in prohibited waters. Enhancement sites are selected by the HREU in partnership with DNR’s commercial shellfish biologist. Enhancement sites are located in one of Georgia’s seven public shellfish harvest areas.

GADNR has an internal Habitat Work Group that vet’s all sites. The HREU is working on an Oyster Restoration and Enhancement Strategic Plan which is GADNR’s 5 year vision for restoration and enhancement activities. This plan will include priorities sites per county for potential restoration and enhancement. This plan is currently under development.

FL: We don't have specific BMPs on oyster restoration, but we do have monitoring guidelines (TNC-Cohen document - I have available as a pdf if we want to include it) that we use. We also have oyster reef maps and our partners are obtaining information on East Coast oysters where we don't have this information. We also reference the 2012 GSMFC oyster reef management plan (I have this pdf if we want to include it as well). There are some permitting guidelines and LSL exemptions for construction of oyster reefs along private property shorelines now in State Environmental Permitting Guidelines as well.

Policy for management— We should have a coherent policy as to how the SAFMC will participate in the management of these habitats. Help direct research that is conducted at the federal, state and local level to obtain the information necessary for managing the habitats outlined in the Habitat Management Plan. There needs to be a link with federal agencies and state and local governments to proceed with best management practices to put these ideas into action and not just come out with documents to distribute and read. This section should include **Regional mapping of oyster reefs. State agency information of current oyster reef mapping efforts underway.**

- **Current mapping of oyster habitat distribution, living oyster distribution/densities, and comprehensive restoration project locations - show the gaps in SAFMC online Atlas.** TNC has a South Atlantic Bight Marine Assessment that will help inform this section.

There are a few models out now for management of oyster, I'm not aware of models for other reef forming species.

(Dave) Section on Linkages to different hab type importance to uplands and other types - Independent Section - (seawalls)

Linkages to adjacent habitats-- In this section we could discuss the important linkage of fringing oyster reef to terrestrial habitat: the importance of oyster reefs to maintain the integrity of the salt marsh and reduce erosion of it, and essentially the uplands adjacent to the marshes. The importance of oyster reef and sub-aquatic vegetation (SAV) as low tide refuge habitats and the importance of their combination with salt marsh to increase habitat heterogeneity and maintain a shallow water wedge restrict predation on forage and juvenile game fishes. Perhaps we need to better describe the linkage of oyster reefs to the production of an initial stable habitat into which other habitats might develop through plant invasion, including salt marsh and SAV. Importance for migration of fringing oyster reefs upland as sea level changes occur during sea-level rise periods and the need to reduce impediments for this migration.

(January) Sub-Section: Upland links, SLR, migration with other wetlands

Sea level rise (SLR) has the potential to impact shell bottom habitats along the Atlantic coast. SLR is dependent upon atmospheric temperature and the dynamics of polar ice masses. Globally, 85 percent of oyster reefs that once dominated the bays and estuaries of the world have disappeared. Eastern oyster (Crassostrea virginica) populations have declined along the Atlantic Coast due to habitat loss, predation, disease, pollution, and harvest pressure. Environmental stressors such as extended changes to temperature regimes, precipitation, and streamflow patterns may also play a role in oyster distributions, growth, reproduction, and survival. Does the eastern oyster possess sufficient resilience to survive ecological and environmental stressors as well as impacts from SLR?

The effects of globally increased water temperatures may initially provide a handful of benefits to oysters and other shellfish. These benefits include 1) increased filtration and growth rates; 2) a longer spawning season; 3) a shorter duration of the planktonic larval phase; 4) range expansion of lower latitude species; and 5) increased subaqueous space allowing for extended vertical accretion. These benefits to oysters and other shellfish may be short lived as warming water temperatures can 1) increase susceptibility to environmental stressors; 2) increase rates of infection from oyster parasites; 3) alter environmental cues for reproduction; and 4) temporal mismatches may occur between larval production and food supply.

Changes in precipitation may influence freshwater inflow, nutrient delivery systems, and salinity regimes. Increased precipitation will decrease salinity in estuarine systems. Nutrient delivery systems may be disrupted as increased precipitation and freshwater inflow cause water column stratification and nutrient enrichment from increased runoff. Oysters are physiologically stressed at salinities less than 10 ppt resulting in reduced rates of filtration and respiration. This stressor can also cause declines in larval oyster production and larval survivorship. Direct effects of physical stress on oysters can result in mortality. Oysters have the potential to freeze to death during the winter season if exposed above the waterline during low tide. In addition, the physiological stress of hypoxic or anoxic aerial exposure can result in mortality.

SLR has the potential to create fundamental shifts in habitat availability, coastal and freshwater wetland distributions, intertidal movements of oysters, and shoreline stabilization. Intertidal oyster reefs may be able to persist sub-tidally if submerged by SLR but increased rates of predation by boring sponges, oyster drills, and blue crabs may limit ability. Rising tidal elevations may potentially affect growth and / or drown sub-tidal reefs. Coastal development may also impede oyster movements landward. Corridor functions between reefs and adjacent tidal marsh may be disconnected as a result of SLR. This in turn will decrease habitat quality for fish and macroinvertebrates that use intertidal reefs.

As global carbon dioxide concentrations increase ocean acidification can occur, a reduction in ocean pH, which is an additional environmental stressor to oyster reefs. Oysters experience dissolution of adult calcareous shells, decreases in 1) growth rates; 2) calcification; and 3) larval development and settlement when pH levels are less than 7.5. Reef development and maturation will slow in these conditions and mature oysters will face increased predation pressure. Decreases in available shell bottom will lead to reduced habitat complexity and biodiversity. Coastal managers have cause for concern in regards to SLR but adaptive management, coastal planning, and modeling may provide hope for the oyster yet.

(Whole team can add once outline is sorted) - Section author Steve - Section:Research Needs and Priorities?

What's there, loss gain, net impact
Better info on hab itself

Threats to oyster reef and shell bank habitats—In this section we could outline some of the threats to these habitats that need to be managed including: over harvest, habitat destruction via removal of biogenic substrate; coastal development issues such as fresh water input as freshets from storm water, road building causing impoundment of existing estuarine areas causing salinity changes and poor hydrologic circulation patterns; land based practices causing oyster recruitment limitations and reef degradation through sedimentation, chemical toxification, eutrophication, pronounced salinity variation and physical destruction of the habitat; threats from oyster disease, parasites and predators, invasive and introduced oyster and predator species;, etc.

Not sure if this is appropriate place, but most shellfish managed as a resource, but not as habitat.

Section on Aquaculture (Steve) -

Living Shorelines (What's up in each state) - restoration aspect - discuss this. If a restoration section is added then should it include information on oyster monitoring techniques?
Section of Socioeconomic's of oyster reefs -Environmental variability, climate change and vulnerability of oyster reefs.

Reef-forming Species

In the western Atlantic, oysters, mussels, and one genus of gastropod build three-dimensional structures that are commonly called reefs (Figure 3.2-14). Wood (1998, 1999) reviews the term reef, and discusses its origin and those taxa and concepts that relate to reefs. The term derives from a Norse term rif, or hazardous rib of sand, rock, or biologically generated substrate near the surface. Wood (1999) includes the following as extant reef producers: corals, coralline and calcareous algae, sabellariid and serpulid polychaetes, oysters, vermetid gastropods, bryozoans, sponges, and stromatolites (i.e., *Cyanophytes*). Other terms such as bars and beds also refer to reef structures that are created by the organisms themselves. Holt et al. (1998) define biogenic reefs as: *solid, massive structures which are created by accumulations of organisms, usually rising from the seabed, or at least clearly forming a substantial, discrete community or habitat which is very different from the surrounding seabed. The structure of the reef may be composed almost entirely of the reef building organism and its tubes or shells, or it may to some degree be composed of sediments, stones and shells bound together by the organisms.*

The focus here includes many shellfish species (e.g., mussels, dense clam beds) that may be classified somewhere between non-reef and reef-forming biotopes. Holt et al. (1998) try to characterize these biotopes, but this is a difficult task. Furthermore, researchers often refer to the structure that a species generates as a habitat, biotope or biogenic reef. We focus on species that create unique and definable areas that are different from the surrounding unstructured sediments. Although many species typically occur on shellfish reefs, the main structural component is formed by the attachment of many individual shellfish to each other. At least three species of oysters occur along the Atlantic coast, in addition to several mussel species and other molluscs (e.g., vermetid gastropods) (Abbott 1974). Of these, only the Eastern (or American) oyster (*Crassostrea virginica*), blue mussel (*Mytilus edulis*), and horse mussel (*Modiolus modiolus*) typically form reefs along the Atlantic coast. In the Chesapeake Bay and elsewhere, there is uncertainty over whether a non-native oyster from the Pacific (*C. ariakensis*) can serve both as a reef builder and suitable fisheries resource substitute for *C. virginica* (NRC 2004; Ruesink et al. 2005).

Figure 3.2-14. Examples of intertidal and subtidal shellfish habitats (Source: ASMFC, 2007). A and B: Pen shell, *Atrina zelandica*, aggregations in New Zealand (Source: Simon Thrush, National Institute of Water and Atmospheric Research, New Zealand); C: *Modiolus modiolus* reefs in St. Joe Bay, Florida (Source: Brad Peterson, State University of New York, Stony Brook); D: Nesting oyster catchers on intertidal shell accumulations along the Intracoastal Waterway (Source: Phil Wilkinson, South Carolina Department of Natural Resources); E: Intertidal oyster reefs at Canaveral National Seashore (Source: Loren Coen, South Carolina

Department of Natural Resources); F: Close-up of intertidal oysters on South Carolina reefs (Source: Loren Coen, South Carolina Department of Natural Resources).

Estuarine and marine mussels

Reef-forming mussels include the *Mytilus* spp. complex (*M. edulis* and *M. trossulus*) and the horse mussel (*Modiolus modiolus*). *Mytilus* spp. (most widely recognized blue mussels) occurs from Labrador to Cape Hatteras, North Carolina, on the western Atlantic coast (Abbott 1974; Suchanek 1978, 1985; Gosling 1992, 2003; Albrecht 1998; Newell 1989; Witman and Sebens 1988; Witman and Dayton 2001; Hellou and Law 2003). In many areas, *M. edulis* and *M. trossulus* are sympatric and hybridize (Riginos and Cunningham 2005). Additionally, the occurrence of *Mytilus galloprovincialis* (originally from the Mediterranean and now cultured throughout Europe and China) and a west coast species, *Mytilus californianus*, further complicate systems as invaders in many areas (McDonald and Koehn 1988; Varvio et al. 1988; Lobel et al. 1990; Seed 1992, 1995; Geller et al. 1994; Suchanek et al. 1997; Riginos and Cunningham 2005).

Gastropods of the family Vermetidae

The only habitat-forming snails on the Atlantic coast are species in the family Vermetidae. Vermetid snails cement themselves together to form dense reefs in intertidal and shallow subtidal waters from southern New England (rarely) to the tropics (Shier 1969; Safriel 1966, 1975; Abbott 1974; Safriel and Ben-Eliahu 1991; Dame et al. 2001). These uniquely cemented gastropods feed using a mucous net.

Worldwide vermetid snails form an often-conspicuous group of sessile gastropods living in shallow tropical and temperate reefs, commonly constructed on *Crassostrea virginica* shell accumulations. In southwestern Florida they extend intermittently as far north as Sarasota. In addition, some researchers have reported that they consider the species that was found in the Ten Thousand Islands area of southwestern Florida extinct, as the reefs were formed during the last interglacial period that drowned the beach ridges that make up the present-day islands.

There are a number of reef-forming vermetid species in Florida waters. The most common Florida species of vermetid snail, *Dendropoma corrodens*, is a small (10 mm) entrenching and encrusting species that is extremely abundant in the Florida Keys. Vermetid reef formation is restricted to the west coast of Florida, involving gastropods of the genus *Petaloconchus* (e.g., *P. macgintyi*) (less than 35 mm length). This genus is gregarious, and may form large (<1 m height) reef structures in some shallow, intertidal waters (Ortiz-Corps 1985). In the Ten Thousand Islands area of Florida, longshore currents carry sand and shells to areas suitable for oysters to become established. These oyster reefs then provide stable substrate for mangroves, another important nursery habitat, to take hold (Lodge 1998). In some areas it has been hypothesized that vermetid gastropod reefs provide a similar substrate for mangrove initiation (Davis 1997). Unfortunately, some researchers note that vermetids appear to be in global decline (R. Bieler, Field Museum of Natural History, personal communication).

Aggregations of Living Shellfish

The term aggregation is used here to refer to shellfish species that are not attached to one another yet occur at densities sufficient to provide structural habitat for other organisms (Figure 3.2-14, Plate D). The term bed is also sometimes used to refer to the same type of structure. Three groups of bivalves— scallops, pen shells, and *Rangia* —form habitat in this way (Figure 3.2-14). Although not molluscan, brachiopods also form dense aggregations that function like other molluscan species. The major habitat-forming scallops that occur along the Atlantic and Gulf coasts are the bay scallop (*Argopecten irradians* with several recognized subspecies), calico scallop (*Argopecten gibbus*), and sea scallop (*Placopecten magellanicus*) (Bourne 1964; Shumway 1991; Blake and Graves 1995).

Pen shells (family Pinnidae) are large bivalves that bury partly into the substrate and are anchored by a substantial byssus (long, fine, silky filament). The upper portion of the shell protrudes above the substrate (often referred to as emergent shellfish beds), which provides habitat for other organisms when they occur in sufficient densities (Figure 3.2-14, Plates A & B). Three species of pen shell occur along the Atlantic coast of the Americas: the saw-toothed pen shell (*Atrina serrata*), the amber pen shell (*Pinna carnea*), and the stiff pen shell (*Atrina rigida*) (Abbott 1974).

The saw-toothed pen shell, *A. serrata,* is typically found in sandy mud at depths of up to 6 m. It ranges from North Carolina to Texas and northern South America, and is relatively common in many areas in North Carolina (Abbott 1974). Several studies have shown that pen shells are adept at repairing damage in a short time, pointing to potentially interesting resource allocation issues (e.g., cost of shell repair) with regard to this relatively large infaunal organism (T. Alphin, University of North Carolina at Wilmington, personal communication). Many small shrimp and crab species spend their adult lives in the mantle cavity of this species and other pen shells, where they find refuge and feed on particles brought into the mantle cavity (Abbott 1974).

Although the amber pen shell, *P. carnea,* is generally found in sandy areas with depths up to 4 m, it rarely is found in the intertidal zone. It ranges from southeastern Florida to northern South America. Finally, *A. rigida* is common in sandy muds from low intertidal to 27 m in depth. It ranges from North Carolina to southern Florida and the West Indies (Abbott 1974).

Shell Accumulations

The shells of dead molluscs sometimes accumulate in sufficient quantities to provide important habitat. The term shell hash refers to accumulations consisting mostly of pieces of broken shell

(Anderson et al. 1979; Street et al. 2005), although this hash can also be composed of intact small bivalves and gastropod shells (e.g., Sanibel Island, FL).

Shell accumulations can occur from estuaries out to the continental slope, with several species present in each zone (Stanley and Dewitt 1983, Stanley 1985, Newell and Hidu 1986, Rice et al. 1989, MacKenzie and McLaughlin 2000, Kraeuter et al. 2003). For accumulations of smaller molluscs, we know little or nothing about their importance (W. Arnold, Florida Fish and Wildlife Research Institute, personal communication).

Accumulations of eastern oyster shells are a common feature in the intertidal zone of many southern estuaries, particularly along waterways impacted by wind and boat wakes (Figure 3.2-14, Plate D) (Anderson et al. 1979; Bahr and Lanier 1981; Grizzle et al. 2002). The dead shells of blue mussels (*Mytilus* spp.) occur intertidally in some northern estuaries. These accumulations, sometimes extending well above the high tide line, have not been well studied. Subtidal shell accumulations, however, provide habitat for many species of commercially and recreationally important fish (Auster et al. 1991, 1995; Holt et al. 1998).

Ecological Role and Function

The ecological processes that depend on the above characteristics of shellfish habitat can be thought of as ecosystem services. Hence, in addition to their direct habitat-related value for managed species, shellfish habitats provide important services for the ecosystem as a whole. Three of the most important of these services are discussed in more detail below: refuge, benthic-pelagic coupling, and erosion reduction (or shoreline protection).

Refuge

The term refuge is used here to describe the protective function that shellfish habitat provides for the shellfish themselves, as well as for other organisms that occur in shellfish habitat. This ecosystem service largely results from the increase in structural complexity in shellfish habitat compared to surrounding areas (particularly soft sediments). In other habitats, such as seagrasses or salt marshes, the concept of structural complexity is often associated with the notion of ―nursery areas, which refer to places where juvenile invertebrates and fish are protected from predators (Lindberg and Marshall 1984; Heck et al. 1995; Benaka 1999; Halpern et al. 2001; Williams and Heck 2001; Beck et al. 2003; Heck et al. 2003; Minello et al. 2003). Shellfish habitat plays a role similar to seagrasses and other structurally complex habitats in this respect. Most of the research dealing with these topics for shellfish habitat has been conducted on the reef-forming species, but some information is available for shellfish aggregations and shell accumulations.

Benthic-pelagic coupling

This term refers to the transfer of materials and energy between the bottom community and the water column. It is probably most often used to refer to the overall effect of suspension feeders as they remove suspended particulates from the water column (Dame 1996). The result is a transfer of materials and energy from the water column to the benthos (Frechette et al. 1989; Meyer and Townsend 2000; Cummings et al. 2001; Dame et al. 2001; Ellis et al. 2002).

These feeding activities also typically cause a reduction in turbidity of the water column which has a positive impact on submerged aquatic vegetation (SAV), allowing more light penetration and higher rates of photosynthesis (Meyer and Townsend 2000). The shellfish release ammonia and other metabolites that are nutrients for the SAV. Therefore, SAV (Peterson and Heck 1999, 2001a, 2001b; Williams and Heck 2001; Heck and Orth 2006) and oyster reefs potentially play mutually beneficial roles (Heck 1987; Newell 1988; Dame 1996; Dame et al. 2001; Newell and Koch 2004) (also see Pomeroy et al. 2006 for a different perspective).

Oyster reefs are likely to reduce eutrophication by mediating water column phytoplankton dynamics and denitrification (Dame 1996; Newell et al. 2002; Newell 2004). A decrease in oysters in the Chesapeake Bay has led to increased phytoplankton numbers and reduced competition with zooplankton. An increase in zooplankton leads to a rise in predators, such as ctenophores and jellyfish. An increase in phytoplankton also leads to a microbial shift and anoxic conditions of deeper waters in areas such as the Chesapeake Bay (Ulanowicz and Tuttle 1992; Newell 1988) (also see Pomeroy et al. 2006 for another view). Models have shown that an increase in oyster abundance would reduce phytoplankton primary productivity and secondary gelatinous consumers (e.g., ctenophores) to historically low levels (Ulanowicz and Tuttle 1992).

Erosion reduction

Estuaries in many areas are threatened by increased coastal population growth and associated industrial, residential, and recreational development and utilization (Vernberg et al. 1999). One major area of recreational growth has been in the number of people with Class A (< 16 ft) and Class 1 (16 to 25 feet) motorized boats utilizing these waterways (NMMA 2004). Some problems related to this increase in the number of small boats have been well documented (Crawford et al. 1998; Cyr 1998; Backhurst and Cole 2000; Bauer et al. 2002; Kennish 2002). For example, increases in seagrass scarring from boat propellers and the number of marine mammal collisions are both positively correlated with increased boating activity (R. Virnstein, personal communication; Sargent et al. 1995).

However, little is known about the direct and indirect impacts of boating on other critical estuarine habitats in the landscape, such as intertidal oyster reefs (Grizzle et al. 2002; Coen and Fisher 2002; Coen and Bolton-Warberg 2003, 2005; Piazza et al. 2005; Wall et al. 2005). Those areas dominated by intertidal oyster reefs form a protective breakwater for fringing *Spartina* marshes, retarding shoreline erosion (Coen and Fischer 2002; Coen and Bolton-Warberg 2005).

Additionally, shoreline erosion in tidal channels is an issue in many states (Cyr 1998; Gabet 1998). Undercutting by wind waves and boat impacts can cause slumping (calving) of large masses of sediment embedded with *Spartina* (Gabet 1998; Chose 1999; Piazza et al. 2005). *Spartina* has been documented to be an important habitat for estuarine productivity (e.g., as a feeding ground for juvenile fishes and their prey) and is known to perform many other ecological functions, such as buffering run-off (Weinstein and Kreeger 2000).

Data collected by researchers from the South Carolina Department of Natural Resources noted significant shoreline losses at numerous study sites (n = 11) across South Carolina (Coen and Bolton-Warberg 2005). By reducing erosion, oyster reefs reduce vegetation loss and preserve other habitat types (Meyer and Townsend 2000). They also stabilize creek banks and help to reduce erosion of marshes (Meyer et al. 1997; Chose 1999; Coen and Fischer 2002; Breitburg et al. 2000; Coen and Bolton-Warberg 2003, 2005; Piazza et al. 2005), but may be easily impacted by boat wake or storm damage (Grizzle et al. 2002; Coen and Bolton-Warberg 2005).

Research on recreational boating impacts on estuarine species is surprisingly still in its infancy (Anderson 1976, 2000; Kennish 2002; Bishop 2003, 2004, 2007; Bishop and Chapman 2004). Productivity, diversity, and survival of estuaries in the southeastern United States are threatened by explosive coastal population growth and associated industrial, residential and recreational development and utilization (Vernberg et al. 1999). In spite of the potentially far excursion distances of motorboats, and the large number of boats on the water on any given day, sparse data exist to quantitatively determine the impact of boat wakes on intertidal organisms.

In conclusion, it should be noted that each of the three types of shellfish habitats differ with respect to their major characteristics and the ecosystem services they provide. Shellfish reefs typically provide the most in the way of services because they consist largely of live animals that provide a food source for many fish and invertebrates, and typically have significant vertical structure. Shellfish aggregations consist mainly of live animals but typically do not occur at densities as high, or with vertical structure as extensive, as shellfish reefs. Shellfish accumulations consist only of the dead shell remains, but they provide hard substrate and may have significant vertical structure. There is a rich literature that documents the importance of all three types of shellfish habitat to many species of fish and invertebrates, including most managed species.

Habitat utilization

Shell bottom provides critical fisheries habitat not only for oysters, but also for recreationally and commercially important finfish, other mollusks, and crustaceans. The ecological functions of oyster reefs related to oyster production are well known and accepted (Coen et al. 1999). These functions include aggregation of spawning stock, chemical cues for successful spat settlement, and refuge from predators and siltation. Oysters have also been described as ecosystem engineers that create biogenic reef habitat important to estuarine biodiversity, benthic-pelagic coupling, and fishery production (Lenihan and Peterson 1998).

Data quantifying fish use of habitats vary from presence/absence and numerical abundance, to actual fish production value. In North Carolina, 18 fishery species have been documented utilizing both natural and restored oyster reefs in Pamlico Sound, including Atlantic croaker, southern flounder, Spanish mackerel, spotted seatrout, weakfish, American eel, and black sea bass (Lenihan et al. 2001). Numerical abundance and production compared to other habitats provides additional information on the importance of habitat for fish. The species found most abundantly on oyster reefs compared to adjacent soft bottom were silver perch, sheepshead, pigfish, pinfish, toadfish, and Atlantic croaker. Southern flounder was collected on both oyster reefs and adjacent soft bottom areas, while bluefish and Atlantic menhaden were not collected near oyster reefs (Lenihan et al. 2001).

Several studies have found higher abundance and diversity of fish on shell bottom than adjacent soft bottom, particularly pinfish, blue crabs, and grass shrimp (Harding and Mann 1999; Posey et al. 1999; Lenihan et al. 2001). A study in Back Sound also found that crabs were more abundant on shell bottom than restored SAV beds (Elis et al. 1996). Breitburg (1998) concluded that the importance of shell bottom to highly mobile species is very likely underestimated, partially due to the difficulty in sampling oyster beds.

Peterson et al. (2003a) estimated the amount of fish production that shell bottom provides in addition to adjacent soft bottom habitats. Using results from numerous studies, they compared the density of fish at different life stages on oyster reefs and adjacent soft bottom habitats. The published growth rates of species were then used to determine the amount of production gained from shell bottom. The species were separated into recruitment-enhanced, growth-enhanced, and not enhanced groups. Recruitment-enhanced species are those having early life stages showing almost exclusive association with shell bottom. For other species with higher abundance in shell bottom, diet and life history studies were used to determine the fraction of their production associated with the consumption of shell bottom-enhanced species. Species consuming relatively more shell bottom-enhanced species were classified as growth-enhanced. Analysis of the studies revealed that every 10m2 of newly constructed oyster reef in the southeast United States is expected to yield a benefit of an additional 2.6 kg of fish production per year for the lifetime of the reef (Peterson et al. 2003a).

Fish that utilize shell bottom can be classified into three categories: resident, transient, and facultative (Coen et al. 1999; Lowery and Paynter 2002). Resident species live on shell bottom and depend on it as their primary habitat. Transient species are wide-ranging species that use shell bottom for refuge and forage along with other habitats. Facultative species depend on shell bottom for food, but utilize other habitats with vertical relief or shelter sites.

At least seven fish species have been identified as resident species—naked goby, striped blenny, feather blenny, freckled blenny, skilletfish, and oyster toadfish (Coen et al. 1999; Lowery and Paynter 2002). These species were also considered recruitment-enhanced by Peterson et al. (2003a). Resident fish are important prey for transient and facultative predator species (Coen et al. 1999). For example, Breitburg (1998) found high densities of juvenile striped bass (15.4 individuals/m2 of reef surface) aggregating near the reef surface feeding on naked goby larvae congregated on the down-current side of the reef. Other common predator species sampled on oyster reefs in North Carolina are red and black drum, Atlantic croaker, sheepshead, weakfish, spotted seatrout, summer and southern flounder, blue crab, and oyster toadfish. Of these species, however, only sheepshead, southern flounder, and oyster toadfish were considered shell bottom-enhanced by Peterson et al. (2003a). Production of black drum, Atlantic croaker, blue crab, and summer flounder were classified as not enhanced by shell bottom. Oyster reefs in higher salinity waters are critical habitat for predators such as juvenile gag, snappers (*Lutjanus* spp.) and stone crab (Wenner et al. 1996; Peterson et al. 2003a).

There is some variation in fish use among salinity gradients as well. Oyster reefs in higher salinity waters tend to support a greater number of associated species than reefs in lower salinity waters (Sandifer et al. 1980). Studies summarized by Coen et al. (1999), which included work in North Carolina, identified 72 facultative, resident, and transient fish species in close proximity to oyster reefs. The ASMFC-managed species categorized as transient and also important to North Carolina‘s coastal fisheries are American eel, Atlantic croaker, Atlantic menhaden, black sea bass, bluefish, red drum, spot, striped bass, summer flounder, tautog, and weakfish. Only black sea bass and tautog were considered shell-bottom enhanced by Peterson et al. (2003a).

A partial list of macrofaunal species observed in collections from oyster habitat is provided in Table 3.2-6. Those species that use shell bottom as spawning and/or nursery areas are identified, as are those species that forage on shell bottom habitat and/or use it as a refuge (SAFMC, 1998a; Lenihan et al., 1998; Coen et al., 1999; Grabowski et al., 2000). More than 30 species are listed in Table 2.6, and there are many more not listed, emphasizing the importance of shell bottom as fisheries habitat.

Table 3.2-6. Partial listing of finfish and shellfish species observed in collections from shell bottom in North Carolina, and ecological functions provided by the habitat (Source: Street et al. 2005).

Resident species, such as gobies (naked and green), Atlantic midshipman, and northern pipefish depend on shell bottom as breeding habitat (Hardy 1978a and b; Johnson 1978; Coen et al. 1999). Other species documented to spawn on shell bottom include the oyster toadfish, mummichog, sheepshead minnow, eastern oyster, grass shrimp, and hard clams (NOAA 2001). Toadfish attach their eggs to the underside of oyster shells, whereas gobies, blennies, and skilletfish place their eggs in recently dead oyster shell (Coen et al. 1999). Well-developed oyster reefs with clean oyster shells in a variety of sizes were shown to accommodate reproduction by the greatest densities of all resident species (Breitburg 1998).

Shell bottom protects oyster spat and other juvenile bivalves, finfish and crustaceans from predators. Juvenile clams, in particular, settle in shell substrate for the protection it provides (Wells 1957; MacKenzie 1977; Peterson 1982; DMF 2001b). The nursery area function of shell bottom was demonstrated by Eggleston et al. (1998) who found that juvenile blue crabs and grass shrimp were equally abundant on shell bottom and SAV in Back Sound, North Carolina. Twelve of the 18 mobile and economically important coastal fisheries species sampled by Lenihan et al. (2001) on natural and restored oyster reefs in Pamlico Sound were juveniles.

In a study where shell structure was added to mud flat reefs, juvenile fish abundance increased on the augmented reefs compared to surrounding soft bottom (Grabowski et al. 2000). The study also found that this initial increase was higher than increases that occurred when SAV and/or salt marsh were added in the same area. The ASMFC considers shell bottom as important nursery habitat for juvenile fish such as sheepshead, gag, snappers, stone and blue crabs, and penaeid shrimps (Lowery and Paynter 2002). An analysis by Peterson et al. (2003a) confirmed that sheepshead, gag, and stone crab were recruitment-enhanced, as well as many non-fishery species, including anchovies, blennies, gobies, oyster toadfish, and skilletfish.

Oyster reefs are home to many important forage species including a number of small crabs, mostly the assorted xanthids lumped under the collective name of mudcrabs, and small mussels including …. The complex community formed by the oysters, crabs and mussels is discussed in Hadley et al. 2010. Wilber et al. 2012 discuss the impacts of sedimentation on crab populations and implications for secondary consumers.

While oyster reefs are the most recognized shell bottom habitat, shell hash concentrations on tidal creek bottoms provide important nursery habitat for young fish. For example, the preferred habitat of juvenile drum species in South Carolina is high marsh areas with shell hash and mud bottoms (Daniel 1988). However, the extent of shell hash in North Carolina tidal creeks is currently unknown; known locations of shell hash include concentrations along the Intracoastal Waterway. The value of designated nursery areas could be enhanced by low-density plantings of cultch material. However, the enhancement of fish stocks provided by planting could be negated if recruitment is not limiting the adult population. The recruitment enhancement provided by low-density cultch planting in nursery areas should be evaluated.

A group of important species that are largely understudied throughout their range, but includes important members of intertidal and subtidal oyster reef communities, are the grass (Caridean) shrimp species within the genus *Palaemonetes*. Grass shrimp are found in large numbers in estuarine waters along the Atlantic and Gulf coasts, where they occur from Massachusetts to Texas. They are a very common estuarine species in southeastern marshes and tidal creeks where they are usually associated with beds of submerged or emergent vegetation, oyster reef habitats, or structures such as oyster shell, fouling communities, woody debris (Ruiz et al. 1993), and docks or pilings (Coen et al. 1981). Caridean shrimp are rarely larger than 5 cm; their small size differentiates them from commercial shrimp, such as the penaieds and pendalids. Grass shrimp are an important species from an ecological perspective because they are instrumental in transporting energy and nutrients between trophic levels in the coastal food web. Grass shrimp are consumed in large quantities by commercially important fishes and forage species, including spotted seatrout, red drum, and mummichogs (*Fundulus heteroclitus*) (Heck and Thoman 1981; Anderson 1985; Wenner et al. 1990; Posey and Hines 1991; Wenner and Archambault 1996).

Although there are no estimates of population sizes of grass shrimp, they are amongst the most widely distributed, abundant, and conspicuous of the shallow water benthic macroinvertebrates in our estuaries, often reaching hundreds to thousands per square meter (Leight et al. 2005; Coen and Luckenbach 2000; Coen et al. 2006a). Grass shrimp can inhabit very shallow areas near the margins of intertidal habitats (e.g., marsh, mudflats, oyster reefs), but have been reported at depths as great as 15 meters. In winter during temperature lows, and in summer when water temperatures approach seasonal highs, daggerblade grass shrimp may move from shallow to relatively deeper water. The extent of the movement of grass shrimp among various depths often coincides with the distribution of oyster shell substrates, which, in some waters, are preferred by both *P. vulgaris* and *P. pugio*. They are abundant in these structured estuarine and marine habitats as shellfish habitats provide abundant food and protection from predators (Thorp 1976; Coen et al. 1981; Heck and Thoman 1981; Heck and Crowder 1991). Consequently, the association of shellfish habitats with primary producers and consumers may prove quite significant, given the importance of low trophic level species as food for managed species.

Shell bottom provides important foraging area for a variety of aquatic organisms. Fish, shrimp and crabs forage on the worms, algae, crustaceans, mollusks, and other invertebrates present on and in shell bottom habitat. Concentrations of prey organisms among the shell attract both specialized and opportunistic predators. Eggs from oysters and other organisms, and larvae from species belonging to the oyster shell bottom community, are eaten by protozoans, jellyfishes, ctenophores, hydroids, worms, mollusks, adult and larval crustaceans, and fishes (Loosanoff 1965). Blue crabs forage heavily on oyster reefs (Menzel and Hopkins 1955; Krantz and Chamberlin 1978; Mann and Harding 1997). Stomach contents of common finfish predators sampled near shell bottom in Middle Marsh, North Carolina, included fish, shrimp, tanaids, amphipods, isopods, polychaetes, bivalves, gastropods, and tunicates, as well as plant, algal and detrital material (Grabowski et al. 2000).

Grabowski et al. (2000) calculated an index of reef affinity (association) for fish species and analyzed the relative proportion of stomach contents originating from oyster reef versus non-reef habitats. Results showed:

Pigfish and pinfish foraged more on reefs (amphipods, bivalves, gastropods and polychaetes).

The ubiquitous spot foraged on both reef and non-reef habitats.

Gulf and southern flounder foraged on species slightly more common on reefs.

Blacktip sharks, spotted seatrout, and bluefish exhibited a feeding preference for oyster reef prey (fish, shrimp and crabs).

Red drum foraged slightly more off reefs.

Blacknose sharks rarely foraged on reef habitats.

The growth-enhanced species/groups identified in Peterson et al. (2003a) included sheepshead minnow, silversides, pigfish, southern flounder, and black sea bass. These results differ somewhat from those of Grabowski et al. (2000). The discrepancies between Peterson et al. (2003a) and Grabowski et al. (2000) could be due to regional differences in fish habitat use, or other unknown factors. Sheepshead also have an affinity for slow or sessile invertebrates found abundantly on shell bottom (Pattilo et al. 1997).

Oyster reefs are also a foraging ground for many juvenile and adult turtle species. Schmid (1998) found that both the Kemp‘s ridley and loggerhead sea turtles feed on organisms that inhabit the reef. Kemp‘s ridley turtles feed on the stone crabs (*Menippe* spp.) and blue crabs (*Callinectes sapidus*) found near the reef‘s surface. Loggerheads also feed on molluscs. Schmid (1998) also found that Kemp‘s ridleys will return to the same oyster reef for up to four years.

Another important species that utilizes intertidal and subtidal oyster reefs as foraging grounds is the blue crab, *Callinectes sapidus* (Coen et al. 1999b). Blue crabs forage heavily on oyster reefs (Mann and Harding 1997; Krantz and Chamberlin 1978), including consuming oyster spat as juveniles. A study by Menzel and Hopkins (1955) showed that juvenile blue crabs consumed as many as 19 juvenile oysters (or spat) per day.

Numerous mammals and birds directly and indirectly utilize intertidal oyster reef habitats and washed oyster shell accumulations, particularly along the IWW (Sanders et al. 2004). These include *Procyon lotor* (raccoon), and birds such as *Haematopus palliates* (American oyster catcher), *Egretta tricolor* (Tricolored Heron), *Nyctanassa violacea* (Yellow-crowned Night Heron), *Nycticorax nycticorax* (Black Heron), *Casmerodius albus* (Great Egret), *Egretta thula* (Snowy Egret), *Limosa fedoa* (Marbled Godwit), *Catoptrophorus semipalmatus* (Willet), *Pluvialis squatarola* (Black-bellied Plover), *Calidris pusilla* (Semipalmated Sandpiper), *Calidris mauri* (Western Sandpiper), *Arenaria interpres* (Ruddy Turnstone), *Tringa melanoleuca* (Greater Yellowleg), and *Tringa flavipes* (Lesser Yellowleg). Some observations in SC suggest that a single oystercatcher may be able to consume over 100 adult oysters per day on intertidal reefs (F. Sanders, South Carolina Department of Natural Resources, personal communication).

Corridor and Connectivity

Shell bottom serves as a nearshore corridor to other fish habitats, such as salt marsh and SAV for finfish and crustaceans; therefore, it plays a significant ecological role in landscape-level processes (Coen et al. 1999; Micheli and Peterson 1999). Vicinity (isolation) and connectivity of intertidal oyster reefs to other fish habitats, especially SAV, are two factors that affect fish utilization of shell bottom. For example, connectivity of oyster reefs to SAV enhanced blue crab predation, whereas isolation of oyster reefs enhanced hard clam survivorship (Micheli and Peterson 1999). In Middle Marsh, North Carolina, gag, gray snapper, and spottail pinfish preferred shell bottom habitat adjacent to SAV beds (Grabowski et al. 2000), allowing access to both refuge and prey.

Species composition and community structure

Eastern oyster (*Crassostrea virginica*)

The eastern oyster's range extends from the Gulf of St. Lawrence to Key Biscayne, and south to the West Indies and the Yucatan Peninsula in Mexico (Galtsoff 1964; Burrell 1986; Kennedy 1996; MacKenzie et al. 1997a). The eastern oyster is mainly an estuarine organism, but does occur in some near-shore coastal waters. These oysters grow sub-tidally throughout most of their range, but from southern North Carolina to northeastern Florida they occur predominantly in the intertidal zone (Figure 2.14) (Bahr and Lanier 1981; Kennedy 1996; Kennedy and Sanford 1999; Burrell 1986, 1997; Coen and Luckenbach 2000; Luckenbach et al. 2005). Although they occur to a depth of 30 m, the oyster's primary habitat is in shallow water less than 6 m, or intertidal (1 m to 5 m) from North Carolina to Florida. A typical feature of *C. virginica* is their extremely variable shell morphology (Galtsoff 1964; Carriker 1996; Kent 1992). Oysters have indeterminate growth; in historical times, prior to the influence of harvesting and other biological and anthropogenic factors, they often grew to sizes significantly greater than what we see today (20 cm or larger shell height).

The preferred substrate for larval settlement is oyster shell, an adaptation that assures the proximity of other oysters, which is essential for successful future reproduction. Oysters are attached to the substrate or to each other by the left valve, which tends to be thicker and more deeply cupped than the right valve (Galstoff 1964; Kennedy 1996; Soniat et al. 2004). Thus, dense reefs are formed by the setting of successive generations of oysters on the shells of their predecessors (Figure 3.2-14). In some places, oyster shell can be several meters deep or more with live animals only on the surface layer.

Long-term reef development is a complex process that involves interactions among a variety of physical and biotic factors (Bahr and Lanier 1981; Kennedy and Sanford 1999; Coen and Luckenbach 2000). In southern Atlantic waters, a reef-like structure may be achieved in three to five years, but in northern waters the process is apparently much slower. The long-term dynamics of oyster reefs have not been well studied, but some reefs in the Chesapeake Bay have persisted for millennia (Smith et al. 2003). In part because estuaries are geologically ephemeral, oysters must cope with changes in sea level, sediment, and climate. In contrast, within the past 50 years, some intertidal reefs in Florida have been completely destroyed and displaced landward by dredging and/or boat wakes (Figure 3.2-15). Hurricanes have also been implicated in a few instances for example, in the destruction of the windrows of shell in surf troughs along the Florida coast (Livingston et al. 1999; Grizzle et al. 2002; Walters et al. in press). Elsewhere, hurricanes may have significant impacts on shellfish habitats, particularly in shallow waters (Andrews 1973; Munden 1975; Lowery 1992; Dugas et al. 1998; Livingston et al. 1999; Perret et al. 1999). Bartol and Mann (1997) observed an increase in oyster survival when oysters settled in the interstitial spaces between shells below the reef surface. Additionally, vertically growing oysters in clusters on intertidal reefs provide oysters with a way to cope with siltation, so that they are not smothered (Coen et al. 1999a; Giotta 1999

Figure 3.2-15. Time series of intertidal oyster reef changes in east-central Canaveral National Seashore (CANA), Florida (Source: ASMFC 2007). Aerial imagery showing increase in dead reef areas (red) compared to living (green) over time, most probably caused by increased boating activities (Source: Grizzle et al. 2002).

Caribbean mangrove oyster (*Crassostrea rhizophorae*)

The Caribbean mangrove oyster is restricted to the south Atlantic and Gulf coasts (Abbott 1974) and does not typically form reefs. *C. rhizophorae* is well adapted to the warmer tropical and subtropical temperatures in its native range (Bacon et al. 1991). *C. virginica* and *C. rhizophorae* oysters are closely related species (Buroker et al. 1979; Hedgecock and Okazaki 1984). Mangroves are typically the primary hard substrate for attachment of these often common and flat oysters. Numerous other species of mangrove oysters have been described, all in the genus *Crassostrea*. For all these species, information is extremely limited, with even less known on how they may enhance habitat complexity along the southern coast of Florida. *C. rhizophorae* is commercially important, can grow to marketable size (50 -70 mm shell height) in 4 to 8 months (Rodriguez and Frias 1992), and is currently cultivated in aquaculture facilities in the Caribbean (Littlewood 1988; Bacon et al. 1991; Newkirk and Field 1991).

Currently, there is very little information on Caribbean mangrove oyster ecology (i.e. densities, filtering, etc.) or potential habitat value for other Florida mangrove-related species. However, it must be noted that the species adds considerable habitat to the recognized three-dimensional mangrove fish nurseries of the Caribbean (L. Stewart, University of Connecticut, personal communication). Presumably Caribbean mangrove oyster reefs are fouled by many different planktonic plant and animal species, thus providing a critically needed substrate for attachment.

In large part resulting from work on *Crassostrea ariakensis* in North Carolina (Grabowski et al. 2003, 2004; NRC 2004; Bishop et al. 2006; Carnegie et al. 2006; R. Carnegie, Virginia Institute of Marine Science, personal communication), researchers have begun to examine the dynamics of poorly studied native oyster species, such as the crested oyster (*Ostreola equestris*). Additional attention has been drawn to novel or endemic *Bonamia* spp. (newly described or observed) that may cause diseases in native or non-native species, or act as parasite reservoirs (Bishop et al. 2006; Carnegie et al. 2006; R. Carnegie, Virginia Institute of Marine Science, personal communication).

Blue mussels *(Mytilus spp.)*

Mytilus spp. occur mainly in shallow coastal waters and estuaries, and are most commonly considered a member of the fouling community because they are often found on rocks, pilings, and other hard substrates (King et al. 1990; Mathieson et al. 1991; Leichter and Witman 1997; Bertness 1999; Witman and Dayton 2001). In many areas mussels play an important role in benthic community structure (Bayne 1976; Witman 1985, 1987; Asmus and Asmus 1991; Lesser et al. 1991; Dame 1993, 1996; Hild and Günther 1999; Norén et al. 1999; Davenport et al. 2000). In some areas mussels also form dense reefs on hard bottom or on soft sediments in the intertidal and subtidal zones (Newell 1989; Nehls and Thiel 1993; Seed and Suchanek 1992; Seed 1996; Côté and Jelnikar 1999; Cranford and Hill 1999). Blue mussel reef formation and development have not been well studied, but they are recognized as being important food and habitat providers for many species (Tsuchiya and Nishihira 1985, 1986; Witman 1985, 1987; Newell 1989; Asmus and Asmus 1991; Seed 1996; Reusch and Chapman 1997; Ragnarsson and Raffaelli 1999). Mussel consumers include crabs, lobsters, starfish, whelks, fish (e.g., tautog), and birds (e.g., ruddy turnstone, American and European oystercatchers) (Marsh 1986; Meire and Ervynck 1986; Raffaelli et al. 1990; Marsh and Wilkinson 1991; Nol and Humphrey 1994; Nagarajan et al. 2002; Sanders et al. 2004). Mussel reefs perform essentially the same functions as oyster reefs; they provide food, filtration, benthic-pelagic coupling, and physical habitat (Verwey 1952; Suchanek 1978, 1985; Wildish and Kristmanson 1984, 1997; Witman and Suchanek 1984; Dame 1996; Smaal and Hass 1997).

Horse mussel *(Modiolus modiolus)*
The horse mussel has a geographic distribution similar to the blue mussels, but occurs mainly in deeper waters on the continental shelf; however, it can be found in intertidal pools or attached to laminarian holdfasts (Holt et al. 1998). It is a widespread mussel, found throughout the northern hemisphere from the White Sea and Norway, off the Faroes and Iceland to at least as far south as the Bay of Biscay and occasionally North Africa. It is also found from Labrador to North Carolina in the Atlantic and from the Bering Sea south to Japan and California in the Pacific. It most commonly occurs partly buried in soft sediments, or attached by byssal threads to hard substrates where it forms clumps or extensive beds (or reefs) that vary in size, density, thickness, and form (Holt et al. 1998; Wildish et al. 1998).

Horse mussel recruitment is often low and may be variable in some populations (JNCC UK 1999). *M. modiolus* is a long-lived species, with some individuals living for 25 years or more. Juvenile *M. modiolus* are heavily preyed upon, especially by crabs and starfish, until they are 3 to 6 years old, at which point they normally reach a size refuge from most of their native predators.

American horse mussel *(Modiolus americanus)*
The American horse mussel is a common mussel that often forms dense associations within seagrass habitats (Figure 3.2-14, Plate C) (Peterson and Heck 1999, 2001a, 2001b). It ranges from South Carolina to the Gulf of Mexico and south to Brazil; it is also found in Bermuda. Adults can reach 100 mm shell height and they occur from the intertidal to approximately 6 m water depth. The American horse mussel can be found in densities as high as 2,000 individuals/m2 with mean densities reaching 625 individuals/m2 (Valentine and Heck 1993). However, these aggregations of American horse mussels are typically quite patchy (L.D. Coen, personal observation). Little is known about the broader ecological importance of the facultative mutualistic association of seagrass and shellfish, but work in St. Joe Bay, Florida in dense seagrass beds has shown a more complex interaction between these abundant filter-feeders and the *Thalassia* beds within which they reside. Specially, the mussels increase seagrass productivity through their filtering activities, changing nutrient availability through mechanisms such as biodeposition and reducing epiphyte loads on seagrasses (L. Coen, personal observation).

Ribbed mussel (*Geukensia demissa*)

The ribbed mussel is a relatively large mussel, growing to nearly 100 mm shell height. The ribbed mussel is found in coastal waters from the Gulf of St. Lawrence to Texas. It is common on both subtidal and intertidal oyster reef habitats (Van Dolah et al. 1999; Coen et al. 2004b; Luckenbach et al. 2005) and in salt marsh (Bertness 1980, 1984; Lutz and Castagna 1980; Bertness and Grosholz 1985). Unlike oysters, ribbed mussels have the ability to reattach if dislodged, which makes this species better able to adapt following a disturbance event.

The basic biology of the ribbed mussel is well understood, but little is known about its habitat value either alive or as dead articulated shells (Lent 1969; Seed 1980; Brousseau 1984; Kraus and Crow 1985; Hilbish 1987; Lin 1989a, 1989b, 1990, 1991; Wilbur and Hilbish 1989; Kemp et al. 1990; Langdon and Newell 1990; Sarver et al. 1992; Stiven and Gardner 1992; Franz 1993, 1996, 1997, 2001; Nielsen and Franz 1995; Kreeger and Newell 2000). Ribbed mussels attach by byssal threads to any hard substrate (like oyster shells and cordgrass stems) and protrude above the surface. Typically, ribbed mussels occur embedded in and amongst salt marsh sediments attached by byssal threads to each other and/or to *Spartina* spp. stalks. Angelini et al. (2015) discuss the role of ribbed mussels in enhancing biodiversity and multifunctionality in southeastern saltmarshes.

Ribbed mussels occur throughout the mid- to low-intertidal regions in most southeastern estuaries. Upper intertidal limits are determined by both exposure to high temperatures and limited food availability during longer periods of tidal exposure. Lower intertidal limits are determined by the availability of effective refuge, mainly from crab predators. Although growth rates decline at higher shore levels, this is offset by increased survival (Bertness 1980; Bertness and Grosholz 1985; Stiven and Gardner 1992; Franz 2001).

A large volume of literature exists for ribbed mussels associated with salt marsh habitats on the east coast of the United States; however, much less is known about this mussel‘s association with oyster reefs. Researchers in South Carolina and Virginia (Coen et al. 1999a; Coen and Luckenbach 2000; Luckenbach et al. 2005) have noted large numbers of ribbed mussels often associated with intertidal and subtidal oyster reef habitats. In South Carolina, there are *G. desmissa* densities of over 500 individuals/m2, cohabiting areas with one or more smaller (2.5 to 5 cm) mussel species (e.g., scorched mussel (*Brachidontes exustus*) and hooked mussel (*Ischadium recurvum*)). Scorched and hooked mussels can also occur at high densities, often exceeding ribbed mussel densities (L. Coen, personal observation). For example, at some restored South Carolina intertidal oyster sites, *B. exustus* densities exceeded 4,900 individuals/m2 and *I. recurvum* densities reached 500 individuals/m2. As a result of these high densities of individuals, mussels can be a significant nuisance species at many Gulf of Mexico oyster reef sites.

Recent emergence of a ribbed mussel fishery in South Carolina raises concerns about habitat damage that could result from widespread harvesting. Ribbed mussels have not been managed as a fishery in any southeastern states although South Carolina is considering a ribbed mussel management plan.

Green mussel (*Perna viridis*)
The green mussel is an invader to the Caribbean, Florida (Benson et al. 2001; Baker and Benson 2002), and Georgia (Power et al. 2004), reaching lengths up to 171.5 mm (J. Fajans, University of Florida, personal observation). This species should not be confused with two morphologically similar alien species, *P. perna* and *P. canaliculus* (Siddall 1980; Benson et al. 2001; Ingrao et al. 2001). Although the green mussel is overgrowing oyster reefs in Florida (Figure 3.2.16), and becoming a serious fouling problem in Florida and Georgia, it may ultimately generate a complex and important habitat not previously observed in the southeast (J. Fajans and S. Baker, University of Florida, personal communication). Collections (October 2006) in Charleston, South Carolina (D. Knott, South Carolina Department of Natural Resources, personal observation), collected *P. viridis*, resulting in a new northern range extension for this non-native fouling mussel species.

Figure 3.2-16. The green mussel, *Perna viridis* (Source: Jon Fajans, Keys Marine Lab, Long Key, Florida) (Source: ASMFC 2007).

Bay scallop (*Argopecten irradians*)
Bay scallops are found on the Atlantic and Gulf coasts from the north shore of Cape Cod, Massachusetts to Laguna Madre, Texas (Waller 1969; Fay et al. 1983). They can reach a maximum size of 60 to 70 mm. Seastars, wading birds, gulls, pinfish, lightning whelks, cow-nosed rays, crabs, starfish, and humans are among the numerous predators of the bay scallop (Peterson et al. 2001a). Scallops are hermaphroditic, with a single individual releasing sperm before eggs (Bricelj et al. 1987). Bay scallops reach sexual maturity within one year, spawning from August through October. The juvenile stage is reached after about 35 days post-fertilization, when they resemble a small adult in shape; their lifespan is less than two years (Peterson et al. 1989).

Bay scallops can migrate *en masse*. In many areas they have declined significantly (e.g., North Carolina). Red tides, often referred to as ―harmful algal blooms, can kill millions of adult and larval bay scallops each year. Scallops grow fastest during the warmer months when food is available. They prefer estuaries and bays where salinities are relatively high, waters are 0.3 to 0.6 m deep at low tide, and seagrasses such as eelgrass (*Zostera marina*) or shoal grass (*Halodule wrightii*) are common (Smith et al. 1988; Prescott 1990; Pohle et al. 1991; Garcia-Esquivel and

Bricelj 1993; Bologna and Heck 1999, 2002; Bologna et al. 2001). These grass beds offer protection from predators as well as sites for juvenile attachment (Pohle et al. 1991; Bologna and Heck 1999).

Atlantic calico scallop *(Argopecten gibbus)*

The Atlantic calico scallop, a relatively small scallop ranging from 25 to 60 mm shell height, is patchily distributed on the Atlantic coast from Delaware Bay south into the Caribbean Sea to about 20° N latitude. It is most commonly found from just north of Cape Hatteras, North Carolina to the Greater Antilles, and throughout the Gulf of Mexico and Bermuda (Allen and Costello 1972; Blake and Moyer 1991). Genetic and morphological similarities (Waller 1969) between Florida and North Carolina populations and coastal currents support a hypothesis that Florida may be an important larval source for North Carolina stocks (Wells et al. 1964; Krause et al. 1994). Calico scallops can be found in depths of 10 to 400 m, but have been reported from shallower waters in Biscayne Bay (Coleman et al. 1993).

Spawning occurs throughout the year, but peaks in late fall and in the spring (Arnold 1995). As with bay scallops, calicos are simultaneous hermaphrodites that release sperm and eggs. Settling calico scallops require shell or other hard substrate to provide an anchor for byssal attachment. Laboratory studies suggest that after drifting freely for 14–16 days, larvae attach to hard substrates, which are often the disarticulated shells (dead accumulations that are separated or broken) from previous generations (Ambrose and Irlandi 1992; Ambrose et al. 1992). They reach a commercial length of 47 to 53 mm in six to eight months.

The maximum life span of an Atlantic calico scallop appears to be about 24 months. Predation (Wells et al. 1964) is a major factor affecting survival during various phases of the calico scallop life cycle. Aggregations of calico scallops provide habitat for numerous species, including other types of scallops, fish, and invertebrates. Schwartz and Porter (1977) collected 111 species of fish and 60 species of macroinvertebrates, including 25 crustaceans, 12 echinoderms, 4 coelenterates, and 1 annelid. Many of the fish caught used this habitat for feeding purposes (Schwartz and Porter 1977). See section 4.1.9 in this document for more detailed information on this species.

Pen Shells

As with other filter feeders, pinnids can filter large quantities of suspended sediments and plankton out of the water column, thereby affecting phytoplankton levels and water clarity. However, high densities generate both feces and pseudofeces affecting the surrounding sediments and associated organisms (Cummings et al. 2001; Ellis et al. 2002). For example, Ellis et al. (2002) showed that sedimentation can significantly impact *Atrina* spp. populations.

All three species -- the saw-toothed pen shell (*Atrina serrata*), the amber pen shell (*Pinna carnea*), and the stiff pen shell (*Atrina rigida*) -- can occur in large numbers and protrude above the sediment's surface (Figure 3.2-14, Plates A & B). Their shells are typically covered with a diverse assemblage of fouling organisms, including barnacles and slipper shells, which create vertical structure and fish habitat (Kuhlmann 1994, 1996, 1997, 1998; Munguia 2004). Many organisms use the shells as shelter, including crabs (e.g., *Pilumnus sayi*, *Menippe* spp., *Portunus ordwayi*) and benthic fishes such as blennies and gobies) within seagrasses (Kuhlmann 1994). Shells can reach densities of over 13 individuals/m2 (Kuhlmann 1994, 1996).

Additionally, the Florida blenny (*Chasmodes saburrae*), feather blenny (*Hypsoblennius hentzi*), clingfish (*Gobiesox strumosus*), and Gulf toadfish (*Opsanus beta*) use dead pen shells as nest sites (Kuhlmann 1994). Females lay a single layer of eggs on the inside of the pen shells. Similarly, Joubin's pygmy octopus (*Octopus joubini*) also lays its eggs on the inside of pen shells. Horse conchs (*Pleuroploca gigantea*) are the primary predators of pen shells (Kuhlmann 1994, 1996, 1997, 1998). Dead pen shells provide nesting sites and shelter for many fish species, but are not permanent benthic features. As the shells begin to break apart, the waves and currents sweep them away, thus changing the dynamics of the populations of the species that depend on them (Kuhlmann 1996, 1998).

The most extensive studies of pen shell communities as habitat were completed by researchers in New Zealand (Keough 1984; Cummings et al. 1998, 2001; Nikora et al. 2002; Gibbs et al. 2005). These habitats are also referred to as horse mussel (*Atrina zelandica* and *Atrina novaezelandiae*) beds. Research has included fine scale boundary layer flow studies (Nikora et al. 2002), mesoscale hydrodynamic interactions (Green et al. 1998), community interactions (Keough 1984; Cummings et al. 1998, 2001), and essential fish habitat delineation for juvenile finfish species (Morrison and Carbines 2006).

Estuarine wedge clam (*Rangia cuneata*)

The estuarine wedge clam is found in Atlantic coastal and Gulf of Mexico oligohaline estuaries (Cain 1975; LaSalle and de la Cruz 1985; Abadie and Poirrier 2000), tidal rivers, and backwater bays with regular inputs of fresh water. It occurs from the upper Chesapeake Bay to Mexico, often dominating benthic biomass in low salinity areas of estuaries (Cain 1975). This clam is regarded primarily as a subtidal species found in coastal areas with a large tidal range (Estevez 2005).

The species serves as an important link in the food chain, filtering large volumes of water when at high densities and serving as a food source for fish, crabs, and ducks (LaSalle and de la Cruz 1985). In North Carolina, *Rangia cuneata* are often found within the most critical oyster habitat areas where shells accumulate over long time periods. In these areas, accumulations of estuarine wedge clam shells provide substrate for formation of oyster reefs. In a majority of cases, both living and dead *Rangia cuneata* occur together. Estuarine wedge clams are more abundant in downstream reaches and as intertidal material in upstream reaches. Interestingly, live *Rangia cuneata* in intertidal areas can be larger than those in subtidal beds (Estevez 2005).

In Lake Pontchartrain, Louisiana, individual estuarine wedge clams have an average life span of four to five years. Deposits of wedge clam shells in the lake bottom supported a shell mining industry from 1933 to 1990 (Abadie and Poirrier 2000). As with oyster shells, clam shells used to be so abundant that they were used for construction of roadways, parking lots, levees, and in the production of cement. Large (> 20 mm) *Rangia cuneata* were abundant in Lake Pontchartrain in the early 1950s, but became rare by the 1970s and 1980s. They can dominate the benthos, with densities reaching 1,896 clams/m2 and dry weight biomass as high as 70 g/m2. However, clams are absent from areas that are subject to anoxia and hypoxia, or saltwater intrusions (Poirrier and Spalding 2005).

R*angia cuneata* studies are seeking to document similar ecological services to oysters, in order to generate interest in its restoration (M. Poirrier, personal communication). Results indicate that increasing clam abundance by decreasing saltwater intrusion will improve water clarity; this in turn should increase submerged aquatic vegetation and add shell for mud stabilization and erosion reduction. These improvements should reduce eutrophication, improve water quality, and enhance fish habitat (M. Poirrier, personal communication).

Carolina marsh clam (*Polymesoda caroliniana*)

This brackish-water corbiculid clam (often reaching sizes over 50 mm, but typically 25- 40 mm) is often common in low salinity marshes comprised of plants such as *Juncus* sp. and near river mouths (Andrews and Cook 1951; Andrews 1977; Duobinis-Gray and Hackney 1982; Marelli 1990). The geographical range of this species is from Virginia through Florida along the Gulf of Mexico to Texas, with adult densities often exceeding 300 individuals/m2 (Duobinis-Gray and Hackney 1982) and juvenile (<20 mm) densities at almost 2,000/m2 (Marelli 1990). The Carolina marsh clam lives primarily in the intertidal zone (Marelli 1990), but may be found subtidally, in mud to fine sediments (Heard 1982). Some researchers have suggested competitive interactions with another common low salinity bivalve, *Rangia cuneata* (more often subtidal, as *Polymesoda* is a poor burrower in intertidal areas) (Duobinis-Gray and Hackney 1982). Early growth can be rapid (>1 mm/month) (Olsen 1973, 1976), and predation, competition, and inundation are often cited as factors controlling the distribution and abundance of this species (Andrews and Cook 1951, Andrews 1977). A related species *P. maritima*, the Florida marsh clam, is common in the Gulf coast region, and southern Florida to the Yucatan (Andrews 1977). Little is known about the habitat value of shell accumulations or live aggregations of *Polymesoda* spp. for other organisms.

Oyster Reefs and Shell Banks as Essential Fish Habitat

The three major types of shellfish habitat (reefs, aggregations, and accumulations) differ in their combinations of habitat characteristics. However, all shellfish habitats have three major features in common that are the basis for their ecological value for managed species: hard substrate (for settlement/refuge/prey), complex vertical (3-D) structure (for settlement/refuge/prey), and food (feeding sites for larger predators).

Perhaps the most fundamental characteristic of shellfish habitat is hard substrate. The shells provide attachment surfaces for algae and sessile invertebrates, such as polychaetes (e.g., sabellids, serpulids), hydroids, bryozoans, and sponges, which in turn provide substrate for other organisms. Planktonic larvae of some shellfish species, such as oysters, need a hard substrate on which to settle in order to grow into adults (Galtsoff 1964). In many estuarine areas, oyster shell and cultch are the primary settlement material for larval oysters (Kennedy 1996; Powell et al. 2006). All three types of shellfish habitat—reefs, aggregations, and accumulations—provide suitable substrate for other shellfish and many other species that require hard substrate on which to grow.

Sufficient accumulations of hard substrate result in complex habitat structure that provides increased vertical relief and internal complexity of the structure itself. Structural complexity has historically been considered an important factor affecting the spatial distribution and diversity of marine and estuarine organisms (Bell et al. 1991). An increase in the physical complexity of an environment is typically correlated with an increase in microhabitat diversity (Sebens 1991). The increase in surface area provides more refuge and feeding sites, which subsequently leads to greater species richness (Bell and Galzin 1984). The interstitial spaces provide recruiting oysters with adequate water flows for growth and refuge from predators, both of which are essential for long-term maintenance of the reef structure (Bartol and Mann 1997; Bartol et al. 1999; Coen et al. 1999b; Powell et al. 2006). Oysters and other reef-forming shellfish can be considered bioengineers because they create habitat that allows many additional species to thrive (Jones et al. 1994, 1997).

All three shellfish habitat types provide food for other organisms, whether it is the shellfish themselves or associated organisms. Oysters and mussels are consumed by many species of fish and invertebrates. Many other species of plants and animals also occur on shell accumulations and provide food for a variety of predators. When considered in combination with the hard substrate and complex structure provided by live shellfish, their direct food value results in shellfish reefs and aggregations being uniquely valuable habitat for many managed species.

SAFMC Fishery Ecosystem Plan II: Seagrasses

3.2.3 Seagrasses

Description and Distribution

Out of the estimated 250,000 flowering plants existing on earth today, only about 50-60 species have adapted to life in the marine environment (den Hartog 1970; Hemminga and Duarte 2000; Green and Short 2003; Larkum et al. 2006). Collectively, we refer to this group of submersed aquatic vegetation (SAV) as seagrasses. Seagrasses are clonal plants which reproduce and disperse by means of sexual and asexual reproduction. Seaweeds (macroalgae) are often mistakenly referred to as ―grasses. Despite the fact that they frequently co-occur and provide similar ecological services, these two plant taxa have distinctly different growth forms and contrasting environmental requirements, the most important of which is the fact that seagrasses anchor themselves in unconsolidated sediments with an extensive root and rhizome system, thus have a very significant influence on sedimentary processes and nutrient cycling. Only one seagrass genus, *Phyllospadix*, does not require unconsolidated sediments and this species does not grow in the South Atlantic.

Taxonomically, seagrasses are divided into two families and 12 genera (den Hartog 1971; Phillips and Meinez 1988; Green and Short 2003). At least 13 species of seagrass occur in United States waters. In the south Atlantic region, with the exception of Georgia and South Carolina where highly turbid freshwater discharges, suspended sediments and large tidal amplitude combine to prevent their permanent establishment, there are 6 genera of seagrasses represented by 8 species. These species range in size from the three smallest, *Halophila decipiens* (paddle grass), *Halophila engelmannii* (star grass) and *Halophila johnsonii* (Johnson‘s seagrass), to the relatively larger species, *Zostera marina* (eelgrass), *Ruppia maritima* (widgeon grass), *Halodule wrightii* (shoal grass), *Syringodium filiforme* (manatee grass) and *Thalassia testudinum* (turtle grass) (Figure 3.2-4).

In the South Atlantic, seagrass habitat occurs in North Carolina and Florida, with Florida having the greatest amount of seagrass habitat (Figure 3.2-5). Along the Atlantic Peninsula and South Florida regions of Florida, there are an estimated 29,769 hectares (ha) and 574,875 ha of seagrass beds, respectively (Madley et al. 2003). The South Florida total includes seagrass in Florida Bay and the continental shelf off of the Keys (Florida Straits). Seagrass estimates in the Florida Straits include areas with continuous SAV as well as areas where SAV is patchy and intermixed with hardbottom. Along the Atlantic Peninsula, seagrasses are most concentrated in the Indian River Lagoon system. This area, while only supporting approximately 3% of the total seagrass coverage along all of Florida, has the highest seagrass diversity, with seven species present (*Zostera mariana* does not occur in Florida), including the federally threatened species,

Comment [1]: SAV stands for Submerged Aquatic Vegetation (not Vascular) and includes marine, estuarine, and riverine species. The term seagrass refers only to the marine/estuarine species. Will need consistent usage throughout.

Comment [2]: Judd Kenworthy: We need to reconsider this term SAV despite whatever terminology is accepted. This is because we need to consider where to draw the line for consideration of submerged plants with respect to salinity. I think we need to carefully distinguish between was are referred to as SAV (many species) in low salinity vs. seagrasses in high saliny. EFH is very different with respect to submerged plants and salinity. How far upstream is EFH going to have jurisdiction. We should consider re-writing this introductory section.

Comment [3]: Judd Kenworthy: But there may be SAV in some of these systems, just not seagrass

Comment [4]: This is different than an actual seagrass meadow. The term seagrass habitat connotes that an area has the potential for seagrass colonization. This is oftentimes referred to as 'potential seagrass habitat'. I would just be careful in qualifying the word seagrass with the word 'habitat'. This qualifier is not used for other marine/estuarine habitats (i.e., you rarely see 'coral reef habitat', 'mangrove habitat' etc.

Comment [5]: There is no Atlantic Peninsula of Florida, the entire state is the peninsula.

Comment [6]: Along the Atlantic coast shoreline and southernmost regions of Florida...

Comment [7]: coast of the peninsula ...

Comment [8]: marina

Halophila johnsonii (Johnson's seagrass) (FFWCC 2003). Over half of all seagrass habitat in Florida occurs in South Florida and Florida Bay supports the largest contiguous seagrass beds in the world with *Thalassia testudinum* (turtle grass) being the most dominant species. On the Atlantic side of the Florida Keys, seagrass habitat is closely associated with hardbottom, patch reefs, and mangroves (FFWCC 2003). North Carolina has the second largest seagrass distribution in the continental United States with an estimated 54,230 ha mapped (Ferguson and Wood 1994). This number includes primarily seagrasses and a small amount of visible oligohaline SAV along the western Pamlico and Albemarle tributaries. Unlike Florida, the seagrass species growing in North Carolina, *Z. marina*, *H. wrightii* and *R. maritima*, are all found within coastal lagoons, protected inland waterways and river mouths all protected by barrier islands. A unique feature of NC seagrasses is the overlap in distribution of a temperate species (*Z. marina*) and a tropical species (*H. wrightii*). Where these species co-occur there is a bimodal seasonal abundance, which extends the total annual abundance of seagrasses for a longer period of time (Thayer et al. 1984).

Figure 3.2-4. Illustration of seagrass species in the South Atlantic Region (Source: NMFS, 1997).

Figure 3.2-5. Illustration and table of the distribution of seagrasses in the South Atlantic Region (Source: NMFS, 1998).

Mapping history in North Carolina

The majority of seagrass habitat in North Carolina was mapped by National Oceanic and Atmospheric Administration (NOAA) using photo-interpretation and groundtruthing of aerial photography taken between 1981 and 1992 (Ferguson and Wood 1994). Bogue Sound was originally mapped in 1981 by Carraway and Priddy (1982), but because of differences in scale and methodology, were not comparable to later mapping. Mapping did not include areas south of Bogue Sound. Most of the oligohaline SAV in Albemarle Sound and western Pamlico Sound tributaries were not mapped during this NOAA project. . However, since then, North Carolina Division of Water Quality (DWQ) and NC Division of Marine Fisheries (DMF) has mapped additional SAV habitat in portions of the Neuse and Pamlico rivers and Pamlico Sound tributaries using field survey techniques, and portions of Albemarle Sound have been mapped by state universities. In 2003, Elizabeth City State University remapped Back Bay, Currituck Sound and Kitty Hawk Bay using aerial photography and specifications recommended by NOAA and Virginia Institute of Marine Science (VIMS) (Finkbeiner et al. 2001; Orth et al. 2001). Although mapping of the coast is not entirely complete, the most recent map of known SAV habitat is shown in Figure 3.2-6. The SAV distribution that is depicted in the figure is a mosaic of multiple projects that used imagery ranging from 1981 to 2003, as well as some mapping conducted

Comment [9]: 56727 hectares of SAV for imagery years 2006-2008. From APNEP. Link: http://data.nconemap.gov/geoportal/catalog/search/resource/details.page?uuid=%7B71EF139A-0AAD-483E-8D69-35A79CBAF9EE%7D

Abstract: With funding from the Albemarle-Pamlico National Estuary Partnership (APNEP) and others [NC Division of Marine Fisheries (NC-DMF), National Oceanic and Atmospheric Administration (NOAA), and US Fish and Wildlife Service (US-FWS)], digital data of coastal submerged aquatic vegetation (SAV) was mapped by NC-DMF and Atkins for imagery years 2006-2008 with a planned update by the NC Department of Transportation for imagery years 2012-2013. In addition to its role as critical habitat for many aquatic fauna species, SAV is an important bio-indicator of environmental health because of its sensitivity to aquatic stressors. The ability to detect SAV is critical in understanding ecosystem health and effects of restoration and protection activities. As such, this dataset was developed and intended for research and/or planning. Because SAV distribution, abundance, and density varies seasonally and annually in response to climatic variability, large-scale SAV changes may occur and because of its dynamic nature, this data needs to be continually updated as monitoring continues in the APNEP region.

Comment [10]: Judd Kenworthy: This who entire section will need to be updated with newest numbers, recent surveys, also changes, e.g., the recent large decline in the IRL and the dynamics going on in the Florida Bay System. We should consider discussing what we don't know and how we are dealing with that.

Comment [11]: Judd Kenworthy: Needs significant updating.

Comment [12]: I am not sure I understand why we need to detail the entrie mapping history. I know the historical data is important with respect to understand changes but I think we will need to revised this section with that in mind, not just a long list of who did what and when.

This will be important because we need to address what we consider potential habitat, not just where it exists at the time of a survey or the publication of a map. Seagrasses and SAV are dynamic systems that mover, and come and go inter- and intrannually in association with temperature and local and regional climate (e.g., precipitation and river dischare)

Comment [13]: Judd Kenworthy: It would seem to me that we should do the biology and ecology first then address the mapping and distribution.

completely from field surveys, and includes both seagrasses and oligohaline SAV. Unmapped or inadequately mapped areas should be a high priority for future mapping.

In 2005 a North Carolina SAV Cooperative Habitat Mapping Program was established among 26 state agencies, federal agencies, universities, and non-profit organizations. The purpose of the multi-agency workgroup and 2006 Memorandum of Understanding between organizations is to enhance and accelerate mapping and monitoring efforts by pooling resources and coordinating mapping efforts. The long-term goal of the program is to manage and conserve SAV habitat in North Carolina and southern Virginia in a comprehensive manner through cooperative research, monitoring, restoration, and education (http://www.apnep.org/pages/sav.html). The Albemarle-Pamlico National Estuary Program coordinates the program and is contributing substantial funds for aerial photography so that the entire coast can be mapped in a short time period. However, there is no comprehensive monitoring program yet underway. In 2005, the NC Coastal Habitat Protection Plan (CHPP) was approved by environmental regulatory commissions. The plan summarized the ecological value and status of coastal habitats in North Carolina, including seagrass habitat, and made management recommendations including mapping and monitoring of submerged aquatic vegetation (Street et al. 2005; http://www.ncfisheries.net/habitat/chppdocs/). The CHPP was updated in 2010, and again in 2015. Through the CHPP and APNEP programs, seagrass management, that includes comprehensive monitoring, should improve over the next few years.

Comment [14]: Judd Kenworthy: Check to make sure the correct web link is here.

Comment [15]: url invalid

Comment [16]: url invalid

Comment [17]: The plan is currently in draft, but will be completed by March, 2016.

Figure 3.2-6. Distribution of seagrasses and oligohaline SAV in North Carolina (compiled by Scott Chappell, NC DMF, 2007. Published sources include Carroway and Priddy 1983; Ferguson and Wood 1994. Unpublished data sources from NC DWQ; NC DMF bottom mapping program; Elizabeth City State University; North Carolina State University).

Comment [18]: Amy, I attempted to address your comment re:this, and somehow your comment disappeared :(. So sorry. This compilation should be the mosaic addressed above.

Mapping history in Florida

Seagrass cover estimates for Florida have been based on photo-interpretation of aerial photography, mostly at a scale of 1:24,000. Sargent et al. (1995) made the first coast wide effort to summarize statewide seagrass distribution, using photography from 1982-1990. Madley et al. (2003) constructed new statewide seagrass maps using photography from 1987 to 1999 (Figures 3.2-7-3.2-13). Seagrass habitat is regularly mapped every two to three years in the Southwest, St. Johns River, and South Florida Water Management Districts. Other agencies, such as Florida Department of Environmental Protection (DEP), Florida Fish and Wildlife Conservation Commission (FFWCC), National Oceanic and Atmospheric Administration (NOAA), US Army Corps of Engineers (USACOE), US Geological Service (USGS), and US Mineral Management Service (USMMS) have mapped other local areas on a sporadic basis.

Comment [19]: Judd Kenworthy: Paul Carlson should be able to update this.

Comment [20]: Judd Kenworthy: Context this with the significant changes that have occurred since this document was produced, e.g., IRL, Mosquito Lagoon, Florida Bay.   Cite Fourqureans work monitoing the FKNMS

Differences in habitat classification schemes and accuracy of methods make overall comparisons difficult. However trend analysis has been done with consistent methodology in several smaller regions of Florida. Overall it appears that seagrass losses have occurred in all regions of Florida, with the largest losses occurring near highly developed areas. Along the Atlantic peninsula, comparison of estimates from recent mapping to estimates in the 1940s found little change had occurred to SAV coverage in the northern Indian River Lagoon and Banana River around the federally protected lands of NASA (FFWCC 2003). Extensive losses have occurred in the southern portion of the Indian River lagoon adjacent to highly developed shorelines. Overall, approximately 59% of what is considered potential SAV habitat (based on SAV presence in 1940 maps) in the Indian River Lagoon is vegetated with seagrass. In South Florida, mapping data has indicated significant declines in SAV coverage in highly developed areas such as northern Biscayne Bay. Seagrass habitat in Dade and Monroe counties has the greatest amount of boat-related propeller damage. Florida Bay has also experienced a large decline in seagrass coverage beginning around 1987. The die-off was attributed to reduced water clarity due to multiple factors including algal blooms, sediment sulfide toxicity, hyper-salinity due to drought, and infection by the slime mold *Labyrinthula*. Although the rate of decline has slowed in recent years, losses continue, which has in turn lead to increased turbidity, further reducing water clarity.

Comment [21]: Judd Kenworthy: We need to extend this concept to North Carolina

In Florida there are several ongoing regional seagrass management programs, primarily in subtropical portions of the peninsula (e.g., Indian River Lagoon, Florida Bay, Sarasota Bay, and Tampa Bay). To improve coordination of and increase support for seagrass monitoring and management efforts, the Florida Fish and Wildlife Conservation Commission (2003) recommended that the state develop:

Consensus-based seagrass management strategies at the regional and statewide level;
a methodologically consistent, statewide seagrass mapping and monitoring program;
a schedule for reporting regional and statewide status and trends information;
a schedule for assessing the state's management strategies and the progress made toward achieving the adopted management goals;
a management-oriented, statewide seagrass research program; and
a statewide, public outreach program focused on seagrass management and conservation.

In both North Carolina and Florida, more funding is needed to support comprehensive SAV mapping and management programs. Maps of SAV in Florida can also be viewed on an internet map service at http://ocean.floridamarine.org/mrgis/viewer.htm

Figure 3.2-7. Seagrass distribution along the east coast of Florida, Indian River Lagoon. (Source: P. Carlson, FFWCC 2007).

Figure 3.2-8. Seagrass distribution along Florida's east coast – Melbourne to Ft. Pierce. (Source: P. Carlson, FFWCC 2007).

Figure 3.2-9. Seagrass distribution along Florida's east coast – Ft. Pierce to Delray Beach. (Source: P. Carlson, FFWCC 2007).

Figure 3.2-10. Seagrass distribution along Florida's southeast coast – Hollywood to Key Largo. (Source: P. Carlson, FFWCC 2007).

Figure 3.2-11. Seagrass distribution along the upper Florida Keys – Key Largo to Marathon. (Source: P. Carlson, FFWCC 2007).

Figure 3.2-12. Seagrass distribution along the lower Florida Keys - Marathon to Marquesas. (Source: P. Carlson, FFWCC 2007)

Figure 3.2-13. Seagrass distribution along lower Florida Keys - Key West to the Dry Tortugas. (Source: P. Carlson, FFWCC 2007).

General distribution of seagrass in the south Atlantic

As indicated previously, no seagrasses have been reported to occur in South Carolina and Georgia. Seven of the eight species that occur in the southeastern U.S. are found in Florida. The exception is *Z. marina* whose southern limit is north of Cape Fear, North Carolina (Thayer et al. 1984). In Florida seagrasses are distributed in protected inland waters as well as oceanic environments. In north central (approximately St. Augustine), and southeast Florida most of the seagrasses occur within protected coastal lagoons and in the Intracoastal Waterway (ICW) including; Mosquito Lagoon, Banana River, Indian River Lagoon, Lake Worth, and Biscayne Bay. The most northern distribution of *H. engelmannii* is in the Banana River at Cape Canaveral. The northern limit of *H. decipiens* and *H. johnsonii* is approximately Sebastian Inlet in the Indian River Lagoon. Beginning around the Palm Beach area and continuing south through the Florida Keys, *Halophila decipiens*, while more common inshore in Palm Beach, it is also found on offshore sandy sediments between reefs down to 30m depth. Open water and oceanic meadows of *H. wrightii*, *S. filiforme* and *T. testudinum* begin just south of Virginia Key on the seaward side of Biscayne Bay and continue through the Florida Keys to the Dry Tortugas in water depths up to approximately 30-40 m. (Sargent et al. 1995)

Comment [22]: Judd Kenworthy: I think this general biology and ecology should go first in the text, not the mapping summaries.

Comment [23]: Judd Kenworthy: Also discuss the ESA listed species Halophila johnsonii in more detail.

The majority of seagrass biomass is distributed in the subtidal zone; however, all of the species, with the exception of *H. decipiens*, can be found growing in the intertidal zone. The maximum depth limits are determined by optical water quality and transparency and sometimes limited by water velocities associated with inlets, tidal channels and unstable sediments. In North Carolina maximum depths average between 1.5 and 2.5 m and are similar to the maximum depths of seagrasses in the lagoons and Intracoastal Waterway (ICW) along the east coast of Florida. In locations near inlets with clear water and stable sediments seagrasses grow to 3-5 m, while in nearshore and offshore areas of southeastern Florida and the Keys seagrasses grow to depths of 30m.

Salinity is a very important parameter in estuaries because of its potential to control physico-chemical attributes of the system that affect nutrient cycling, water transparency, floral and faunal composition, and productivity. Salinity also undergoes frequent fluctuations and may act as an important stressor. Given the fact that the south Atlantic region has extensive natural and manmade freshwater sources flowing into coastal systems, salinity is a critical parameter controlling seagrass distribution and abundance (Doering and Chamberlain 1999; Estevez 1999). The spatial distribution of seagrasses in coastal systems is controlled locally by salinity, especially the upper reaches of penetration by different seagrass species (Estevez 1999). Seagrass distribution throughout an estuary can also be affected by long-term modification of freshwater inflow such as has occurred in the St. Lucie River in east central Florida.

Comment [24]: Judd Kenworthy: We need to connect this to overall seagrass and SAV distributions.

Comment [25]: Judd Kenworthy: We are beginning to gain a better understand of salinity and SAV distribution in NC and need to incorporate this into the discussion

Of the eight species of seagrass, *R. maritima*, has the widest tolerance to salinity and can grow and thrive from freshwater to hypersaline conditions (Kantrud 1991). When matched with its fecundity, these two characteristics enable *Ruppia* to occur in a wide range of estuarine conditions as well as having the ability to thrive in fluctuating environments. *Ruppia* is a very important species in marginal and transitional environments which are not as suitable for other seagrasses. *H. wrightii* is considered to be the next most tolerant species for relatively lower salinities, and similar to *Z. marina* (McMillan and Moseley 1967; Thayer et al. 1984). Both of these species are considered euryhaline and regularly reported growing at salinities ranging from very low salinities (5-10 ppt) to full strength seawater. *Thalassia* is considered euryhaline and tolerant of salinities as low as 6-10 ppt for brief periods of time; optimum salinities range from 17-36ppt (Doering and Chamberlain 1999).

Comment [26]: Judd Kenworthy: We also need to point out how ephemeral this and other lower salinity species are.

The salinity tolerances of *Halopila* spp. have not been well studied, however, reports of distribution indicate they are euryhaline and found growing well upstream in estuaries experiencing low salinities and out into the open ocean (Dawes et al. 1989; Toquemada et al. 2005; Kenworthy 2000). The wide range of salinities tolerated by the species of seagrass in the South Atlantic is an important aspect of their function as essential fish habitat. Salinity tolerances enable them to be more widely distributed across the estuarine landscape and are therefore available as habitat to a broader spectrum of fishery species.
As in terrestrial grasslands, seagrass meadows may be seasonal or perennial. The meadows are usually defined by a visible boundary delineating unvegetated and vegetated substrate and vary

in size from small, isolated patches of plants less than a meter in diameter to a continuous distribution of grass tens of square kilometers in area. This natural variation in ~~grass bed~~ patch morphology and patch spatial configuration is related to seagrass dynamics and affects the function of seagrasses as habitat (Fonseca 1996; Murphey and Fonseca 1995; Fonseca and Bell 1998; Fonseca et al. 2002). Seagrass meadows are dynamic spatial and temporal features of the coastal landscape which actually move and can disappear and reappear periodically (den Hartog 1971; Patriquin 1975; Fonseca and Bell 1998; Fonseca et al. 1998; Fonseca et al. 2002). The presence of a seagrasses canopy does not necessarily signify whether or not a location is capable of supporting seagrass habitat. Some species are ephemeral, for example, in North Carolina, shallow *Z. marina* meadows may completely exfoliate in late summer in response to warm temperatures, leaving a signature suggesting there are no seagrasses in the area when, in many instances, the meadows recovers in winter or spring. Because of this, identification of seagrass habitat at certain times of the year can be difficult to determine from visual inspections, which complicates the ability to properly permit water dependent activities such as dredging or marina construction. Environmental characterization of SAV habitat and the better understanding of the processes driving SAV occurrence and temporal changes in distribution are needed to properly identify and protect SAV habitat.

In the South Atlantic region all seagrasses occur on unconsolidated sediments in a wide range of physical settings and different stages of meadow development leading to a variety of cover patterns, ranging from patchy to continuous. Seagrasses patches form and migrate across the sea bottom. In high current environments and areas exposed to wave turbulence, movement is considerable and beds tend to remain in a continuously patchy state. Whereas in low energy embayments and areas protected from large fetch, contiguous perennial beds will tend to form. Seagrass beds developing from seed and mature beds in relatively high energy environments may have similar patchy signatures, but very different physical and chemical characteristics (Kenworthy et al. 1982; Kenworthy 2000).

Depending on the species and the environmental conditions, a meadow may attain full development of density and biomass in a few months (e.g., *Z. marina and Halophila* spp.). Meadows that develop rapidly usually reproduce by seed, forming annual meadows that completely disappear during unfavorable growing conditions. For example, on the east and southeast coasts of Florida between Sebastian Inlet in the Indian River Lagoon (IRL) and North Biscayne Bay, *H. decipiens* forms annual meadows in water generally deeper than 1.5-2.0 m (Dawes et al. 1995; Kenworthy 2000). These depths are where the winter light levels cannot support the larger perennial species such as *R. maritima*, *H. wrightii*, *S. filiforme* and *T. testudinum* (Kenworthy and Fonseca 1996; Kenworthy 2000). In the relatively deeper water the smaller opportunistic *H. decipiens* is capable of germinating seeds in summer months when light levels are adequate. This life history strategy, combined with a thin leaf structure, minimal self

Comment [27]: These spatial configurations of seagrass meadows are primarily driven by hydrodynamics (wave exposure + local tidal currents).

Comment [28]: This sentence contradicts itself. I'm pretty sure that if a seagrass canopy is present, then the area is capable of supporting seagrass.

Comment [29]: Judd Kenworthy: Add references here with the recent work of Jarvis.

Comment [30]: Judd Kenworthy: This is important in defining the scope of "potential habitat" shading, and relatively low non-photosynthetic biomass make the genus *Halophila* ideally suited for growth in fluctuating and highly disturbed environments (Kenworthy et al. 1989; Kenworthy 2000).

Comment [31]: Judd Kenworthy: We need to mention and discuss vegetative fragment dispersal.

These dynamic features of seagrass meadows are not just restricted to the genus *Halophila*. In North Carolina annual meadows of a large bodied species, *Z. marina*, are common in shallow, protected embayments where excessively high (> 30o C) summer water temperatures eliminate *Zostera* beds that thrive in winter and spring when water temperatures are optimal (Thayer et al. 1984). These shallow embayments are replenished annually by seed stocks of *Zostera*, whereas in North Carolina during the summer months when water temperatures exceed 25-30°C, *Zostera* thrives only in relatively deeper water or on tidal flats where water movement is nearly continuous so that the plants are insulated from lethal temperatures and desiccation. In general, whether they are found in the warm temperate coastal waters of North Carolina or the subtropical environment in southeastern Florida, seasonal fluctuations in the abundance of seagrass biomass in the subtidal is normal (Dawes et al. 1995). The range of these seasonal fluctuations tends to increase from south Florida to North Carolina. North Carolina is a special case where seasonal fluctuations may be minimized in water bodies and meadows where *Z. marina* and *H. wrightii* co-occur. These two species are at their southern (*Z. marina*) and northern (*H. wrightii*) range limits, and when one species is limited by seasonal thermal extremes the other species may be abundant.

Comment [32]: Judd Kenworthy: Cite and discuss recent studies by Jarvis et al. in NC

Alternatively, meadows formed by the larger bodied species which have either limited or irregular sexual reproduction may require decades to reach full maturity. For example, the slowest growing species in the south Atlantic region, *T. testudinum*, produces relatively few fruits and seeds at irregular intervals (Tomlinson 1969; Moffler and Durako 1987; Whitfield et al. 2004). When *T. testudinum* is compared to its congeners, *H. wrightii* and *S. filiforme*, it has the slowest rate of vegetative expansion (Fonseca et al., 1987; Kenworthy et al. 2002). Depending on the environmental conditions, rates of vegetative expansion for *H. wrightii* and *S. filiforme* are normally 4 to 10 times faster than *T. testudinum* (Kenworthy et al. 2002). Thus, *T. testudinum* meadows form more slowly than any of the other species, yet if the environmental conditions allow the full development of a *T. testudinum* meadow its biomass and productivity will usually exceed any other seagrass (Zieman 1982).

Regardless of developmental stage or species composition, small seagrass patches and entire meadows can move, the rate of which may also vary on a scale of ~~hours~~ weeks to decades. These dynamic spatial and temporal features of seagrass meadows are important aspects of fishery habitats. Seagrass habitats must be recognized as including not only continuously vegetated perennial beds but also patchy environments with the unvegetated areas between patches as part of the habitat. In fact, available data show that patchy habitats provide many ecological functions similar to continuous meadows (Murphey and Fonseca 1995; Fonseca et al. 1998). Also, it must be recognized that the absence of seagrasses in a particular location does not necessarily mean that the location is not viable seagrass habitat. It could mean that the present conditions are unfavorable for growth, and the duration of this condition could vary from months to years.

Comment [33]: Judd Kenworthy:This is the essence of the concept of "potential habitat".

Ecological Role and Function

The ecological role and function of seagrass habitat has been described by Hemminga and Duarte (2000), Larkum et al. (2006) and Duffy (2006). For more specific information of seagrasses in the South Atlantic region we recommend two U.S. Department of Interior Community Profiles: Thayer et al. (1984) and Zieman (1982). A *Symposium on Biodiversity in the Indian River Lagoon* published in Volume 57 of the Bulletin of Marine Science (Swain et al. 1995) is an excellent compendium of the biology, ecology and biodiversity of seagrass communities on the east coast of Florida. Another important source document is the *Symposium on Subtropical-Tropical Seagrasses of the Southeastern United States* (Durako et al. 1987). Additionally, other published books on the general biology and ecology of seagrasses have information pertaining directly to use of seagrass habitat by managed species and their food sources (McRoy and Helfferich 1977; Phillips and McRoy 1980; Larkum et al. 1989; Bortone 1999; Short and Coles 2001). Additionally, *The relationship of submerged aquatic vegetation (SAV) ecological value to species managed by the Atlantic States Marine Fisheries Commission (ASMFC): summary for the ASMFC SAV Subcommittee* by R. Wilson Laney (1997) provides detailed descriptions and literature citations of seagrass use by species managed by the ASMFC and the South Atlantic Fishery Management Council. Following is a brief summarization of the most important aspects of marine seagrasses which pertain directly to their distribution, abundance and function.

Comment [34]: Judd Kenworthy: Check to see what we can add here.

Seagrasses are rooted plants that can become nearly permanent, long-term features of coastal marine and estuarine ecosystems either as perennial or annual meadows. Because they are rooted, seagrasses directly link the sediments to the water column. No other marine plants are capable of providing this ecological service, to the extent that seagrasses can. Ecological functions provided by seagrass habitat that enhance conditions for fish species include: 1) primary productivity, 2) structural complexity, 3) the provision of substrate for attachement and productivity epiphytes, 4) modified energy regimes and stabilization of sediment and shorelines, and ~~4)~~ 5) nutrient cycling. 7. water quality, and 8) carbon sequestration.

Comment [35]: Judd Kenworthy: We need to also incorporate some discussion about macroalgae and not dismiss their role in the seagrass ecosystem.

On a unit area basis seagrasses are among the most productive ecosystems in the world (McRoy and McMillan 1977; Hemminga and Duarte 2000). High rates of primary production lead to the formation of complex, three dimensional physical structures consisting of a canopy of leaves and a dense matt of roots and rhizomes buried in the sediments. The presence of this physical structure provides substrate for attachment of organisms, shelter from predators, frictional surface area for modification of water flow and wave turbulence, sediment and organic matter deposition, and the physical binding of sediments underneath the canopy. Linked together by nutrient absorbing surfaces on the leaves and roots, and a functional vascular system, seagrass organic matter cycles and stores nutrients, and provides both direct and indirect nutritional benefits to hundreds of species of micro-organisms, meiofauna, carnivores, herbivores and detritivores. The most important aspects of these functions are listed below.

Primary productivity

Seagrass meadows provide four important sources of primary organic matter, 1) their own tissues, 2) dissolved organic matter released from their tissues during metabolism, 3) the epiphytic microscopic and macroscopic plants that attach to the surfaces of the seagrass leaves and live among the canopy, 4) the plants that live on the sediments among the seagrass shoots, (micro- and macroalgae) and 5) the residual organic matter which decomposes in the sediments, on the sediment surface and in the water column. The high rates of primary productivity ensure an abundant supply of organic matter available to be used as an energy source in many different food webs. In some instances a significant portion of the organic matter is exported to adjacent ecosystems (e.g., beach wrack, mangrove forests, open ocean, deep ocean canyons) where it is processed into the food chain. Some fishery organisms consume seagrasses directly (e.g., amphipods and parrot fish), but the majority of the secondary fishery production in the meadows begins with the consumption of epiphyte communities, benthic algae and the utilization of organic detritus. Thus, the food webs supported by seagrass primary production are complex and include many intermediate steps involving microorganisms, meiofauna, small invertebrates such as isopods, and amphipods, as well as the thousands of species of macroinfauna and epifauna in the sediments, on the sediment surface, and in the water column.

Structural complexity

Leaf canopies formed by seagrasses range in size from just a few centimeters (*Halophila* spp.) to more than a meter tall. Where several species co-occur, the three dimensional canopy may take on multiple layers and forms, with long (1.25 m) cylindrical stems and blade surfaces (*S. filiforme*) combined with relatively shorter strap-shaped leaves (*T. testudinum* or *H. wrightii*). No matter what species are present, the existence of leaf surfaces provides structures for attachment of smaller organisms and space between shoots for shelter from predators and adverse environmental conditions. The leaf area in a seagrass meadow may effectively increase the surface area available for colonization by an order of magnitude compared to an unvegetated substrate. While at the same time, the leaves and stems create a large volume of water column sheltered within the canopy and partially obscured by self-shading of the leaves. Within the canopy there is an enormous physico-chemical microenvironment structured and maintained by the seagrasses. This structural influence extends into the sediments where the roots and rhizomes

Comment [36]: Judd Kenworthy: This section needs some good citations.

Comment [37]: Judd Kenworthy: needs some good citations

Comment [38]: Along these lines, the spatial configuration of the meadow also plays a role in faunal utilization and distribution in seagrass. There are a whole host of papers addressing this topic, enough to warrant consideration of a second subsection that speaks directly to this. Some of these papers are relevant to NC and FL. I will add them to the Basecamp site.

stabilize the substrate and form a large pool of organic biomass and a matrix for meiofauna and macrofauna (Kenworthy and Thayer 1984). The additional structure and productivity, in turn, can support a greater diversity and abundance of species. Several studies have shown significantly greater species richness and abundance in SAV beds compared to unvegetated bottom (Thayer et al. 1975; Heck et al. 1989; Ross and Stevens 1992; Irlandi 1994; ASMFC 1997; Wyda et al. 2002).

Comment [39]: Judd Kenworthy: Add other references here.

Modification of energy regimes and sediment stabilization

The leaf surfaces and the collective structure of the canopy provide frictional drag slows water motion and reduces wave turbulence (Zieman 1982). This process promotes the deposition of particles in the meadows, including but not restricted to inorganic sediments, dead organic matter and living organisms. The addition of all of these materials enhances the productivity, stability, and biodiversity of coastal systems with seagrasses. By promoting sediment deposition and stabilization, coastal habitats coupled to seagrasses meadows by water movement receive both direct and indirect benefits.

Comment [40]: Judd Kenworthy: Add fonseca references here.

Nutrient cycling

The high rates of primary production and particle deposition make seagrass meadows important sources and sinks of nutrients. During active periods of growth the constant and high rate of leaf turnover and epiphyte growth provides nutrients directly for herbivores and indirectly to consumers as well as a mechanism for nutrient export and retention. Temporary and permanent retention of nutrients within seagrass meadows is encouraged by particle deposition and burial as well as the formation of organic matter in the sediments by the roots and rhizomes which also sequester carbon.

Comment [41]: Judd Kenworthy: Add recent references

Seagrasses are sensitive to the availability and abundance of nutrients in their surrounding environment and often retain nutrient signatures representing environmental conditions they have experienced, both spatially and temporally (Fourqurean et al. 1992). The variation in tissue nutrient composition is an important factor in fishery utilization of seagrass derived organic matter.

Comment [42]: Judd Kenworthy: Significantly update this paragraph.

Species composition and community structure

Seagrass habitat supports other types of aquatic plants in addition to submerged grasses previously described. Macroalgae (benthic, drift, and floating forms) often co-occur with SAV and provide similar ecological services, but the plant taxa have distinctly different growth forms and contrasting life requirements. Macroalgae grow faster than SAV and do not require unconsolidated substrate for anchoring extensive root systems. Because of this growth pattern, macroalgae do not provide as much sediment stabilization as submerged rooted vascular plants, but do contribute to productivity and biodiversity. Macroalgal genera include salt/brackish (*Ulva*,

Comment [43]: Judd Kenworthy: Good, we have started to capture the function of macroalgae.

Codium, *Gracilaria*, *Enteromorpha*, *Ectocarpus*, and *Cladomorpha* (Thayer et al. 1984; Mallin et al. 2000). In Florida, calcareous benthic algae, such as *Penicillus* and *Halimeda*, grow among seagrasses and contribute a significant source of calcareous sediment to the system.

Comment [44]: Judd Kenworthy: We need to highlight the rates of primary production for epiphytes and their role in the food web up to fisheries.

Epibiota are another important component of SAV habitat. Epibiota are organisms that attach or grow on the surface of a living plant and may or may not derive nutrition from the plant itself. Micro- and macroalgae (i.e., seaweed) can grow on the leaves of SAV. Invertebrates attached to the SAV leaves include protozoans, nematodes, polychaetes, hydroids, bryozoans, sponges, mollusks, barnacles, shrimps and crabs.

Perhaps seagrass meadows are best known for their source of attachment and/or protection for invertebrates such as bay scallops (*Argopectin irradians*) and hard clams (*Mercenaria mercenaria*). Scientific evidence also indicates that blue crabs (*Callinectes sapidus*), pink and brown shrimp (*Farfantepenaeus duorarum*, *F. aztecus*), and lobster (*Panulirus argus*), just to name a few invertebrates, have a strong reliance on seagrass habitats including seagrass-supported trophic intermediaries.

The three dimensional structure provides protective cover for small resident fish and invertebrates and juvenile fish species. Because of this, the nursery role of SAV is critical for many estuarine dependent fishery species in the South Atlantic region such as gag groupers, flounders, red drum, weakfish, striped mullet, pinfish, pigfish, and silversides, just to list a few of the fish taxa documented to utilize seagrass habitats (Thayer et al. 1984; DMF 1990; ASFMC 1997). Sampling in seagrass beds in North Carolina in the 1980s documented over 150 juvenile fish and invertebrate species, of which 40 were commercially important species. In addition, at least 49 adult fish species were reported from beds in eastern Pamlico Sound (DMF 1990). ASMFC compiled a list of ASMFC managed species that utilize SAV for some portion of their life cycle. Over 30 species were documented potentially using SAV as larvae, juveniles, or adults for various functions (Table 3.2-5).

Comment [45]: Over 70 benthic invertebrates have been reported in eelgrass beds along the east coast (Thayer et al. 1984).

While there have been few studies dealing with larval fish settlement and use of seagrass habitats, there have been numerous publications listing juvenile and adult fishes collected in seagrass meadows. The same ecological characteristics of seagrass beds that make the habitat favorable for juveniles should also benefit larval fish and invertebrates. Seagrass beds are important for the brooding of eggs (for example, silverstripe halfbeak, *Hyporhamphus unifasciatus*) and for fishes with demersal eggs (e.g., rough silverside, *Membras martinica*). Larvae of spring-summer spawners such as anchovies (*Anchoa* spp.), gobies, (*Gobiosoma* spp.), pipefish (*Syngnathus fuscus*), weakfish (*Cynoscion regalis*), southern kingfish (*Menticirrhus americanus*), red drum (*Sciaenops ocellatus*), silver perch (*Bairdiella chrysoura*), rough silverside, feather blenny (*Hypsoblennius hentzi*), and halfbeaks are present and use seagrass beds

Table 3.2-5. Ecological functions provided by seagrass habitat for various life stage(s) of ASMFC fishery species (Source: ASFMC 1997). **Life stage documented to use SAV for function listed 1**

SPECIES	**REFUGE/ ATTACHMENT2**	**SPAWNING3**	**FOOD4**	**PREY5**
Atlantic croaker	L,J,A	J?	J,A	
Atlantic menhaden	L,J,A	J,A		
Red drum	L,J	A?	J,A	
Spanish mackerel	J?	J?,A?		
Spot	L,J,A	J,A		
Spotted seatrout	J,A	A	L,J,A	
Striped bass	J?	J?,A?		
American eel	J	J,A?		
Black sea bass	J	J,A?		
Scup	L,J,A?	A?	L?,J,A?	
Tautog	J, E2	E,A	L?,J,A	
American lobster	J?	J?,A?	J?,A?	
Atlantic herring	L?,J?	L?,J?,A?		
Atlantic sturgeon	J?	J?		
Bluefin	J	L?,J,A?		
Northern shrimp	E?,L?2,J?,A?	A?	J?,A?	L?,J?,A?
American shad	J?	J?,A?		
Hickory shad	J?	J?,A?		
Alewife	J?	J?,A?		

<table>
<tr><td colspan="3">Blueback herring</td><td colspan="4">J?</td><td colspan="3">J?,A?</td></tr>
<tr><td colspan="3">Summer flounder</td><td colspan="4">J,A</td><td colspan="3">J,A</td></tr>
<tr><td colspan="2">Weakfish</td><td colspan="3">L,J,A</td><td colspan="3">A?</td><td colspan="2">L,J,A</td></tr>
<tr><td colspan="3">Winter flounder</td><td colspan="4">J?,A?</td><td colspan="3">J?,A?</td></tr>
<tr><td colspan="3">Southern flounder</td><td colspan="4">J,A</td><td colspan="3">J,A?</td></tr>
<tr><td colspan="2">Striped mullet</td><td colspan="3">J,A</td><td colspan="3">J?,A?</td><td colspan="2">L?,J?,A?</td></tr>
<tr><td>White mullet</td><td colspan="3">L,J,A</td><td colspan="2">A?</td><td colspan="3">J?,A?</td><td>L?,J?,A?</td></tr>
<tr><td colspan="3">Rainbow smelt</td><td colspan="4">J,A?</td><td colspan="3">J?,A?</td></tr>
<tr><td>Black drum</td><td colspan="3">L?,J?,A?</td><td colspan="2">A?</td><td colspan="3">J?,A?</td><td>J?,A?</td></tr>
<tr><td>Bay scallop</td><td colspan="3">E?2,L?,J2,A</td><td colspan="2">A?</td><td colspan="3">J?,A?</td><td>J,A</td></tr>
<tr><td colspan="2">Brown shrimp</td><td colspan="3">J,A</td><td colspan="3">J,A</td><td colspan="2">J,A</td></tr>
<tr><td colspan="2">Pink shrimp</td><td colspan="3">J,A</td><td colspan="3">J,A</td><td colspan="2">J,A</td></tr>
<tr><td colspan="2">White shrimp</td><td colspan="3">J?,A?</td><td colspan="3">J,A</td><td colspan="2">J?,A?</td></tr>
<tr><td colspan="2">Blue crab</td><td colspan="3">J,A</td><td colspan="3">J,A</td><td colspan="2">J,A</td></tr>
<tr><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td><td></td></tr>
</table>

Additional Comments
Focus: Link organisms that are in SAV, particularly those under Council Management.
Boost literature review portion.
Boost EFH support and use of habitat by life stage.
Species by life history, where it exists.
FWRI is doing a literature search on species use of habitat by life stage.

Judd Kenworthy:
1. One issue we will need to tackle from the start is making a clear distinction between high (seagrass) and low (SAV) salinity species and their habitats. In its present form this document is really a seagrass document and doesn't adequately address the lower salinity species and their habitats. I am not sure where the agency wants to draw the line on this, but as far as I am concerned the low salinity species habitats in the river mouths and inner estuarine environments are well documented as essential fish habitat and they have very different morphologies, life histories, seasonal cycles and distributions than the higher salinity seagrasses that have a bearing on management. This is especially true for North Carolina, and may be important in some areas of SC and GA where there are no seagrasses, but may have SAV, possibly less important in Florida. Maybe we should be looking at how they dealt with the salinity gradient, species composition and habitat distribution in the Chesapeake EFH consideration as a model to work with.
2. I think the document is structured backwards. I would suggest we do the background, biology and ecology first, follow by the distribution and abundance (mapping) second.
3. We have a lot of updating to do on all subject matters with more recent empirical and survey studies to add support to the essential functions and services of seagrass and SAV.

4. Distributions and abundances need to be updated, but more importantly we should be identifying and discussing the recent dynamics which include some significant losses (e.g., IRL in FL)and gains as well as the unknowns (e.g., the distribution in low salinity NC).
5. We should tie the distribution and mapping data into the concept of "potential habitat", especially the fact that maps are snapshots that do not capture the inter- and intra-annual fluctuations and distributions of SAV and seagrass. With that in mind we need to be recommending that management develop detailed bathymetry maps to incorporate in a "habitat suitability" GIS for SAV and seagrass. We have habitat suitability indices for most of the seagrassesand some of the SAV that can be applied here.
6. If we are going to continue describing the detail of the history of mapping we should put it in a context that is meaningful and not just a narrative list. Otherwise maybe just have a table for the list.
7. I imagine in our next group conference we will be making decisions on how to parcel out responsibilities, correct? I will be prepared to discuss what I think I can best contribute.
8. There is nothing in this document directly addressing climate change and acidificaton so we probably need to address that in our discussions.
9. What about inclusion of examples of how EFH determinations have gone since this document was first prepared. Lessons learned and gaps identified in the process of applying EFH would be helpful for improving this document.

SAFMC Fishery Ecosystem Plan: Coral Reef Habitat Recommendations

Knowledge Gaps:
-Tropicalization: effects of shifting species assemblages Josh
- Mapping in intermediate depths (how much coral is there?) Josh
- Lionfish:

- While there have been multiple studies documenting local-scale effects of invasive lionfish (*Pterois* spp.) predation on native fish species (add citations), none of those studies have occurred in SAFMC-managed waters (a majority of the studies were performed in Bahamian waters), and no studies in any area have assessed the effect of lionfish predation over relatively broad scales. Research is needed to assess the realized effects of lionfish, via predation and potentially competition, on coral reef fish community structure at broader spatial scales (e.g., sub-regional, regional, ecosystem).
- There is considerable interest in controlling, reducing or depleting local lionfish populations through culling efforts (e.g., via spearfishers). Research is needed into (1) the effectiveness of culling efforts, in terms of the frequency and intensity of culling needed to maintain lionfish below targeted densities, (2) what target densities are most appropriate (e.g., near-zero, low, moderate…) in terms of reducing probable ecosystem impacts, and (3) assessing the trade-offs between the costs of culling efforts and the benefits (ecological and fishery-related) derived from those efforts.

- Large dredging projects occur in the area and often the dredge operations occur near coral or the hardbottom areas necessary for coral recruitment. The fill placed on beaches erodes into the water and can be being frequently resuspended increasing long-term turbidity in nearshore coastal systems. (various citations here, e.g., Wanless, H. R., & Maier, K. L. 2007. An evaluation of beach renourishment sands adjacent to reefal settings, southeast Florida. Southeastern Geology 45(1):25-42.)

Understanding the impacts or effects that low level acute and chronic, and well as higher level acute and chronic, turbidity or sedimentation can have on coral colonies and coral reef habitat is lacking. There is a need for better understanding of what chronic lower level turbidity and/or sedimentation can have on coral populations including the impact on existing coral colonies as well as recruitment. This would help to set appropriate standards during coastal construction projects to help ensure the survival of corals and success of their larvae.
- cumulative, especially long term effects from tissue to community level Goldberg papers, others

Potential Management recommendations? (This is a list of candidates that needs discussion by the whole group)

-Water Quality recommendations available in Gregg 2013, http://www.dep.state.fl.us/coastal/programs/coral/reports/LBSP/LBSP_EFH_Lit_Review_and_Synth_Final.pdf

- ❖ Water quality numerical standards (including turbidity related to dredging impacts)

- Coral reef habitats suffer ongoing and repeated damage from dredging and coastal construction projects in the region. Much of this damage should be preventable under existing regulations, but improvements in permitting, implementation, compliance and enforcement are needed. Specific recommendations for such improvement are provided in Lindeman and Ruppert (2011) include

- ❖ Development of a template by permitting agencies with standard language for 'special conditions' to protect coral
- ❖ Development by NMFS of regulatory criteria to identify 'destruction or adverse modification' of ESA Critical Habitat, replacing the current working definition.
- ❖ Develop a common permit tracking system among coastal permitting agencies to enhance compliance and enforcement

-From Uhrin et al 2014: "Focused removal of submerged trap debris from especially vulnerable habitats such as reefs and hardbottom, where trap debris density is high, would mitigate key habitat issues but would not address ghost fishing or the cost of lost gear."

-Diadema restoration? (This is recommended in the Acropora Recovery Plan)

- Monitor need for targeted management of herbivores/parrotfish

- Our Florida Reefs (OFR)
It is recommended that the SAFMC work with the FDEP CRCP and SEFCRI as necessary to understand the outcomes of the Our Florida Reefs community planning process and the recommended management actions developed therein.

> There are several draft recommended management actions currently in development that the SAFMC should consider following the progress of, or be engaged in should those recommendations be included in the final document.
>
> Some draft recommendations include actions in federal waters, while others are within State waters but may be of interest or benefit from input from the SAFMC. The final

product from this process will be a prioritized list of Recommended Management Actions (RMAs). Those RMAs will be distributed to the appropriate bodies for implementation consideration. The lead agency or non-agency partners have been identified in these recommendations, but could be updated to include others according to the scope and scale of the finalized language. Because the final product of OFR is a list of recommendations (not government mandates), moving these proposed strategies forward would involve incorporating ideas into the existing work plan schedule of implementing entities.

While these recommendations are not final, several key recommendations include seasonal protection of spawning aggregations; reducing spearfishing activity on SCUBA; adjusting Spiny Lobster recreational catch limits within State waters; enforcing better protection of key herbivorous species from fishing (e.g. Parrotfish); and finally the recognition of southeast Florida reefs need for a coordinated management plan which includes a network of areas-of-interest for place-based management. This network of place-based management could include coordination with existing managed areas (e.g. see State Parks and Aquatic Preserves), creation of areas for seasonal protection, marine reserves, and areas identified for restoration.

- Lionfish control

CHAPTER: Shallow Coral
Section 4 - Management

Scleractinian corals are currently managed under a zero-take FMP and are protected as Essential Fish Habitat - Habitat Areas of Particular Concern. Seven species in the region are also protected as threatened species under the US Endangered Species Act, with one of those (Dendrogyra cylindrus) previously managed as a Threatened species by the state of Florida. Hence, an ESA Recovery Plan (for Acropora palmata and A.cervicornis) and Florida Species Action Plan (for D. cylindrus) both provide relevant actions for coral conservation and restoration in the region.

OCTOCORAL FISHERY MANAGEMENT

Octocorals are currently managed by the State of Florida under chapter 68B of the Florida Administrative Code (FAC). The State of Florida defines octocorals as “any erect, non-encrusting species of the Subclass Octocorallia, except the species *Gorgonia flabellum* and *G. ventalina*" which are prohibited (FAC 68B-42.002). Up to six octocoral colonies per day may be collected recreationally with a Florida Recreational Saltwater Fishing License (FAC 68B-42.005). There are no limits on the harvest of octocorals for commercial purposes. However, the annual quota for octocorals harvested in State of Florida and adjacent Federal waters is 70,000 colonies (FAC 68-42.006). No power tools may be used to harvest colonies and only one inch of attached substrate around the perimeter of the base of the octocoral holdfast may be removed (FAC 68B-42.006, 68B-42.007, FAC 68B-42.008). Octocorals must be collected and landed live and stored in a re-circulating live-well or oxygenated system aboard the collection vessel (FAC 68B-42.0035).

Areas that are closed to octocoral collection include Atlantic Federal waters north of Cape Canaveral, Biscayne National Park, and in the Stetson-Miami Terrace Deep Water Coral Habitat Area of Particular Concern, as well as the Pourtales Terrace Deep Water Habitat Area of Particular Concern adjacent to Florida state waters (FAC 68B-42.0036). Additional area closures for marine life collection exist in southeastern Florida, including National Parks (Everglades, Biscayne, Dry Tortugas), John Pennekamp Coral Reef State Park, and the Florida Keys National Marine Sanctuary, including the Key Largo Management Area (formerly Key Largo National Marine Sanctuary), the Looe Key Management Area (formerly Looe Key National Marine Sanctuary), and various smaller no-take zones including sanctuary preservation areas, special-use/research-only areas, and ecological reserves (Miller et al. 2014). For further information, Miller et al. (2014) prepared an in-depth description of the U.S South Atlantic Octocoral Fishery.

Section 5-Recommendations (Margaret, Kate, and Mark; Kate to help coordinate with Protected Species chapter w.r.t. ESA Recovery Plan).
Knowledge gaps to be filled?

<u>Knowledge gaps:</u>

- Efficacy and improvement of coral (proactive) restoration strategies (Hunt & Sharp 2014)
- Efficacy and improvement of coral mitigation strategies
- Monitoring & research needs in the Acropora Recovery Plan and Florida Pillar coral Action Plan
- Determine causal factors in coral disease impacts, especially regarding interaction of temperature and local anthropogenic stressors.

<u>Potential management recommendations</u> (candidates for discussion by the whole group)
-Coral population enhancement
- Other ESA Recovery Actions

- OFR recommended management actions
- Florida Pillar Coral Species Action Plan
    - ❖ enhanced mooring balls in sensitive areas
    - ❖ mitigating trap debris
    - ❖ Diadema enhancement/restoration
    - ❖ Enhanced legal enforcement of Florida Coral Reef Protection Act

<u>Recommended Management Actions from A Species Action Plan for the Pillar Coral (FWC)</u>

Habitat Conservation and Management

Action) To reduce anchor damage, install mooring buoys in locations that support sensitive or ecologically critical colonies occurring at popular diving, snorkeling, and fishing locations.
One of the greatest challenges to protecting corals is to protect the overall coral reef habitat. Corals are vulnerable to physical damage from anchors. Buoys should only be installed in locations that are currently frequented by divers, snorkelers, or fishers as installation of buoys in locations not currently frequently used may serve to increase use in that area. Installing mooring buoys is a feasible action provided there is an entity willing to commit to long-term maintenance of the buoys. Mooring buoys are used throughout the FRT to provide convenient access to the reefs by boaters without the need for deploying an anchor. Installation of mooring buoys may effectively protect coral reef habitat, including pillar coral colonies.

Action) Support establishment of long-spined sea urchin nurseries and outplant urchins at pillar coral locations.
The long-spined sea urchin, *Diadema antillarum*, is an herbivore that is important to maintaining conditions favorable for settlement of scleractinian corals, including the pillar coral. As corals need algae-free substrate in order to settle, adding grazers like long-spined urchins to the reefs should increase the amount of habitat available for pillar coral recruitment. Long-spined urchin populations suffered a severe die-off in the 1980s (Lessios 1988) and populations have not yet rebounded. Current research on rearing and outplanting them looks promising, but is still in its infancy. Although long-spined urchin relocation is potentially feasible in the future, it is not an urgent action, as there are many other factors impacting successful settlement and recruitment of pillar coral larvae.

Research led by the FWRI in the Keys is currently underway to gain information on how urchins interact with artificial structures under natural conditions and to test whether these structures enhance their survival rates. These results will serve as a guide for resource managers seeking to develop and implement a comprehensive ecosystem-based coral reef restoration strategy that includes re-establishing a stable long-spined urchin population along the FRT. The FKNMS has identified long-spined sea urchin research as an essential component of restoring the health and resiliency of Florida's reef ecosystem (B. Sharp, FWRI, personal communication). This research will provide information beneficial to future long-spined urchin outplanting efforts. Increasing these grazer populations will increase the potential of settlement success for pillar coral by maintaining suitable settlement habitat.

Action) Implement actions to protect coral from fishing gear.
Some abandoned fishing gear, such as lobster and crab traps, can cause damage to corals if the traps are deployed on or coral or are abandoned. Trap movement during storms poses a great threat to species such as *Dendrogyra cylindrus* due to their fragile columnar growth. Coral colonies of any size are susceptible to fragmentation, breakage, and abrasion from traps and trap lines. Even traps initially placed in locations devoid of corals can be moved by storms into reef habitats and cause damage. Non-tropical storm systems can move traps 30.48 m (100 ft) from their original locations (Lewis et al. 2009). Abrasion can scour tissue away, leaving the colony vulnerable to disease.

<u>Population Management</u>

Action) Map and groundtruth existing populations; research genetic structure and assess genotypic diversity.
In order to effectively manage the population, it is necessary to define the population. Identifying and mapping existing populations will provide information critical to developing effective management strategies. Mapping and groundtruthing of existing colonies of the population are feasible research activities that are of utmost urgency.

Different management and restoration actions will be taken depending on the genetic diversity (or lack thereof) of the existing population. Research into the genetic diversity of select species is ongoing. Researching genetic structure and assessing genotypic diversity is necessary to determine the stability of the current population and to assist in ensuring that multiple genotypes are eventually represented in Florida nurseries. Documentation of sex ratios and gametogenesis will determine sexual reproductive capacities of coral populations. This action may ultimately enable researchers to manipulate and increase genetic diversity and potential larval production of populations.

Action) Acquire rearing techniques, collect gametes and corals of opportunity, rear, and outplant to wild sites

Collecting corals of opportunity for nurseries, developing rearing techniques, and eventual outplanting is the next step after establishing the population baseline. Collecting gametes for genetic and larval rearing studies and collecting corals of opportunity to build a nursery is a feasible activity that is currently being successfully conducted for another species of coral (i.e., staghorn coral [Acropora cervicornis]).

The successful propagation and outplanting of corals, particularly those under threat, can increase populations and restore ecosystems. Improved survival of nursery and outplanted corals maximizes recovery efforts for these Threatened species and the return on nursery-based restoration projects for the time and money spent.

Monitoring and Research

Action) Conduct synoptic and demographic surveys and include disease and health assessments. To assess population status and trends, synoptic and demographic surveys should include assessments of current disease and health status.
General coral reef demographic surveys are currently being conducted by a number of research groups including the FWRI Coral Program, as well as targeted pillar coral demographic surveys (funded by FWC's SWG program). Synoptic surveys have been done along the FRT in the past by the Coral Reef Monitoring and Assessment team, led by Dr. Steven Miller of NSUOC. Since 2005, TNC has implemented the Florida Reef Resilience Program (FRRP) to provide a unified method of dividing up and looking at the reef tract to assess long-term resiliency. The sampling design was created by reviewing existing maps, data, and biophysical information and by conducting workshops and acquiring expert input. The goal of the program is to monitor coral reef health after disturbances. The focus of this program has been on climate change disturbances, such as coral bleaching. Trained experts survey corals and bleaching on the FRT during peak annual temperatures. Follow-up surveys are conducted after moderate and severe bleaching years. The survey methodology is also used to assess other disturbances such as hurricanes, cold-water events, and oil spills. This program provides a general baseline of coral condition across the FRT, provides valuable data regarding species diversity, and will assist in identifying sensitive colonies and their conditions on the FRT (Johnson and Lustic 2011).

Law Enforcement

Action) Collaborate with DEP and Law Enforcement entities to enforce the CRPA.

The CRPA authorizes DEP, as the state's lead trustee for coral reef resources, to protect coral reefs through timely and efficient assessment and recovery of damages to coral reefs. DEP may enter into delegation agreements with other state or local government agencies with coral reefs in their jurisdiction to carry out the intent of the CRPA. The CRPA includes language that would provide for other law enforcement entities, such as the FWC, to support enforcement of the Act by entering into an agreement with the DEP, but at this time no other law enforcement entities have entered into such an agreement. Facilitating agreements between DEP and other law enforcement agencies, including FWC, to allow other enforce the CRPA would strengthen the effectiveness of Act at reducing anchor damage to corals.

Citations from above:

Jordan, L.K.B., K.W. Banks, L.E. Fisher, B.K. Walker, and D.S. Gilliam. 2010. Elevated sedimentation on coral reefs adjacent to a beach nourishment project. Marine Pollution Bulletin 60(2):261-71.

Lindeman, K.C. and T. Ruppert. 2011. Policy recommendations and training to improve agency permitting, compliance, and enforcement for coral resource conservation in southeast Florida. Florida Dept. of Environmental Protection, Southeast Florida Coral Reef Initiative, 207 pp.

Mora. C. (ed.). 2015. Ecology of fishes on coral reefs. Cambridge Univ. Press

SAFMC. 1998a. Final comprehensive amendment addressing essential fish habitat in fishery management plans of the south Atlantic region. Including a final environmental impact statement/supplemental environmental impact statement, initial regulatory flexibility analysis, regulatory impact review, and social impact assessment/fishery impact statement. Charleston, South Carolina. 136 p.

SAFMC. 1998b. Final habitat plan for the South Atlantic region: essential fish habitat requirements for fishery management plans of the South Atlantic Fishery Management Council. Charleston, South Carolina. 639 p.

Wanless, H. and K. Maier. 2007. An evaluation of beach renourishment sands adjacent to reefal settings, Southeast Florida. Southeastern Geology 45(1):25-42.

6.0 Threats to the South Atlantic Ecosystem

6.1 Adverse impacts of non-fishing activities

The waters and substrate that comprise essential fish habitat (EFH) as defined by the Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act), and under jurisdiction of the South Atlantic Fishery Management Council (SAFMC), are diverse, widely distributed, and closely affiliated with other aquatic and terrestrial environments. These characteristics make them readily susceptible to a large number of human activities.

The Essential Fish Habitat (EFH) Interim Final Rule (Federal Register 62 FR 244) defines EFH as ―those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity. The following definitions apply for interpreting the definition of the EFH rule: Waters include aquatic areas and their physical, chemical, and biological properties that are used by fish and invertebrates, and where appropriate may include areas historically used by fish and invertebrates; Substrate includes sediment, hard bottom, structures underlying the waters, and biological communities; Necessary means the habitat required to support a sustainable fishery and a healthy ecosystem; and Spawning, breeding, feeding, or growth to maturity covers species' full life cycle.

Fish habitat is the geographic area where the species occurs at any time during its life. This area can be described by ecological characteristics, location, and time. EFH includes waters and substrate that focus distribution; (e.g., coral reefs, marshes, or submerged aquatic vegetation), and other characteristics that are less distinct such as turbidity zones, water quality, and salinity gradients. Habitat use may change or shift over time due to climatic change, human activities and impacts, and/or other factors such as change with life history stage, species abundance, competition with other species, and environmental variability in time and space. The type of habitat available, its attributes, and its functions are important to species productivity, diversity, health, and survival.

Convention for Threats Identification

The ecological requirements for managed species and biotic communities, including identification of EFH, are addressed in this document. Threats to those habitats are described in terms of those that generally occur landward of the shoreline (Threats to Estuarine Processes) and those that occur oceanward of the shoreline (Threats to Offshore Processes). Threats to Estuarine Processes include but are not limited to agriculture; aquaculture; silviculture; urban/suburban development; commercial and industrial activities; navigation; recreational boating; mining; hydrologic modifications; transportation projects; and natural events and global change. Threats to Offshore Processes include navigation; dumping; offshore sand and mineral mining; oil and gas exploration, development, and transportation; commercial and industrial activities; and natural events and global change. A more comprehensive list of individual activities that may be considered as threats is provided in Section 6.3.17.

Every reasonable effort was made to identify the principal non-fishing and fishing-related threats to EFH and to provide examples and information concerning the relationship between threat-related activities and EFH. Other information sources and examples undoubtedly exist and related studies are underway or are in various stages of publication. Accordingly, the following discussion is a starting point for the identification of threats to EFH. While it meets the strict time limitations imposed by the Magnuson-Stevens Act, regular updating is required to ensure comprehensive and current coverage of the topic addressed.

6.1.1 Freshwater/estuarine/inshore processes

Many species of the South Atlantic region are dependent during at least some life history stages on near-shore waters vulnerable to impacts from land-based sources. Especially vulnerable are species or species groups that require estuaries or freshwater tributaries as primary larval or post-larval habitat. In the southeast, these species include anadromous fish such as striped bass, blueback herring, alewife, American shad, hickory shad, and sturgeons; and brackish species including Atlantic menhaden, summer and southern flounder, red drum, spot, croaker, weakfish, penaeid shrimp, blue crab, and others (Epperly and Ross 1986).

Nearshore EFHs at risk from land-based impacts include submerged shellfish beds; subtidal and intertidal mudflats and shell hash; SAV beds, including eelgrass (*Zostera marina*), Cuban shoal grass (*Halodule wrightii*), and widgeon grass (*Ruppia maritima*); tidal freshwater forested wetlands dominated by tupelo-cypress communities (*Taxodium distichum*, *Nyssa aquatica*), and emergent tidal marshes including both saltmarshes dominated by smooth cordgrass (*Spartina alterniflora*) and brackish marshes dominated by black needlerush (*Juncus roemerianus*). These habitats may be affected both by direct destruction and by degradation of water quality or other factors such as hydrologic modification. Elimination or degradation of wetlands not immediately adjacent to EFH also may diminish the quality and productiveness of downstream estuaries.

The precise relationship between fishery production and habitats is undetermined. Accordingly, the exact degree to which habitat alteration has affected fishery production is also unknown, but is thought to be substantial. Turner and Boesch (1987) assembled and examined evidence of the relationship between the extent of wetland habitats and the yield of fishery species that depend on coastal bays and estuaries. The evidence examined show that fishery stock losses follow wetland losses and fishery stock gains follow wetland gains. While most of the studies were related to shrimp production, other fisheries are likely follow this trend. In the southeastern U.S., the dominant sources of land-based impacts include major land-disturbing activities such as agriculture, silviculture, and residential and commercial development. The following discussions characterize major threats in the coastal zone of the southeast, summarize ways that EFH is impacted, and characterize the current extent of such impacts. Impacts can occur at three scales: immediate watersheds of EFH; broader watersheds of important estuarine nurseries; and distant or indirect impacts mediated through more widespread movement of water and its chemical and physical make-up.

6.1.1.1 Agriculture

Agriculture in the southeast has undergone dramatic changes over time. Most operations were at one time individual and small-scale enterprises, but in recent years have transformed into highly integrated, large-scale industries. Besides the extensive conversion of wetlands to crop and animal production, the most dramatic change in southern agriculture is the large scale expansion in animal production that has occurred during the last decade. The most dramatic increases have occurred in corporate hog operations in North Carolina. According to North Carolina Agricultural Statistics, the 1996/1997 hog numbers (8,969,200) for the 44 coastal counties are more than quadruple the 1986 numbers (2,117,800) for the same area. At the same time, the number of hog farms has declined precipitously, by a factor of three.

Other southeastern states have not yet experienced the same increase in swine herds. South Carolina's coastal counties, in fact, experienced a net reduction in swine herds from 374,000 head in 1986 to 194,900 head in 1996 (South Carolina Agricultural Statistics). Georgia had a similar decrease in the coastal plain counties, decreasing from 400,911 head in 1987 to 317,795 head in 1992 (Georgia Agricultural Statistics). Florida numbers experienced a decline in Atlantic watersheds from about 23,541 head in 1987 to 12,482 head in 1992 (Florida Agricultural Statistics). Part of the reason for the differences in hog production among the states is the development of industrial hog-growing technologies in North Carolina, plus differences in state regulatory programs. South Carolina, for instance, recently adopted very stringent and restrictive new laws governing hog-growing operations.

Poultry production, a second major agricultural animal product, has also increased substantially in the southeast. Again, North Carolina leads the nation in several poultry categories. In 1996, 313,735,000 birds were produced in coastal North Carolina; up from 45,588,966 birds in 1986. South Carolina coastal counties also showed a significant increase in production over this decade: 57,834,000 birds were produced in 1986 and 140,038,000 in 1996. The increases in the Georgia and Florida Atlantic coastal counties were much more moderate from 1987 to 1992, with production rates of 12,907,265 to 15,438,031 birds, and 2,780,706 to 2,886,335 birds, respectively (all data from state agricultural statistics).

Patterns in cropland use also have been in flux. In the North Carolina coastal plain, harvested cropland has remained almost static during the past decade, at about three million acres. However, fertilizer use has increased from 848,927 tons in 1986 to 2,006,251 tons in 1996 (not including swine and other animal waste land application). During the same period, South Carolina has experienced a net decrease in harvested acreage in the coastal plain, from 1,759,162 acres to 1,589,420 acres, but a net increase in fertilizer usage of about 38% to 331,597 tons. Harvested cropland along the Georgia coast is up slightly, to about 900,000 acres in 1992. Comparable data on fertilizer usage are not yet available. Harvested cropland in the Florida Atlantic coastal plain is down from about 1.1 million acres in 1992 to 675,081 acres in 1996. (All data from state agricultural statistics).

The overall pattern in crop production is one of great intensification of use on a fairly stable land base. Large increases in fertilizer usage and manure-based nitrogen fluxes (from surface and groundwater and from airborne sources) have occurred during the last decade in at least some southeastern states, including watersheds that were already artificially enriched.

Nutrient pollution can result in cascading ecological and economic impacts, including fish kills due to oxygen depletion, seagrass die-offs, excessive and sometimes toxic algal blooms, changes in marine biodiversity, increases in human illnesses, and loss of tourism (NRC 2000). For example, in southeast Florida nutrient inputs to Lake Okeechobee from central Florida agriculture activities (primarily sugar) are then discharged to important estuaries including the St. Lucie estuary. Timed releases associated with flood control activities result in large quantities of nutrient-laden water inputs to the St. Lucie estuary. Between 2004 and 2005, it is estimated that approximately 320 billion gallons of this water was diverted to the St. Lucie estuary. Many researchers have suspected that algal blooms and resulting fish kills in 2005-2006 were a result of this activity.

Potential Threats to EFH from Agriculture

Potential threats include: conversion of wetlands to agricultural lands, or for farm related purposes such as roads and irrigation ponds; direct and non-point source discharge of fill, nutrients, chemicals, and surface and ground waters into streams, rivers, and estuaries; hydrologic modification of ditches, dikes, farm ponds and other similar structures and water control devices; damage to wetlands and submerged bottoms by livestock grazing and/or movement; and cumulative and synergistic effects caused by association of these and other related activities.

Certain agricultural activities present a threat to EFH in the southeast. The major components of this threat include wetland conversion, nutrient over enrichment with subsequent deoxygenation of surface waters, shading by excessive algae and plant growth, and stimulation of toxic dinoflagellates; sedimentation; and delivery of toxicants into sensitive waters. Agriculture (including silviculture) accounted for 87% of all wetland losses observed nationally between the mid 1950's and mid 1970's (Tiner 1984). This loss has been estimated at more than 458,000 acres per year between the mid 1950's and mid 1970's in the coterminous U.S. (Tiner 1984). The most extensive losses observed in the southeast were in Florida and North Carolina where agricultural drainage continues to destroy large tracts of wetlands (Tiner 1984). Current agriculture conversion statistics for the southeast show that:

During the mid-1970s to the mid-1980s ―Florida showed a net wetland loss of 260,000 acres, mainly from the destruction of palustrine wetlands. Two-thirds of the loss of palustrine wetlands was attributable to agricultural development...‖ (Hefner et al. 1994).

―Between the mid-1970s and mid 1980s, more than 100,000 acres of freshwater forested wetlands in Georgia were destroyed, mostly because of conversion to land uses such as agriculture‖ (Dahl et al. 1991).

Between 1982 and 1989, South Carolina lost 155,500 acres, of this amount agriculture was responsible for 28% (Dahl 1997).

In North Carolina about one-third of the wetland alteration in the coastal plain has occurred since the 1950s. Of this amount, agriculture was responsible for about 42% (Cashin et al. 1992).

Excessively enriched waters often do not support desirable species or populations of fish and invertebrates. They also may not support food chain and other ecological assemblages needed to sustain desirable species and populations. When overly abundant, nutrients such as nitrogen (ammonia) and phosphorus may degrade or eliminate EFH and its flora and fauna through several processes. Most problematic of these is the process whereby dissolved oxygen in the water is reduced by decaying plant life that prospered under nutrient rich conditions. In severe oxygen depletion situations fish and invertebrates may suffocate from oxygen deprivation.

Nutrient enrichment may also lead to direct toxicity when toxic organism populations ―bloom‖ or become excessively large -- situations that are becoming more prevalent and which are discussed in detail in subsequent sections. Although affected by acidity, water temperature, and other factors, total ammonia concentrations in excess of about 2 mg/L normally exceed the chronic exposure level for fish (Mueller and Helsel 1996). In alkaline water at high temperature, the criteria may be exceeded by total ammonia concentrations of less than 0.1 mg/L. The natural conversion of ammonia to nitrate in streams removes oxygen from water and, therefore, may also harm fish (Mueller and Helsel 1996). While less problematic in estuarine and marine environments, phosphorus is a major factor in nutrient enrichment and eutrophication of freshwater systems. There are no minimum discharge standards for phosphorus; however, the U.S. EPA recommends that phosphates should not exceed 0.05 mg/L when discharged into streams entering lakes and reservoirs (Muller and Helsel 1996). Since freshwater systems may be used directly by anadromous fish, and they may also discharge into coastal waters, the quality of these waters has considerable bearing on many commercially and recreationally important aquatic resources and their habitats, including EFH. The nutrient inputs from central Florida agriculture (i.e., sugar) to Lake Okeechobee, the St. Lucie estuary, and Indian River Lagoon are suspected to have caused algal blooms, seagrass die-offs, and notable bivalve and fish kills in 2005-2006. In addition, the nutrient inputs are also suspected to have adversely impacted reefs located just outside the St. Lucie Inlet (e.g., Peck‘s reef).

In extreme situations living resources may be temporally or permanently displaced due to shifts in the aquatic food web, or by the physical presence of certain plant life. Excessive plant growth may also impede requisite functions (e.g., photosynthesis) of desirable plant life, hence EFH, as in the case of SAV where leaves may become covered with dense growths of algae, diatoms, and other biota such as bacteria and fungi.

Agriculture is believed to be the single largest contributor of nutrients into southeastern watersheds. The largest human additions of nitrogen result from an increased use of inorganic fertilizers (NRC 2000). In the Tar-Pamlico Estuary Basin in North Carolina, agriculture is responsible for approximately 45% of total nitrogen loading to the estuary, and 55% of phosphorus loading (NCDEHNR 1997a). An additional 33% of nitrogen and 17% of phosphorus comes from atmospheric sources that include, but is not limited to agriculture (NCDEHNR 1994, 1997a). In the adjacent Neuse River Basin, 54% of nitrogen is estimated to arise from agricultural sources (NCDEHNR 1993, 1997b). These two tributaries discharge into Pamlico Sound, the nation's second largest estuary, and the largest in the southeast.

Animal production is a threat to southeastern estuarine nutrient balances. The current usual management practice for manure from swine and other confined domestic mammals is storage and treatment in anaerobic lagoons followed by land application. This process relies on volatilization of nitrogen to account for roughly 80% of the total produced nitrogen, with concomitant downwind delivery in a zone of influence of roughly 100 kilometers (Rudek 1997). Airborne deposition of nitrogen into coastal waters in the region has been verified from field data to be a major source of enrichment in a number of southeastern estuaries. The most complete work at this time is focused on the Neuse River Estuary in North Carolina, where primary production was boosted two to three times by atmospheric deposition at ambient levels (Paerl et al. 1995a, 1995b). Actual plant uptake by crops on land-application fields accounts for no more than 10% of nitrogen use. Surplus nitrogen is delivered to shallow groundwater systems which, in turn, feed warm-season surface flows into adjacent streams and rivers. Thus, the vast majority of this material is redeposited on land and in surface waters.
Studies by Barker (1997) and Barker and Zublena (1995) also show that many North Carolina coastal counties are receiving swine-based nitrogen and/or phosphorus at levels in excess of total

crop-plant growth needs. This analysis actually underestimates the problem, because it considers only direct land-applied nutrients and ignores swine-based atmospheric deposition in these counties. A report compiled for Senator Tom Harkin (D-IA) analyzed manure production patterns nationally by county and found zones of very high production in coastal North Carolina and in individual counties in the other three southeastern states. That document also reports excessive production above crop growth needs in many areas (Minority Staff 1997). A recent estimate of agricultural emissions of ammonia from the North Carolina coastal plain is about 200.3 million lbs of nitrogen from animal waste, and 15 million lbs of nitrogen from fertilizers. Hogs alone contribute about 135 million lbs of nitrogen emissions in coastal North Carolina; larger than the entire National Atmospheric Deposition Program estimate of airborne deposition from all sources in the North Carolina coastal plain (Rudek 1997).
In response to nutrient enrichment problems and public concern, the North Carolina General Assembly has moved to impose a two-year moratorium on the development of new or enhanced hog farms, pending the replacement of current anaerobic lagoon technology with a more acceptable alternative.

High nutrient loadings also have been documented in other southeastern river basins and estuaries. Among seven river basins in Florida and Georgia examined recently by the U.S. Geological Survey, two in Georgia (the Altamaha and the Satilla) were found to be very high in nitrogen inputs at 5,470 (kg/yr)/km2 and 5,430 (kg/yr)/km2, respectively. Animal waste was the dominant source of nitrogen loading in both basins. Fertilizer was the biggest source in the St. Johns River Basin in Florida, and the Ogeechee Basin in Georgia. The most dominant sources of nutrient loading are non-point-source in origin, and predominantly agricultural (USGS 1997).

The National Water Quality Assessment Program is also examining the Santee Basin and nearby coastal drainages in South Carolina. Data from 1994 covering 24,868 square miles in South and North Carolina are being considered for this analysis. Although definitive information is not yet available, nutrient pollution of lakes and the rivers themselves has been identified as a major water quality issue for the program (USGS 1994). The first reports from this program are now available and include an annotated bibliography of water quality databases and recent publications on the water quality of the region (Abrahamsen et al. 1997).

Impacts of sediment from non-point-sources including agriculture and silviculture remain at the top of the water pollution list nationally (USEPA 1990) and in the southeastern states (NCDEHNR 1996b). While sediment-based impacts are typically considered to be most acute in freshwater systems, sediment pollution can also threaten EFH. Because sedimentation is a natural process in most aquatic systems it is generally not problematic except where deposition rates vastly exceed ambient conditions. In these situations, benthic animals and plants and demersal fishes that are unable to adjust or relocate may be buried or undergo disruption in growth and reproduction. Lethal and non-lethal effects of turbidity include ingestion of non-food particles by shellfish and polychaete worms, clogging of pores and gills, erosion of gills and other apparatuses such as fins, tentacles, and cilia that may be used for locomotion and feeding, burial of eggs and juveniles, and burial of substrates that may be needed for cover, attachment, and reproduction. In areas that support SAV, primary production levels may be reduced where light penetration is limited by increased turbidity. While generally less important as a potential threat to EFH in the South Atlantic region, sediment deprivation may be locally troublesome since subsidence and erosion of wetlands and other habitats may result. Impounded coastal wetlands used for rice culture and other agricultural crop production in North Carolina, South Carolina, and Georgia are notable since large areas have been permanently altered even though tidal flow has been restored in many cases. In the Altamaha River Estuary in Georgia vast areas of freshwater and brackish tidal forested wetlands have been converted to emergent wetlands following construction of dikes and ditches that interrupted both deposition of alluvial materials and other processes.

Sediment pollution from agriculture is widespread in the coastal zone of the southeastern states. For example, North Carolina's "303d list," the listing of degraded water bodies required to be compiled by the Clean Water Act, contains an array of coastal streams degraded at least in part by agricultural sediment pollution. These include tributaries of the northeast Cape Fear River and Black River; Potecasi Creek (Chowan River); Trent River (Neuse Basin); Little River (Pasquotank Basin); Tranter‘s, Grindle, Conetoe and Town creeks (Tar-Pamlico Basin); and Newport River (NCDEHNR 1996a).

Pathogens from agricultural sources also threaten EFH, especially shellfish waters. The biggest single threat is probably poorly managed animal waste. A secondary source is land-disturbing activity related to putting new land into agricultural production. This may result in additional delivery of fecal coliform bacteria in quantities of potential concern.

The most dramatic cases of contamination of EFH from agricultural sources include spills of animal waste into coastal watersheds. North Carolina has suffered a number of recent spills, including many in the summer of 1995. A large swine lagoon rupture in 1995 spilled about 25 million gallons of waste into the New River Estuary causing severe anoxia, stimulating toxic algal blooms, and elevating fecal bacteria concentrations in both the receiving waters and sediments. Effects of this event persisted for over 61 days (Burkholder et al. 1997). Similar, but smaller, events were documented into tributaries of the Cape Fear River Estuary, North Carolina, from both swine and poultry sources. Impacts included large nutrient delivery, algal blooms, and contamination with huge loads of fecal bacteria; including pathogenic *Clostridium perfringens* (Mallin et al. 1997). This study documented 30 animal waste spills in North Carolina in 1995 and 1996.

Bacteria from other agricultural sources also may contribute to contamination of shellfish waters. As wetland landscapes are developed for agriculture, offsite water delivery is enhanced (Skaggs et al. 1980). Many scientists believe that this hydrologic effect may contribute to elevated fecal coliform counts in receiving waters. This is suggested by preliminary studies in Otter Creek, Broad Creek, and the South River, North Carolina (J. Sauber, personal communication).
The variation in the scope and composition of agricultural non-point-source discharges and in the receiving waters creates an almost endless range of possible effects on aquatic resources, including EFH. Exposure of estuarine finfish and shellfish to toxic levels of insecticides, herbicides, and fungicides may occur, resulting in significant declines in populations (Scott 1997). Sublethal effects also are evident. For example, many compounds released by agricultural operations may adversely affect hormones such as estrogen and androgen that are linked to immune suppression (Scott 1997). These compounds usually do not kill the animal immediately, but reduce its life span and often its ability to reproduce.

Agricultural compounds that have been identified as having properties damaging to aquatic organisms include the commonly used herbicides aldicarb and atrazine and others such as, endosulfan, chlorpyrifos, and trace metals such as copper and mercury.

The enormous variation in the scope and composition of agricultural nonpoint source discharges and in the environmental nature of the receiving waters creates an almost endless range of possible effects on aquatic resources, including EFH. As noted in Scott (1997):

"Agricultural nonpoint source (NPS) runoff may result in significant discharges of pesticides, suspended sediments and fertilizers into estuarine habitats adjacent to agricultural areas or downstream from agricultural watersheds. Exposure of estuarine finfish and shellfish to toxic levels of insecticides, herbicides, and fungicides may occur, resulting in significant declines in field populations. Development of new management techniques such as Integrated Pest Management (IPMs), Best Management Practices (BMPs), and Retention Ponds (RP) are risk management tools which have been used to reduce contaminant risk from agricultural NPS runoff."

In association with Scott's (1997) observations, the National Ocean Service (NOS), Charleston Laboratory examined effects of NPS agricultural runoff on living marine resources in an attempt to define impacts on fishery resources and to develop risk reduction strategies to minimize/mitigate impacts. Investigations involving coastal estuarine ecosystems in South Carolina examined several sites used for vegetable farming (e.g., tomatoes, cucumbers, snap beans), where varied levels of risk reduction strategies were employed. The studies used grass shrimp (*Palaemonetes pugio*) and the mummichog (*Fundulus heteroclitus*) as well as other macropelagic populations. These two species represent more than 85% of the total macrofaunal (greater than 15mm) densities in small tidal creek nursery grounds in South Carolina and they are important due to their role in estuarine food webs. The studies demonstrated that pesticide exposure caused fish and invertebrate abundance reductions and mortality. Comparison of field results with laboratory toxicity tests clearly established that implementation of an integrated risk reduction strategy can significantly reduce NPS agricultural pesticide runoff. At intensively managed (IPM, BMPs, and RP) agricultural sites where strict NPS control techniques were administered, instream pesticide (azinphosmethyl, endosulfan, and fenvalerate) levels were reduced by 89-90% (Preceding from Scott 1997).

According to Scott (1997) the commonly used herbicides aldicarb and atrazine are potential endocrine disrupting chemicals (e.g., compounds that adversely affect hormones such as estrogen and androgen) and are linked to immune suppression. A 1992, Texas investigation found atrazine at concentrations greater than 60 ug/L in 98% of surface water samples that were taken on an annual basis. Laboratory toxicity tests of atrazine effects on estuarine phytoplankton revealed that chronic, low level atrazine exposure over multiple generations lead to enhanced sensitivity of phytoplankton and combined alachlor and atrazine exposure caused greater than simple additive toxicity in phytoplankton (Scott 1997).

The chronic effects of agriculture derived non-point source discharge have been extensively studied in Florida where impacts are occurring on a large scale basis. Essentially all of Florida

Bay has undergone significant and undesirable biological, chemical, and physical change due to large scale agricultural practices, including hydrologic modification, in the Everglades. While these changes are occurring primarily in waters that lie outside of SAFMC jurisdiction, they are notable because of their size, magnitude, and complexity. Two basic lessons from the Everglades/Florida Bay situation also have application in watersheds found along the South Atlantic. They are: (1) the chronic environmental and ecological effects of regional agricultural practices may be extremely large and devastating and (2) the financial costs associated with analyzing and remedying these effects are likely to be enormous and possibly ineffective.

The factors associated with EFH degradation by agricultural related hypoxia are only poorly understood, but are of concern. Thus far, the extensive hypoxic zones and conditions observed in the Gulf of Mexico have not occurred in the South Atlantic region. Exceptions include relatively small, yet harmful, localized events in portions of North Carolina and South Carolina. In this region, North Carolina's estuarine waters are particularly vulnerable due to their shallow depths, poor flushing characteristics, and the abundance of hog farms found in the coastal zone. Although the most conspicuous effect of hypoxia is the mortality of larger fish and possibly invertebrates, even greater harm may be occurring with sensitive larval and juvenile forms since they are most vulnerable to oxygen depletion and other forms of environmental perturbation.

6.1.1.2 Aquaculture

Potential Threats to EFH from Aquaculture

Potential threats include: dredging and filling of wetlands and other coastal habitats and other modification of wetlands, submerged bottoms, and waters through introduction of pens, nets, and other containment and production devices; introduction of waste products and toxic chemicals; and introduction of exotic organisms; in addition to competition with wild stock for food sources.

Nationwide aquaculture is a vibrant industry with the annual value of product sold exceeding $866 million in 2005, although revenues have declined somewhat over the past 10 years (U.S. Department of Agriculture 2006). Within the Atlantic southeastern U.S., the annual value of product sold amounted to over $94 million in 2005, with Florida ($57.4 million) and North Carolina ($24.7 million) leading Georgia ($4.5 million) and South Carolina ($4.7 million). All aquaculture facilities in these states are located either on uplands or in coastal waters and no offshore aquaculture farms presently exist in the Atlantic southeastern U.S. The primary aquaculture operations in the Atlantic southeastern U.S. are shellfish farms (including hatcheries for production of seed stock), production of marine species in closed-recirculation systems, and production for enhancement of native fishery stocks.

The growing demand for seafood reflects both the growth of the U.S. population and the increased awareness of health benefits that result from a diet that includes seafood (Nesheim and Yaktine 2007). Currently, more than 80% of the U.S. seafood supply is imported, with over 40% of that amount coming from foreign aquaculture operations. Considering the substantial economic incentive to increase aquaculture production in the U.S. and the gradual elimination of technological barriers, expansion of the domestic aquaculture industry is expected over the next decade. Offshore areas may receive particular attention for development (Stickney et al. 2006).

Aquaculture and Fishery Habitats

Aquaculture has long been a source of human food. Within the last century, the technology of aquaculture has changed dramatically allowing application of semi-intensive and intensive farming systems. While this concentrates aquaculture activities to relatively small spatial areas and sets the stage for potential environmental conflicts, these concerns can be mitigated through appropriate management measures (Marine Aquaculture Task Force 2007). Balancing the demand for seafood and economic growth with the need to maintain coastal and marine ecosystems is a challenge that aquaculture accepts.

Nash et al. (2005) used the framework of an ecological risk assessment to examine common perceptions about the impacts of aquaculture on coastal and offshore habitats. The framework for this assessment was developed by the United Nations World Health Organization, has undergone extensive peer review, and is widely applied nationally and internationally. Ten types of potential impacts from aquaculture are noted: (1) increased organic loading from fecal material, uneaten food, and the decomposition of dead fish; (2) increased inorganic loading from fecal material and uneaten food; (3) residual heavy metals from uneaten food (primarily zinc) and from antifouling treatments (primarily copper); (4) transmission of disease to wild populations; (5) transmission of residual therapeutants to wild populations; (6) biological interactions from non-native species or genetically modified organisms with native populations from escapees, eggs, and gametes; (7) physical interactions with native populations through entanglement with nets, moorings, and other structures; (8) physical impacts on habitat from dredging, filling, nets, moorings, or other structures needed to establish a facility; (9) reductions in native populations from use of wild-caught juveniles for grow out; and (10) harvesting of industrial fisheries for use as fish feed. The assessment concludes that the level of risk from these sources is none to low when proper management measures are in place, including siting facilities to avoid areas with low water circulation or high boat traffic, judiciously managing stocking densities and managing waste, carefully selecting grow-out stock, and adhering to best management practices to control fouling, escapes, predation, diseases, and so forth. Use of geographic information systems (GIS) has led to spatial models that aid the examination of alternative sites for aquaculture operations (for an example from the southeastern U.S., see Arnold et al. 2000). NOAA is building a broad based aquaculture program to enable expansion of all suitable forms of marine aquaculture within the context of complementing seafood production from wild catch, safeguarding environmental resources, and balancing multiple uses. An important objective of this program is to establish a comprehensive regulatory program for marine aquaculture operations. This program will complement existing regulatory programs that already apply to aquaculture operations, such as regulation the U.S. Army Corps of Engineers and U.S. Coast Guard of the placement of structures within navigable waters, regulation of water quality by the U.S. Environmental Protection Agency and individual states, regulation of therapeutants by the Food and Drug Administration, and oversight of interactions with fisheries and endangered species by NOAA‘s National Marine Fisheries Service.

6.1.1.3 Silviculture

Forested wetlands are the most abundant wetland type along the eastern seaboard. They include such diverse types as black spruce bogs, cedar swamps, red maple swamps, and bottomland hardwood forests (Tiner 1984). Scrub/shrub and forested wetlands account for over 59.4 million acres within coastal counties from North Carolina to Florida (Field et al. 1991). These wetlands also have been the most affected by forestry practices and, to a lesser degree, development. At a national level, from the mid 1950's to the mid 1970's, about 440,000 acres/year of palustrine wetlands (including forested wetlands) were lost (Tiner 1984). About 87% of this loss is accounted for by agricultural development; including silviculture (Tiner 1984). Trends in the southeast follow the national trend with North Carolina and Florida registering the most extensive wetland losses (Tiner 1984).

Potential Threats to EFH from Silviculture

Potential threats include: conversion of wetlands to silviculture production sites or for tree removal and other silviculture related purposes such as roads and irrigation ponds; direct and/or non-point-source discharge of fill, nutrients, chemicals, and surface and ground waters into streams, rivers and estuaries; hydrological modification to include ditches, dikes, irrigation ponds and other similar structures and water control devices; damage to wetlands and submerged bottoms by timber harvest activities; connected actions such as the construction of roads, and cumulative and synergistic effects caused by association of these and other silviculture and non-silviculture related activities.

The southeastern United States produces more industrial timber than any other region of the world. This timber production is from a forest base that includes almost one-half of the world's industrial forest plantations (Lee et al. 2005). Silviculture presents a significant threat to EFH largely due to the concentration of this activity in landscape positions near certain EFH, especially anadromous fish spawning and nursery areas and brackish primary and secondary nursery areas. Although silviculture typically is a less intensive land use activity than agriculture or urban development (Hughes 1996), the periodic intense disturbances associated with harvest, the installation and maintenance of dense drainage systems in wetlands and former wetlands, changes in vegetation, and the use of nutrient supplements and toxicants can significantly and adversely affect surface waters, EFH, and their associated biota.

The most important fundamental change with installation of intensive silviculture pertains to the water management system. Dense drainage systems allow the removal of significant amounts of water from hydric soil sites, intercept rain, and dewater stored groundwater. The effect on the wetlands can be serious if water tables are lowered such that hydric soils lose their water content. Organic constituents of hydric soils can then be oxidized, causing soil subsidence and liberation of previously bonded metals and nutrients. Clearing vegetation from wetland soils may also divert surface water into runoff pathways to the extent that both annual average runoff and event-related peak flows are exacerbated (Daniel 1981; McCarthy and Skaggs 1992). This runoff is a threat because it can change salinity regimes in receiving brackish water systems and it carries excess nutrients and other potential pollutants into sensitive waters and EFH (Pate and Jones 1980).

Conversion of mixed forested wetland and depressional cypress dome areas to silviculture is known to significantly reduce the water table. Studies have shown that slash pine (*Pinus elliottii*) through evapotransport can reduce the water table in an area by up to 36-inches depending on tree maturation. This reduction in subsurface water is higher than wetland canopy species that might have been originally found in a converted wetland area and contribute to soil subsidence and oxidation (value loss). Further this change in land-use (conversion of a wetland to silviculture) and the accompanying hydrological alterations change how these areas are regulated. In Florida, some silviculture areas are not regulated by state or federal agencies as wetlands even though many of the wetland characteristics are still evident (hydric soils, wetland vegetation, and hydroperiod). As a result conversion of these areas to commercial and residential development is expedited and compensatory mitigation for wetland function loss (albeit impaired or reduced) is not sought (Kruczynsky, personal communication).

The sensitivity of EFH to water balance perturbations is variable and poorly understood. Although some important species are highly sensitive to excessive salinity changes at young age classes (e.g., brown shrimp; Hunt et al. 1980), relatively little is known about the overall implications of flow modification from drained silvicultural areas. Limited studies on pumped drainage water in North Carolina showed minor impact to juvenile and adult spot and Atlantic croaker in response to pumping (Broad Creek Study Report). Effects on spring post-larval settlement periods for brown shrimp remain speculative since the effects of rainfall during pumping have not been determined.

In the Altamaha drainage in Georgia, water balance disturbance is thought to be a key factor in declining catch per unit effort of blue crab and shrimp (J. Holland, personal communication) and an in-depth hydrological investigation of that area has been proposed. Livingston et al. (1997) showed that reductions in freshwater inflow to the Apalachicola River Estuary in Florida led to initial turbidity reductions and increased primary productivity. Over time productivity reductions and major food web shifts were observed, probably in response to decreased nutrient delivery. As reported by Livingston et al. (1997) food web shifts remained minor so long as river flow did not greatly exceed natural limits. There is a concern that southeastern watersheds would respond in a similar manner.
Silviculture also has the potential to significantly affect nutrient delivery patterns into EFH, both through soil amendments with nitrogen and phosphorus and through changes in nutrient processing and delivery systems. Modification of these delivery patterns can be a threat to EFH. Typical forestry operations in the southeast add limited nitrogen and phosphorus during the growing cycle (Amatya et al. 1996). In addition, typical wetland soils are effective at removing incident nitrogen through nitrification and denitrification pathways. Wetlands are important sinks for atmospherically derived nitrogen. As such, riparian and isolated wetlands may buffer EFH from vehicle and animal waste-derived nitrogen enrichment. Drainage networks effectively short-circuit this buffering capacity by reducing retention periods and denitrification opportunities (Whigham et al. 1988; EDF and WWF 1992).

The huge areas involved and their proximity to sensitive estuaries makes forestry a major player in nutrient enrichment. For instance, in North Carolina's Neuse River Estuary, forests account for 17% of total nitrogen delivery (NCDEHNR 1993). The adjacent Pamlico Basin reflects a forestry contribution for nitrogen of about 10% (NCDEHNR 1994).

Sediment yields from silviculture in the coastal zone are not considered a substantial threat to EFH. Sedimentation is typically lower than Piedmont or mountain sites as a result of lower terrestrial slopes and enhanced opportunity for deposition in the slower moving receiving waters, including canal systems.

Information is poor on forestry contributions to fecal coliform contamination in the southeast. Initial studies have found relationships between elevated runoff rates after clear cutting and fecal coliform delivery, but other factors were also at work (J. Sauber, personal communication). Non-nutrient pollution from silviculture is also of concern, though poorly documented. A number of studies have shown release of mercury and other metals from peat soils subjected to intensive drainage (Evans et al. 1984; Gregory et al. 1984). Elevated mercury concentrations also have been found in organic sediments in riparian coastal watersheds (Otte et al. 1987). In North Carolina, fish from the Waccamaw Basin show elevated mercury levels (NCDEHNR 1996b) and metal levels in sediments are elevated throughout the Albemarle-Pamlico Region due to a variety of sources (Riggs et al. 1991). Although not directly related to silviculture, real estate ventures by timber companies have converted large areas of forest land to residential property. This has resulted in much faster rates of surface water runoff and discharge of waters that contain higher concentrations of pesticides and fertilizers. In coastal areas and in inland locations bordering rivers and streams, property values may be greatly increased and the conversion of forest land to residential and commercial property is proceeding at a rapid rate. Further, connected actions, such as the construction of access roads to silviculture sites increase the overall area of impact.

6.1.1.4 Urban/Suburban Development

The southeastern United States has undergone one of the highest rates of landscape changes in the country, in part due to changing demographics and land use practices over the last few decades (Milesi et al. 2003). In particular this trend has been observed in the coastal regions of the southeast. Nine of our nation‘s ten largest cities are located in coastal watersheds (Bureau of the Census 2002). With its extensive and accessible coastline and mild winter climate the southeast coastal zone is one of the nation‘s fastest growing regions. The regional growth rate here is more than four times the national average (Chambers 1992) and between 1980 and 2010 the South Atlantic coastal population is expected to increase by as much as 73% (Chambers 1992). While coastal watershed counties comprise less than 25% of the land area in the United States, they are home to more than 52% of the total U.S. population. A study of coastal population trends predicts average increases of 3,600 people a day moving to coastal counties, reaching a total population of 165 million by 2015. These figures do not include the 180 million people who visit the coast every year (U.S. Commission on Ocean Policy 2004).

As the population increases so does urbanization. People require homes and related infrastructure such as roads, schools, water and sewer facilities, power transmission lines, etc. These needs often are met at the expense of EFH since residential growth has led to large scale modification of wetlands and other irreplaceable environments. Research indicates that nearby water bodies can become seriously degraded when more than 10% of the watershed is covered by roads, parking lots, roof tops, and similar surfaces (NRDC 1999). Tiner (1984) estimates that about 8% of the national rate of wetland losses that occurred from the mid 1950's to the mid 1970's resulted from urban development. Other effects of urbanization include increased sedimentation rates during and after construction, loss of surrounding upland recharge areas and wetland biofiltration and habitat functions. These effects could be ameliorated to some extent by maintaining sufficient buffers and less exploitive developmental patterns. The effect could be dampened by constructing within existing land contours and removing only the canopy necessary for project success. Currently in areas under development all existing vegetation is cleared and burned, all contours are removed and wetland soils are removed and replaced or filled over. Buffer ordinances, if they exist, are typically between 30 and 50 feet adjacent to estuarine systems; this width is not strongly supported by scientific literature.

Chemicals produced and used by people also find their way into the waters as point-source and non-point source runoff. Examples include oil from roads and parking lots, and pesticides, herbicides, and fertilizers from golf courses and residential lawns. This has reduced water quality in waters and wetlands adjacent to urban developments. As a result, the quality of EFH is often much reduced and thousands of acres of shellfish waters are closed. The South Carolina Department of Natural Resources‘ (SCDNR) Tidal Creek Project (TCP) provides insight into the effects of urbanization and suburban development on South Carolina tidal creeks (Holland et al. 1996, 2004; Sanger et al. 1999a,b). This study has implications for other states as well. The study examines developmental effects on salinity, dissolved oxygen (DO), and pollution in tidal creeks having trophic, shelter, and nursery functions required by commercially, recreationally, and ecologically important fish and invertebrates. The study reveals the complexity of the environmental and ecological factors involved and shows correlations between development; changes in tidal creek chemical, physical, and biological characteristics; and alteration of species distribution, composition, and abundance. In general, the physical-chemical characteristics of headwater creeks were significantly altered when the amount of impervious surface exceeded 10-20% and living resources were altered when the amount of impervious surface exceeded 20-30% cover.

The TCP identified salinity as a major factor in controlling the distribution and abundance of living marine resources (Holland et al. 1996, 2004). In watersheds having the greatest areas of roofs, roads, and parking lots it was found that recruitment and colonization by benthic fauna in these areas was less predictable than in more stable environments. TCP confirms that suitable DO concentrations are essential for maintaining balanced indigenous populations of fish, shellfish, and other aquatic biota in tidal creeks and that pollution-related decreases in DO may pose the greatest threat to the environmental quality of estuaries (Holland et al. 1996, 2004). With respect to contaminants, an examination of both metal and organic contaminants taken in connection with the TCP study indicate that metal contaminants were 2-10 times lower in forested watersheds compared to industrial/urban watersheds (Sanger et al. 1999a). Organic contaminants, such as PAHs, PCBs, and DDT were also much lower in forested creeks compared to the industrial/urban creeks.

In another study at larger watershed scales (14-digit Hydrologic Unit Code), Van Dolah et al. (in press), noted significant correlations in the concentrations of inorganic and organic contaminants and fecal coliform bacteria concentrations with the amount of urban suburban development. The correlation between contaminant concentrations and urban/suburban land cover, was stronger in tidal creek habitats within these watersheds, compared to data obtained from larger open water habitats within these watersheds. Additionally the percentage of sites within the watersheds having elevated contaminants and fecal coliform bacteria was much greater in watersheds having greater than 50% urban/suburban development compared with those watersheds having less than 30% urban/suburban cover.

As the linkage between urban and suburban development and declining fish abundance and health or quality is reinforced, the implications of anticipated population growth in coastal areas become even greater. This situation is especially critical in the southeast where recreationally and commercially important species are almost totally dependent on estuaries for their survival and for about $5.5 billion in annual commercial fishery benefits (Chambers 1990).

Potential Threats to EFH from Urban/Suburban Development

Potential threats include conversion of wetlands to sites for residential and related purposes such as roads, bridges, parking lots, commercial facilities, reservoirs, hydropower generation facilities, and utility corridors; direct and/or nonpoint-source discharge of fill, nutrients, chemicals, cooling water, and surface waters into ground water, streams, rivers and estuaries; hydrological modification to include ditches, dikes, flood control and other similar structures; damage to wetlands and submerged bottoms; and cumulative and synergistic effects caused by association of these and other developmental and non-developmental related activities.

Wetlands and other important coastal habitats continue to be adversely and irreversibly altered for urban and suburban development. (Note: certain related activities such as navigation are discussed in later sections). Of major concern is the piecemeal elimination of wetlands by filling for houses, roads, septic tank systems, etc. Wetland filling can directly eliminate or diminish the functional value of EFH and associated areas and resources. While the total area of wetlands affected by development is unknown, the rate of conversion was once estimated at 8% of the national average loss of 458,000 acres or 36,640 acres per year (Tiner 1984). Requests to alter coastal areas remain high and between 1981 and 1996, for example, in the southeast the NOAA Fisheries Service reviewed more than 23,871 proposals requesting to alter wetlands for housing, shoreline structures, docks, roadways, and other related activities. A survey of 5,622 of these proposals involved 19,729 acres of wetlands (see Tables 26, 27, 28, & 29). Between 1996 and 2006, NOAA Fisheries Service reviewed an additional 1,962 applications to fill wetlands to construct housing and 1,886 applications for shoreline modifications. Note that the acreage cited would not include wetland impacts from nationwide permits, dock footprint, loss of bottom area under pilings, or a great percent of shoreline fortification that is designated as —*di minimus*‖ by the COE and typically can range one to three feet from an existing seawall or bulkhead.

Another major threat posed by urban and suburban development is that of non-point-source discharges of the chemicals used in day to day activities, in operating and maintaining homes, roads, vehicles, etc. In addition to chemical input, changes that affect the volume, rate, location, frequency, and duration of surface water runoff into coastal rivers and tidal waters are likely to be determinants in the distribution, species composition, abundance, and health of southeastern fishery resources and their habitat.

Results of various studies in the South Atlantic Bight indicate that chemical contaminants from industrial, urban/suburban, and agricultural sources may cause impacts in estuarine ecosystems. Highest contaminant concentrations and greatest impacts were observed in the headwaters of small tidal creeks, which are nursery grounds for fish, crustaceans and molluscs. Protection and management of nonpoint-source runoff loading into these watersheds is essential in protecting habitat quality (Scott et al. 1997). In the long-term, impacts of chemical pollution (e.g., petroleum hydrocarbons, halogenated hydrocarbons, metals, etc.) are likely to adversely impact fish (Schaaf et al. 1987). Despite current pollution control measures and stricter environmental laws, toxic organic and inorganic chemicals continue to be introduced into marine and estuarine environments. Results of the previously mentioned TCP investigation confirm that suitable DO concentrations are essential for maintaining balanced indigenous populations of fish, shellfish, and other aquatic biota in tidal creeks and that pollution related decreases in DO may pose the greatest threat to the environmental quality of estuaries. The study found that:

DO in tidal creeks fluctuated with phases of the moon, time of day, and tidal stage.

DO in tidal creeks in developed and undeveloped watersheds often did not meet the state water quality standard of 4mg/L.

The most stressful DO levels occurred during early morning and at night-time low tides.

The DO levels in tidal creeks in developed watersheds were less predictable and had greater unexplained variance than those of undeveloped watersheds.

Point in time DO measurements in tidal creeks do not adequately represent exposure of living resources stressful low DO levels.

Living resources in tidal creeks in developed watersheds were more frequently exposed to stressful low DO levels than those inhabiting tidal creeks with undeveloped watersheds.

The factors that contribute to low DO in South Carolina tidal creeks need further study and a DO budget for tidal creeks and associated saltmarshes is needed so that the major factors controlling low DO conditions can be identified and addressed from a management perspective.

With respect to contaminants, bioassays of sediments taken in connection with the TCP study indicate that potentially toxic conditions for living marine resources may occur in the upper reaches of tidal creeks in developed watersheds. Polyaromatic hydrocarbons in sediments were highest where surface runoff from roads was discharged into tidal creeks and sediment bound pesticides were more prevalent in the marsh and near houses.
(Preceding is a summary taken from Holland et al. 1996).

Finally with regard to urban/suburban development, and in particular regard to nonpoint- source discharges, the South Carolina Statewide Water Quality Assessment for FY 1992-1993 (SCDHEC 1994) provides an indication of the role of non-point source discharges in one southeastern state. According to the Assessment:

Nonpoint-source (NPS) pollution is the most responsible factor for nonsupport of classified water uses in rivers, lakes, and estuaries in the state.

Of the 26,313 river miles assessed via water quality monitoring stations, 10,534 miles, or 40%, were determined to be partially supporting or not supporting overall use. NPS sources of pollution were identified as the contributing factor 33% of the time. These NPS sources included agriculture, pasture land, silviculture, construction, urban runoff/storm sewers, resource extraction, and hydromodification.

South Carolina has approximately 945 square miles of estuaries, including marshes. The assessment analyzed data collected from 342 square miles of estuaries. About 30% of the estuarine areas do not fully support overall use. NPS pollution sources were identified as the contributing factor 38% of the time.

Of the 135 shellfish areas assessed, 63% were impacted by NPS, including marinas, 22% were impacted by point sources, and 27% were unconditionally approved (the percentages totaled exceed 100% due to multiple source impacts). The South Carolina NPS Task Force listed the 32 highest priority water bodies/watersheds that are targeted for implementation action. Of these water bodies/watersheds, 15 are located in the coastal zone. Sixty-two watershed units are located in the coastal zone. Based on information from the Statewide Assessment and from more recent Watershed Water Quality Management Strategies, 44% of these units have been impaired by NPS pollution; 39% have been impaired by unknown sources of pollution; 24% have been impaired by point sources; 16% have been impaired by natural or other sources; and 30% have no known impairment [The percentages totaled exceed 100% due to multiple source impacts. Also, based on the Statewide Assessment, 38 of the 62 watershed units (or 61%) have not been fully assessed]. Point source discharges related to urbanization derive mainly from municipal sewage treatment facilities or storm water discharges that are controlled through Environmental Protection Agency (EPA)-mandated regulations under the Clean Water Act and by state water quality regulations. Threats related to these discharges are probably less important than the other factors previously discussed because efforts are underway to improve treatment. The primary concerns with municipal point-source discharges involve treatment levels needed to attain acceptable nutrient inputs and overloading of treatment systems due to rapid development of the coastal zone. It is also important to consider that the portion of water entering estuaries from sewage treatment plants is increasing. In locations where treatment is poor, or water conditions are unsuitable for adequate dilution of discharges, EFH may be adversely affected. Of primary concern is excessive nutrification of receiving waters, but other factors such as those associated with nonpoint -source discharges also apply. The EPA withdrew the storm water Phase II direct final rule published on April 7, 1995 (60 FR 17950) and promulgated a new final rule in its place (60 FR 17958). This action by the EPA instituted changes to the National Pollutant Discharge Elimination System (NPDES) stormwater permit application regulations under the Clean Water Act for Phase II dischargers. Phase II dischargers generally include all point-source discharges of storm water from commercial, retail, light industrial and institutional facilities and from municipal separate storm sewer systems serving populations of less than 100,000. This rule establishes a sequential application process in two tiers for all Phase II stormwater discharges. The first tier provides the NPDES permitting authority flexibility to require permits for those Phase II dischargers that are determined to be contributing to a water quality impairment or are a significant contributor of pollutants to waters of the U.S. ―Permitting authority‖ refers to the EPA or States and Indian Tribes with approved NPDES programs. The EPA expects this group to be small because most of these types of dischargers have already been included under Phase I of the storm water program. The second tier includes all other Phase II dischargers. This larger group will be required to apply for permits by the end of six years, but only if the Phase II regulatory program in place at that time requires permits. The EPA has stated that it is open to, and committed to, exploring a number of non-permit control strategies for the Phase II program that will allow efficient and effective targeting of real environmental problems. As part of this commitment, the EPA has initiated a process to include stakeholders in the development of a supplemental Phase II rule under the Federal Advisory Committee Act. This rule was finalized March 1, 1999 and determines the nature and extent of requirements that apply to the various types of Phase II facilities prior to the end of the six-year application period defined by the rule. However, in practice, the EPA‘s NPDES for Phase II dischargers program, can be slow to implement and has limited enforcement authority. Further, stormwater requirements in the State of Florida have resulted in the loss/conversion of wetlands as required treatment ponds are commonly placed in wetlands whose capacity to assimilate contaminates far exceeds any benefit provided by the area loss for stormwater abatement. Further conversion of wetlands to stormwater ponds permanently eliminates these areas ability to contribute dissolve and particulate detrital organic carbon and their ecological habitat functions. These conversions are not seen or recorded as wetland losses although the lost ecologically contribution of these areas has an enormous impact on fisheries.

6.1.1.5 Transportation Transportation projects such as the construction and maintenance of bridges and roadways typically involve long-term planning and permit consultation with NOAA Fisheries Service. Such projects can occur over estuarine waters, within estuarine emergent wetlands, and/or other important wetlands that are hydrologically connected to tidal waters. From 1996 to 2005 NOAA Fisheries Service reviewed 2,352 actions related to transportation. Potential threats to EFH from transportation projects *Potential threats include fragmentation of the ecosystem by isolation and bifurcation of EFH, storm water discharges and runoff, shading of submerged aquatic vegetation from bridges, and blasting associated with bridge or structure demolition.* Transportation project can lead to habitat fragmentation, which results in the isolation of EFH from certain life history stages of recreationally and commercially important fisheries. This isolation limits the food chain by not allowing certain assemblages of organisms to easily traverse from one ecotype to another. This is especially true for fisheries such as the snapper grouper complex that use mangroves swamps and seagrass beds for one or more life history stages. This fragmentation could also potentially limit movements of catadromous and anadromous fishes by isolating populations from a spawning or nursery ground. Fragmentation can also result in the isolation of large tracts of freshwater wetlands. Through this isolation, the trophic functions provided by these wetlands are limited and allochthonous input is cut off to downstream estuaries and EFH. Flushing of upstream wetlands and EFH can be impacted by fragmentation. If mitigation measures (e.g., culverts and bridges) are not taken to maintain adequate flow on both sides of a roadway, waters can become stagnant and limit the benefits to commercial and recreational fisheries. Storm water discharges are a concern where bridges or roadways cross or are adjacent to EFH. Runoff from roadways could impact EFH if water is not collected and treated prior to discharge. The treatment of the storm water, including surface water management systems, should be located outside of EFH. Blasting and demolition pose threats to EFH and managed fisheries. Direct and indirect impacts to EFH should be avoided and best management practices utilized when demolition occurs. This can include detonating small charges (otherwise known as test blasts or fish scares) to direct fish away from the area where the demolition will take place. Bubble curtains are also used in some cases to minimize fish kills. Direct and indirect affects to EFH can also result from construction. Submerged aquatic vegetation can be impacted directly or indirectly from the installation of pilings and shading associated with bridges. The areas adjacent to bridges can be impacted as well from the shadow cast from the structure. These impacts must be considered when evaluating the effects of a transportation project on EFH.

6.1.1.6 Industrial/Commercial Activities The southeastern U.S. is a prime location for industrial siting. The climate is favorable, economic incentives exist, land is readily available and relatively inexpensive, an adequate labor base exists, and the infrastructure for shipping of supplies and products is well developed. Further, the region‘s many rivers and streams provide an abundance of water needed for textile mills, paper mills, and heavy manufacturing (e.g., steel fabricating) and other similar facilities. In addition to a favorable setting for industrial development, commercial growth is ever expanding. Although less conspicuous in many areas, the tourism industry also is a vital part of the coastal economy and many of the South‘s most popular vacation spots are located on or near the coast. With expansion of this industry, new hotels, related businesses, marinas, roads, and other facilities are being built. The increase in visitors and resource users is expected to continually grow and may diminish only when, as a result of overuse and development, the environmental quality of the area is reduced. Population growth and tourism bring many benefits to coastal communities, including new jobs and businesses and enhanced educational opportunities. Burgeoning industries associated with tourism and recreation in coastal areas (such as hotels, resorts, restaurants, fishing and dive stores, vacation housing, marinas, and other retail businesses) have created one of the nation‘s largest and fastest-growing economic forces (U.S. Commission on Ocean Policy 2004). In just four southeast Florida coastal counties, recreational diving, fishing, and ocean-watching activities generate $4.4 billion in local sales and almost $2 billion in local income annually (Johns 2001) and more than 2.9 million people visit the Florida Keys each year (Leeworthy and Vanasse 1999). Potential Threats to EFH from Industrial/Commercial Activities Potential threats include conversion of wetlands to industrial and appurtenant sites such as roads, parking, and administrative and distribution centers; point and nonpoint-source discharge of fill, nutrients, chemicals, cooling water, air emissions, and surface and ground waters into streams, rivers, estuaries and ocean waters; hydrological modification to include ditches, dikes, water and waste lagoons; intake and discharge systems; hydropower facilities; and cumulative and synergistic effects caused by association of these and other industrial and non-industrial related activities. In addition to ongoing activities, previous industrial and commercial activities have, in many locations, led to deposition of harmful materials that are subject to resuspension and reincorporation into aquatic food chains

Industrial and commercial development can affect EFH in a number of ways. Most apparent is the conversion of wetlands and upland buffers to sites for buildings, plants, parking, storage and shipping or materials and products, and treatment or storage of wastes or by-products. Because of an abundance of hard impervious surfaces associated with industrial and commercial operations they are often major contributors of non-point-source contaminants into aquatic environments, including those that support EFH. Many industries, (e.g., paper mills), consume and pollute large volumes of water needed to sustain a healthy coastal environment. Industries may also produce airborne emissions that contain contaminants. These contaminants have been shown to reappear in coastal waters and EFH. A readily observable example is acidification of waters from atmospheric deposition of industrial emissions and coal fired power plants.

Commercial development along the South Atlantic coast also has been extensive and relatively few coastal areas are free of commercial development. Past development practices were especially detrimental and before adequate regulation it was not uncommon to excavate and fill marshes and shallow water environments for residential, commercial and industrial uses. Such practices have been largely eliminated because most of the coast is either developed or protected from such practices. However, uplands are a decreasing commodity in the coastal zone and the demand for filling wetlands and other aquatic sites is likely to persist. Consequently, proposals aimed at altering wetlands for commercial and other purposes will continue to require local, state, and federal involvement if significant adverse impacts to EFH are to be effectively controlled.

The total amount of EFH that has been eliminated or degraded by commercial and industrial development is unknown, but it is extensive. NOAA Fisheries Service data show that between 1981 and 1996, 1,466 proposals were received for industrial and commercial development in wetlands that are subject to the regulatory provisions of the Rivers and Harbor Act and Section 404 of the Clean Water Act. In association with this, 430 proposals sought approval to alter about 3,202 acres of EFH (see Tables 26, 27, 28, & 29). Between 1996 and 2006, NOAA Fisheries Service reviewed approximately 2,126 applications for industrial and commercial activities and associated wetland impacts in the South Atlantic area. Point-source discharges from commercial activities may be similar to those associated with urban and suburban development. Accordingly, the information and discussions contained in Section 4.1.1.3 should apply. Pollution and water use may alter the flow, pH, hardness, dissolved oxygen, and chemical composition parameters that affect individuals, populations, and communities (Carins 1980). Within aquatic systems industrial point-source discharges also may alter species and population diversity, nutrient and energy transfer, productivity, biomass, density, stability, connectivity, and species richness and evenness both at the point of discharge and downstream locations (Carins 1980). Growth, visual acuity, swimming speed, equilibrium, feeding rate, response stimuli, predation rate, photosynthetic rate, spawning seasons, migration routes, and resistance to disease and parasites of finfish, shellfish, and related organisms may be altered by chemical and thermal changes. Some industries, such as paper mills, are major water users and associated effluent can dominate and control conditions in substantial portions of rivers and other water bodies where they are located. Usually parameters such as substrate, currents, dissolved oxygen, pH, nutrients, temperature, and suspended materials are key factors affecting the distribution and abundance of EFH. The direct and synergistic effects of other discharge components such as heavy metals and various chemical compounds are not well understood, but current research shows that these constituents may be of greater importance than previously thought. For example, more subtle factors such as endocrine disruption in aquatic organisms and reduced ability to reproduce or compete for food are being uncovered (Scott et al. 1997).

The cumulative effect of many types of discharges on various aquatic systems also is not well understood, but attempts to mediate their effects are reflected in various water quality standards and programs in each state and within the various water systems. Industrial wastewater effluent is regulated by the EPA through the NPDES permitting program. This program provides for issuance of waste discharge permits as a means of identifying, defining, and controlling virtually all point-source discharges. The complexity and the magnitude of effort required to administer the NPDES permit program limit overview of the program and federal agencies. Consequently, the NOAA Fisheries Service and the FWS generally do not provide comments on NPDES application notices. For these same reasons, it is not presently possible to estimate the singular, combined, and synergistic effects of industrial (and domestic) discharges on aquatic ecosystems.

Where chronic non-point-source discharges and accidental releases of harmful or toxic substances mix, especially harmful effects on aquatic life and habitat, including EFH, is likely. An added concern with industrial operations is the release of contaminants into the atmosphere. Such materials may be transported various distances and directly and indirectly deposited into aquatic ecosystems (Baker et al. 1993). In the southeast, surface water acidification and mercury accumulation in sediments are of particular concern since sources of these material lie in other regions and are not subject to local and regional (southeastern) controls. In view of this, the regulation of surface water contamination from atmospheric pollution should be addressed from a local, regional, and international perspective.

6.1.1.7 Navigation

Support for navigation in the southeast Atlantic region has resulted in widespread modification of subtidal and intertidal areas used by commercial and recreational vessels. Significant modification to offshore habitats has also occurred and this is discussed in the Marine/offshore Processes Section. Primary threats to EFH from navigation in estuarine waters include the construction, maintenance, and expansion of thousands of miles of waterways such as the Atlantic Intracoastal Waterway and the myriad of other channels that lead to marinas, ports, turning basins, and harbors. Construction and maintenance of existing ports and recreationally-based marinas and basins have altered substantial areas of EFH. Expansion of existing channels and waterways to accommodate larger vessels, primarily mega-yachts and Post-Panamax vessels, is becoming an increasing threat to inshore EFH, namely seagrasses. Dredged material disposal and disposal of contaminated sediments is also an issue. Filling of wetlands and conversion of EFH from shallow to deep water habitats are persistent threats associated with new facilities and the maintenance and expansion of existing facilities. Where coastal inlets are stabilized and maintained for navigation purposes effects on nearshore environments and fish and invertebrate populations may be substantial in addition to blockages of littoral sediment transport.

A second major concern related to navigation is the host of environmental problems associated with vessel operations. These range from contamination of water by oil, grease, anti-fouling paints, and discharges of sewage, garbage, and debris to the direct destruction of EFH by grounding, anchor damage, propwashing, scarring, etc. Most physical damage is accidental; however, activities such as propwashing could be avoidable for example, through better signage in waterways near shallow SAV habitats and a greater of level of enforcement. However, regarding the latter, it should be recognized that many State and local enforcement programs are severely understaffed and underfunded.

Potential Threats to EFH from Navigation

Navigation related threats to EFH located within estuarine waters can be separated into two categories: Navigation support activities and vessel operations. Navigation support activities include, but are not limited to, excavation and maintenance of channels (includes disposal of excavated materials); construction and operation of ports, mooring and cargo handling facilities; construction and operation of ship repair facilities; and construction of channel stabilization structures such as jetties and revetments. Potentially harmful vessel operations activities include, but are not limited to: discharge or spillage of fuel, oil, grease, paints, solvents, trash, and cargo; grounding/sinking/prop scaring in ecologically/environmentally sensitive locations; exacerbation of shoreline erosion due to wakes; salt water intrusion into brackish systems; and transfer and introduction of exotic and harmful organisms through ballast water discharge.

Navigation Support Activities

The most conspicuous navigation-related activity in many estuarine waters is the construction and maintenance of navigation channels and the related disposal of dredged Fishery materials. The amount of subtidal and intertidal area affected by new and maintenance dredging is unknown, but undoubtedly great. Orlando et al (1988) analyzed 18 major east coast estuaries from North Carolina to Florida east coast and found over 703 miles of navigation channels and 9,844 miles of shoreline modifications related to navigation works. Between 1981 and 1986 the NOAA Fisheries Service received over 4,877 proposals for new navigation projects in the South Atlantic region. A detailed analysis showed that 1,692 of these proposals involved plans to alter 24,825 acres of EFH through dredging and filling (Tables 26, 27, 28, & 29). From 1996-2006, NOAA Fisheries Service received 1,055 applications for maintenance dredging related activities and 720 application-related to construction of marinas and navigation channels in the South Atlantic area.

However, the potential threats to EFH from widening and deepening navigation channels warrant close examination. In many South Atlantic areas, marina owners and inland navigation districts have submitted applications to the Corps of Engineers for widening and deepening activities to accommodate mega-yachts and provide navigation access for mega-yacht vessels to private interior berthing, testing, and repair facilities located in the vicinity of inlets. Mega-yachts are typically classified as private luxury recreational motor or sailing vessels that are greater than 80 feet in length and there are approximately 735 that would access South Atlantic navigation channels (FWS 2005). In Palm Beach County, Florida alone proposed impacts associated with Atlantic Intracoastal Waterway and other channel expansion projects exceed 30 acres of seagrass habitat within Lake Worth Lagoon and typically involve dredging deeper than the Water Resources Development Act Congressionally authorized depths, for example from -10 NGVD to -16 NGVD. The seagrass habitats located around inlets are typically unique and ecologically significant due to the influence of clear oceanic waters that enter through the inlet and provide water clarity that cannot be found in locations further from the inlet. For example, the seagrass habitat located in close proximity to the Lake Worth Inlet (Florida) allows seagrass to grow at depths of over 10 feet as opposed to more remote seagrass habitat, which may only reach depths of 4 feet.

According to a FWS report, the overriding factor in the decline of estuarine and marine wetlands in the U.S. between 1998 and 2004 was the loss of emergent saltmarsh to open saltwater systems due to and manmade activities such as dredging, water control, and commercial and recreational boat traffic (Dahl 2006). While channel excavation itself is usually visible only from the surface while the dredge or other equipment are in the area, the need to dispose of excavated materials has left its mark in the form of confined and unconfined disposal sites, including those that have undergone human occupation and development. Chronic and individually small discharges and disturbances routinely affect water and substrate and may be significant from a cumulative or synergistic perspective. EFH impacts include, direct removal/burial of organisms as a result of dredging and placement of dredged material; turbidity/siltation effects, including increased light attenuation from turbidity; contaminant release and uptake of nutrients, metals, and organics; release of oxygen consuming substances; noise disturbance to aquatic and terrestrial organisms; and alteration of hydrodynamic regimes and physical habitat.

The maintenance and stabilization of coastal inlets also is a prominent navigation activity. Studies and reports by the COE, the NOAA Fisheries Service, and others link jetty construction to possible changes in plankton movement (USACE 1980; USDC 1991; Miller 1988; Miller et al. 1984). This is a major concern since significant modification of inlet hydrodynamics may diminish the ability of sub-adult fish and invertebrates to reach estuarine nursery grounds. Where significant reductions in recruitment (into estuarine waters) of desirable species is realized, production declines in ecologically, recreationally and commercially important species may result. The use of jetties to stabilize navigation channels at coastal inlets also has been linked to changes in coastal geomorphology that affects nearshore environments. For example, coastal geologists have expressed concern that construction of jetties at Oregon Inlet on the North Carolina Outer Banks could cause catastrophic beach erosion and accelerate barrier island migration (Pilkey and Dixon 1996). Such change could adversely affect the extensive and highly productive submerged vegetation beds which are located behind the coastal barriers.

The relocation of freshwater/saltwater transition zones due to channel deepening may be, in some cases, responsible for significant environmental and ecological change. As an example, salinity shifts after channel deepening and water diversion in the lower Savannah River caused vegetation shifts from freshwater to brackish species in surrounding wetlands. In the lower Savannah River, increased mortality of sub-adult striped bass also has been linked to salinity increases caused by navigation-related modifications such as channel deepening and flow diversion. Modifications that increase estuarine salinities may also create more hospitable conditions for shellfish predators such as boring sponge, oyster drill, and keyhole limpet.

In southeast Florida, increased channelization by dredging and the addition of rocky structures may have favored shifts from estuarine assemblages to reef assemblages because of comparatively higher abundances and diversities of incoming ichthyoplankton, higher inshore salinities, and replacement of vegetation with hard structure that favors reef species (Lindeman 1997). Similar situations are possible in other watersheds where dredging and dredged material disposal are prominent features; however, little documentation of these changes is available. Another example includes the St. Johns River in North Florida. The St. Johns River's watershed encompasses 50% or more of the east coast of Florida flowing north and in the 1800's flowed out onto an alluvial flood plan of shallow non-navigable sand bars. Construction of the Jacksonville Port has deepened and channelized the river mouth, now -52 NGVD. As a result, the amount of salt water intrusion has completely altered the estuarine system of the lower St. Johns River.

The expansion of ports and marinas has become an almost continuous process due to economic growth, competition between ports, and significant increases in vessel numbers and vessel size. Elimination or degradation of aquatic and upland habitats are commonplace since port and marina expansion almost always require the use of open water, submerged bottoms, and riparian zones. Ancillary related activities and development often utilize even larger areas, many of which provide water quality and other functions needed to sustain living marine resources. Vessel repair facilities use highly toxic cleaners, paints, and lubricants that can contaminate waters and sediments. Modern pollution containment and abatement systems and procedures can prevent or minimize toxic substance releases; however, constant and diligent pollution control efforts must be implemented. The operation of these facilities also poses an inherent threat to EFH by adversely affecting water quality in and around these facilities. The extent of the impact usually depends on factors such as flushing characteristics, facility size, location, depth, and configuration. When facilities such as marinas are constructed it is common to restrict shellfish harvest in a set or established zone that may be affected by sewage and other hazardous materials. It is now common practice to consider safe zones with respect to public health and aquatic resources when siting marina and port facilities.

Major ports in the South Atlantic region include Morehead City and Wilmington in North Carolina; Georgetown, Charleston, and Port Royal in South Carolina; Savannah and Brunswick in Georgia; and Fernandina Beach, Jacksonville, Port Canaveral, Port Everglades, Fort Pierce, Palm Beach, and Miami in Florida. Many eastern seaboard ports are subject to proposals to widen and deepen to accommodate Post-Panamax vessels or deep-draft vessels too large to fit through the Panama Canal. Impacts resulting from these projects can be substantial and can involve alternatives to dredge through coral reef, hardbottom habitat, and seagrasses.

In 2005, the Port of Miami, located in Biscayne Bay which is a State of Florida designated Outstanding Florida Water, completed a harbor deepening project that used confined blasting to fracture rock that was too hard to be removed via conventional dredge. In 2004, the Corps of Engineers finalized an Environmental Impact Statement to widen and deepen the entrance channel and other interior areas of the Port to -50 NGVD. The Recommended Plan would impact approximately 415 acres of habitat including over 6.3 acres of seagrass habitat, 28.7 acres of low-relief hardbottom/reef habitat, 20.7 acres of high relief hardbottom/reef habitat, 123.5 acres of rock/rubble habitat, and 236.4 acres of unvegetated bottom habitat (COE 2003).

The COE recently finalized a Reef Report for Port Everglades Outer Entrance Channel Expansion Project that concluded that over 150,000 corals and 21 acres of reef could be lost through proposed expansion activities (COE 2006). This project is in the feasibility phase and the COE proposes to release the draft Environmental Impact Statement in October 2007. In addition to the reef impacts, this project could impact up to 5 acres of seagrass (including one acre of the federally listed *Halophila johnsonii*), 11.55 acres of mangroves (8.48 acres of which are currently held in a conservation easement for impacts from previous Port activities), and 20.09 acres of previously dredged hardbottom, for which no compensatory mitigation is currently proposed (FWS 2005).

Cargo arriving and departing through these ports is diverse and ranges from highly toxic and hazardous chemicals and petroleum products to relatively benign materials such as wood chips. Major spills and other discharges of hazardous materials are uncommon, but are of constant concern since large and significant areas of estuarine habitat and fishery resources are at risk. Expansion of these facilities and certain operation and maintenance activities are likely to occur at the expense of EFH.

There have been recent positive trends in the development of beneficial uses for clean dredged materials. For example, the deepening of the Wilmington Harbor navigation channel in North Carolina generated rock that is being used for creation of an offshore reef. Similar activities are being investigated in connection with planned deepening of Charleston Harbor in South Carolina. These activities will require monitoring to evaluate their success, but if beneficial other uses of dredged material could be developed. On a cautionary note, conversion of one habitat type to another may not be desirable since associated ecological trade-offs could be harmful to desirable or managed species. The classic example of this is the Winyah Bay, South Carolina dredged material disposal site, where submerged and intertidal bottoms have been converted to emergent marsh without any assessment of the ecological role of the disposal site.

Dredging and disposal of excavated materials is a major component of all southeastern ports and many marinas. Dredged materials are often contaminated and extensive testing for heavy metals and other contaminants is required. At many locations finding suitable disposal sites for dredged materials is also difficult and costly. Whenever contaminated dredged materials are placed in offshore waters, or in locations where decant is discharged into surrounding waters there is high probability that these contaminants will reenter aquatic food webs. As existing upland disposal sites are filled this problem is likely to be exacerbated. Already, direct overboard dispersal of dredged material occurs at some location such as in reaches of the Atlantic Intracoastal Waterway in North Carolina. In other locations such as the Savannah River, Georgia, a technique referred to as ―agitation dredging‖ is used. In this case, about 200,000 cubic yards of materials are resuspended from ship berths each year by bottom dragging or by hydraulic excavation with direct disposal into the adjacent navigation channel. In addition, hydraulic bottom scour systems are presently in place in Wilmington, North Carolina, and experimental use of these devices is planned at one facility in Savannah and at the U.S. Navy‘s Kings Bay, Georgia, Submarine Base. The environmental impact associated with the use of this technique is unclear, but significant use of bottom scouring devices could be problematic since planktonic and weak swimming fish and invertebrates could be impinged or entrained in intakes and plumbing, and turbidity and sedimentation could be exacerbated. Of particular concern is those aquatic environments that contain anadromous fish since planktonic and weak swimming fish could be heavily impacted.

An additional, but more limited dredging practice is the prop dredging of bottoms, mostly by recreational vessels, to obtain navigable depths. This practice is generally performed without benefit of state or federal permits and is almost always destructive.

The SAFMC is opposed to open water disposal of dredged material into aquatic systems when adverse impacts to habitat used by fisheries under its jurisdiction are likely. The SAFMC urges state and federal agencies, when reviewing permits considering open water disposal, to identify the direct and indirect impacts such projects could have on fisheries habitat. It is also their view that the conversion of one naturally functioning aquatic system at the expense of creating another (marsh creation through open water disposal) must be justified using the best available information.
Construction of piers and docks also affects EFH, but the degree of the impact is often disputed. Impacts are dependent on the size, location, and number of similar structures in a given area. Pier and dock construction often involves jetting of pilings and this causes temporary and localized affects on EFH due to increased sedimentation and habitat displacement. Sedimentation may be a problem in systems such as SAV that are already stressed and are declining or have marginal value due to low water clarity. The pilings are treated and toxic chemicals are released into the waters and sediments, but this is not perceived to be a major problem since the pilings are eventually covered with encrusting and fouling organisms. Perhaps the greatest threats from piers and docks are those associated with marsh and SAV shading and the erosion, due to wave action, of substrates in the vicinity of support piles. Substantial harm to SAV and benthic communities may also result from secondary effects associated with boat use, including constant grounding due to wave and tidal action.

The overall biological effects of piers and docks has not been well quantified. However, between 1981 and 1996, the NOAA Fisheries Service reviewed requests for almost 6,000 piers and docks along the southeast coast between North Carolina and Florida. Between 1996 and 2006, NOAA Fisheries Service reviewed an additional 7,540 applications to construct docks and pilings. In areas having marginal depths and especially where SAV is present, habitat damage in the vicinity of piers and docks may be substantial and disproportionately large in cases where such structures are abundant (Ludwig et al. 1997). These structures represent a substantial feature in southeastern watersheds and they warrant continued monitoring and regulatory review. In response to this, NOAA Fisheries Service and U.S. Army Corps of Engineers Jacksonville District jointly developed *Dock Construction Guidelines in Florida for Docks or Other Minor Structures Constructed in or over Submerged Aquatic Vegetation, Marsh or Mangrove Habitat* in addition to the *Key for Construction Conditions for Docks or Other Minor Structures Constructed in or over Johnson's seagrass (Halophila johnsonii)* (see http://www.saj.usace.army.mil/permit/hot_topics/Dock_Guidelines/dockindex.htm). In general, these guidelines provide environmentally responsible access to Florida waters.

Vessel Operations
In connection with watercraft operation and support the USEPA (1993) has identified several principal concerns. These include pollutants discharged from boats; pollutants generated from boat maintenance activities; exacerbation of existing poor water quality conditions; pollutants transported in storm water runoff from parking lots, roofs, and other impervious surfaces; and the

physical alteration or destruction of wetlands and of shellfish and other bottom communities during the construction of marinas, ramps, and related facilities.

Marinas and other sites where vessels are moored or operate often are plagued by accumulation of anti-fouling paints in bottom sediments, by fuel spillage, and overboard disposal of trash and wastewater. In areas where vessels are dispersed and dilution factors are adequate, the water quality impacts of vessel operations are likely to be offset to some degree. In a study of marinas in North Carolina it was found that marinas may contribute to increases in fecal coliforms, sediment oxygen demand, and chlorophyll a, and decreases in dissolved oxygen (NCDEHNR 1990). In addition, boating and other activities (e.g., fish waste disposal) may contribute to increased water temperature, bioaccumulation of pollutants by organisms, water contamination, sediment contamination, resuspension of sediments, loss of SAV and estuarine vegetation, changes in sediment composition loss of benthic organisms, changes in circulation patterns, shoaling, and shoreline erosion. Pollutants associated with marinas include nutrients, metals, petroleum hydrocarbons, pathogens, and polychlorinated biphenyls (USEPA 1993).

Marina personnel and boat owners use a variety of boat cleaners, such as teak cleaners, fiberglass polish, and detergents and cleaning boats over the water, or on adjacent upland, creates a high probability that some cleaners and other chemicals will enter the water (USEPA 1993). Copper-based antifouling paint is released into marina waters when boat bottoms are cleaned in the water (USEPA 1993). Tributyl-tin, which is a major environmental hazard, has been largely banned except for use on military vessels. Fuel and oil are often released into waters during fueling operations and through bilge pumping. Oil and grease are commonly found in bilge water, especially in vessels with inboard engines, and these products may be discharged during vessel pump out (USEPA 1993).

Sewage and other wastes discharged from recreational boats may be most problematic in marinas and anchorage sites where vessels are concentrated. Despite existing federal and state regulations involving discharges of sewage and other materials, detection and control of these activities are difficult and discharges still occur. According to the 1989 American Red Cross Boating Survey, there were about 19 million recreational boats in the U.S. (USEPA 1993). About 95% of these boats were less than 26 feet in length and a large number of these boats used a portable toilet, rather than a larger holding tank. Given the large percentage of smaller boats, facilities for the dumping of portable toilet waste should be provided at marinas that service significant numbers of boats under 26 feet in length (USEPA 1993).

Increased recreational boating activity may contribute significantly to pollution of southeastern coastal waters by petroleum products. All two-cycle outboard engines require that oil be mixed with gasoline, either directly in the tank or by injection. That portion of the oil that does not burn is then ejected, along with other exhaust products, into the water. In 1990, 52,030 boats were registered in coastal North Carolina (North Carolina Wildlife Resources Commission, personal communication). Based on this number, conservative estimates indicate that about 84,549 gallons per year of oil (in fuel) is discharged annually into North Carolina‘s coastal waters (Hoss and Engel 1996). For comparison purposes, hydrocarbon discharges for coastal North Carolina in 1982, from boating and urban runoff are about 470 and 2,270 tons, respectively. Increased use of personal water craft such as jet skis has added to the volume of hydrocarbon being introduced into southeastern waters since the engine exhaust from these vessels is discharged directly into the propellant water jet. Similar problems are inferred for other states and areas having high concentrations of boats. The chronic effects of vessel grounding, prop and jet ski scarring, and anchor damage are generally more problematic in conjunction with recreational vessels. While grounding of ships and barges is less frequent, individual incidents can have significant localized effects. Propeller damage to submerged bottoms occur in all areas where vessels ply shallow waters. In addition, direct damage to multiple life stages of associated organisms, including egg, larvae, juveniles, and through water column de-stratification (temperature and density), resuspending sediments, and increasing turbidity (Stolpe 1997; Goldsborough 1997) have been observed in connection with vessel operation. This damage is particularly troublesome in North Carolina and Florida, the two South Atlantic states with submerged rooted vegetation in their coastal waters. In North Carolina, no official quantitative estimate of SAV damage has been performed; however, preliminary observations indicate that damage to the state‘s 135,000 acres of SAV is localized around marinas or other boat access points (R.L. Ferguson, personal communication). Scarring estimates for Florida indicate that about 173,000 of the state‘s 2.7 million acres of SAV are scarred (Sargent et al. 1995). On the Atlantic coast of Florida there are about 69,360 acres of SAV and 3,770 acres (18%) have been scarred by prop and other water craft action.

The ever increasing number of registered power boats along the South Atlantic coastal zone, and those temporarily entering coastal areas through tourism ensure that this threat is likely increase over time. Power boat registrations on Florida‘s east coast, not including sailboats, totaled 108,048 vessels in 1992-93. Of these, 95% were pleasure craft (Sargent et al. 1995).

The rapid increase in popularity of jet skis or ―personal water craft‖ is also problematic. While these vessels are not propeller driven, the water jet removes sediment from seagrass roots and rhizomes and can cause damage. Further, these craft can operate in shallower waters and can access seagrass areas with relative ease, in addition to direct impacts to grassbeds. These machines are exceedingly loud and can create large wakes. It is reasonable to hypothesize that the audio and physical environment of shallow nursery areas may be disrupted in manners which stress postlarval life stages. The degree of stress is currently uninvestigated.

Incidences of commercial groundings are few, but where they occur on hard bottom habitats damage may be extensive and long-term. For example, groundings in the Florida Keys National Marine Sanctuary have caused extensive damage to coral reefs and signs of recovery are slow to appear.
The cumulative effect of anchor scarring in seagrass beds is not as damaging as that caused by propeller and jet powered vessels. On coral reefs, however, damage caused by anchoring of recreational boats is significant (Davis 1977). Dragging or pulling anchors through coral beds breaks and crushes the coral, destroying the coral formation. Most reef damage of this type occurs in the Florida Keys and in nearshore waters.

The effects of vessel induced wave damage have not been quantified, but may be extensive. The most damaging aspect relates to the erosion of intertidal and SAV wetlands located adjacent to marinas, navigation channels, and boating access points such as docks, piers, and boat ramps. Wake related erosion in places along the Atlantic Intracoastal Waterway and elsewhere is readily observable and has undoubtedly converted substantial areas of emergent wetlands to less important habitat such as submerged bottom. In heavily trafficked areas bottoms may become unstable and colonization by bottom dwelling organisms may not be possible. Indirect effects may include the resuspension of sediments and contaminants that can affect EFH. Where sediments flow back into existing channels, the need for maintenance dredging, with its attendant impacts, may increase.

The introduction of exotic species by vessel operations is linked largely to the world wide movement of commercial vessels. Exotic species may be brought into the U.S. by several methods, but capture and release in ballast waters is of most concern. With the introduction of the zebra mussel into the Great Lakes and its rapid dispersal into other waters, considerable attention is being directed at this problem. According to one estimate, two million gallons of foreign ballast water are released every hour into U.S. waters (Carlton 1985). This possibly represents the largest volume of foreign organisms released on a daily basis into North American ecosystems. The introduction of exotic organisms threatens native biodiversity and could lead to changes in relative abundances of species and individuals that are of ecological and economic importance. This has already been observed in other parts of the world. While EFH has not been directly affected, recent introduction of a brown mussel into the Gulf of Mexico is of concern and is being investigated. It is anticipated that technology such as use of filters or open ocean exchange of bilge waters can be used to reduce the spread of non-native species. Considering the extent of port development and shipping along the South Atlantic, addressing this issue is of paramount importance.

6.1.1.8 Inshore Mining

Inshore mining, as a category of EFH threats, is generally confined to a few specific locations where associated effects may be substantial. Between 1981 and 1996 the NOAA Fisheries Service received only 434 of these proposals for review. Of these, 307 were from Florida and involved phosphate mining. While these activities undoubtedly have a dramatic effect on local landscapes and wetlands, the majority are well inland of most EFH locations. Where these activities occur along the coast, phosphate rock, sand, gravel, stone, and marl are generally mined. Phosphate rock is sought mostly for fertilizer production and the other materials are used mostly for fill, roadbed construction, and concrete production. The products of mining operations may eventually be transported to other locations and construction and operation of shipping facilities and navigation channels could involve EFH.

Threats to EFH from Inshore Mining Activities *Potential threats include conversion of wetlands to mine pits and uplands, or to reclaimed aquatic sites and uplands that lack pre-mine habitat and fishery production values; direct and/or non-point-source discharge of fill, tailings, chemicals, cooling and processing water, and surface and ground waters into streams, rivers and estuaries; hydrological modifications including those associated with ditches, dikes, water and waste lagoons, intake and discharge systems; and cumulative and synergistic effects associated with other mining and non-mining activities. Related shipping, storage, and processing facilities also can threaten EFH.*

Where mining activities occur in areas identified as EFH, the local effect is often dramatic and extremely damaging. In eastern North Carolina phosphate mining has essentially eliminated an entire estuarine creek ecosystem in Beaufort County. The only phosphate mine in North Carolina is found in Beaufort County and located adjacent to the estuarine waters of Pamlico and South Rivers which are tributaries of Pamlico Sound. A 2006 proposal for continuation of mining would result in the loss of about 3,000 acres of wetlands of a variety of types, including the loss of approximately 30 acres of fresh and brackish estuarine emergent wetlands and freshwater/brackish water submerged aquatic vegetation located in the upper reaches of 5 estuarine creeks whose headwaters would be within the proposed mine expansion‘s footprint. Wetlands losses of this magnitude are significant on an ecosystem scale and the extent to which mitigation would offset these losses is uncertain at best. Alternative mining plans are available to the applicant that would be less damaging to wetlands and EFH; however, the company was opposed to these alternatives based on economic issues including profit margin.

In Dade and Monroe Counties, Florida, limestone removal operations have converted large areas of wetlands to open pits. The majority of these operations occur in the ―Lake Belt‖, which is an approximately 57,515-acre area that was established by the Florida Legislature in 1997 for the purpose of implementing the Miami-Dade County Lake Belt Plan. The area lies west of Miami and east of Everglades National Park. To date, mining in the Lake Belt area has thus far converted approximately 4,900 acres of freshwater wetlands into lakes. The Clean Water Act Section 404 permits authorized by the COE require the mining industry to fund acquisition, restoration, and long-term management of lands in the Pennsuco wetlands, which is the area sandwiched between the Lake Belt and the Florida Everglades.

While most state and federal regulations require restoration of mine sites, such action is costly and often fails to produce environments that are similar in ecological character and productivity to those that were destroyed. EFH designation could further fishery management opportunities in certain locations and in the case of certain mining activities. In locations where suitable mitigation cannot be provided, the creation of new mines and expansion of existing operations may be curtailed or prohibited. Other less intrusive mining operations, such as minor removal of sand and gravel, are likely to continue, but needed environmental protection measures (e.g., seasonal work restrictions) could be specified to minimize impacts to fishery resources and prevent significant harm to EFH. However, this is not always the case as illustrated by a proposed 750 acres mineral mining project in New Hanover County, North Carolina that would adversely impact about 300 acres of tidally influenced forested wetlands located adjacent to the northeast Cape Fear River. The wetlands to be impacted and the adjacent waters in the river are designated as fish management areas by the North Carolina Division of Marine Fisheries and are therefore EFH. While approval of wetland losses of this type is unlikely, the frequency of this type of mining activity is likely to increase given the increase in development in coastal states and the need for aggregate fill for highway and commercial construction.

The construction and operation of mining-related facilities such as storage, processing, and shipping facilities and other related infrastructure such as roads, also presents a threat to EFH. Discussions found in Sections 6.1.1.6 and 6.1.1.7 address these factors.

6.1.1.9 Hydrologic Modifications

Alteration of freshwater flows into coastal marine waters, typically via the construction of canals, has changed temperature, salinity, and nutrient regimes, reduced the extent of wetlands, and degraded estuarine and nearshore marine habitats (Reddering 1988; Whitfield and Bruton 1989). The following summary is largely taken from Serafy et al. (1997). Profound changes to the south Florida ecosystem have occurred with the construction of an extensive inland and coastal canal system by the COE which began as early as 1917 (Hoffmeister 1974; Teas et al. 1976). Today, the system constitutes a 1400-mile network of canals, levees, locks and other flood control structures which modulates fresh water flow from Lake Okeechobee, the Everglades, and coastal areas. These areas, which serve as nursery areas for a wide diversity of organisms, have experienced drastic changes in both the amount of freshwater they receive, and in the fashion in which it is delivered. For example, in southern Biscayne Bay, Florida, canal locks are all that separate this occasionally hypersaline lagoon from the entirely freshwater canal systems. When the locks open, the salinity of marine waters downstream often drops 20 ppt within 60 minutes before recovering as rapidly (Wang and Cofer-Shabica 1988). This may occur several times a day and over several months, particularly during the rainy season (i.e., May to October) when water temperatures are also at maximum levels.

Potential Threats to EFH from Hydrologic Modifications

Most hydrologic modifications are performed with other activities that are identified as having potential to adversely impact EFH. As such, the activities involved are similar or identical to those identified in other sections. Other threats are possible with mosquito control, aquaculture, wildlife management, and flood control projects and activities. Hydrologic modification can involve entire watersheds and drainage basins for large scale water diversion projects, where silviculture and/or agriculture activities are large in scale and/or intensity, and where runoff from urban and suburban development is substantial. Threats related to hydrologic modification can involve any activity that alters water quality or the rate, duration, frequency, or volume at which water enters or moves through an aquatic system. Consequently, activities associated with industrial, urban, and suburban development (including those occurring on uplands), ditching, draining, diking, and impounding may all qualify as hydrologic modification related threats.

Rapid salinity fluctuations can represent a significant stress for a marine organism, depending on its osmoregulatory ability and/or its behavioral response (Serafy et al. 1997). In fishes, abrupt salinity changes can cause mineral imbalances in the blood which tends to become diluted as salinity drops, and concentrated as it rises -- either of which can be lethal (Mazeaud et al. 1977). Rectification of proper osmotic balance in response to salinity stress requires energy expenditure, often at the cost of growth, reproduction and/or resistance to other stressors, including high temperature (Moore 1972; Schreck 1990). The combination of high temperatures and low salinity pulses on marine organisms has received only limited attention (Moore 1972; Albertson 1980).

Only one study has examined the combined effects of high temperature and freshwater pulses on subtropical marine fishes of the Western Atlantic. Serafy et al. (1997) combined a field survey of nearshore fishes in Biscayne Bay, Florida, with a series of laboratory-based freshwater pulse experiments. A 13-month trawl project was supplemented with high temperature - low salinity challenge experiments on eight fishes: five species that dominated canal-influenced habitats (*Eucinostomus gula*, *Lagodon rhomboides*, *Haemulon sciurus*, *Opsanus beta*, and *Lucania parva*) and three species that were less common in these areas (*Cynoscion nebulosus*, *Haemulon favolineatum*, and *Cyprinodon variegatus*). Of the five fishes that dominated the nearshore habitats, three exhibited no mortality when subjected to freshwater pulses, while *L. rhomboides* and *L. parva* exhibited 12.5% and 50% mortality rates, respectively. Mortality was 100% for the three species that were less common in habitats influenced by canals. These laboratory and field results support the hypothesis that anthropogenic changes to fresh water delivery regimes can play a partial role in determining the species compositions of nearshore fish assemblages within Biscayne Bay, Florida.

Holland et al. (1996) found that salinity was a major factor in controlling the distribution and abundance of living marine resources in South Carolina estuaries. In watersheds having the greatest areas of roofs, roads, and parking lots it was found that surface water discharges tended to be ―flashier‖ and that recruitment and colonization by benthic fauna in these areas was less predictable than in more stable environments.

Mosquito control activities and associated threats to EFH have become better understood in recent years. Between 1996 and 2006, NOAA Fisheries Service reviewed 203 applications for mosquito control and related activities in the South Atlantic area. Although efforts to alleviate the hydrologic modifications resulting from this activity are underway (27,000 acres of reconnected impoundments in the Indian River Lagoon) much of the area altered by ditching and draining of saltmarsh throughout the east coast has not been addressed. Although tidal water still flows into most of these saltmarsh areas it flows in prescribed dredged channels and does not interact with much of the marsh surface expect through extreme high tide events. Without sheet flow of water across the marsh surface much of the ecological benefit of saltmarsh is underutilized. Some of these areas are receiving hydrological restoration but efforts have been under funded and go largely unrecognized.

6.1.1.10 Dams, Impoundments, and Other Barriers to Fish Passage
Natural river systems throughout the world have been extensively modified for a variety of societal purposes including withdrawals for irrigation, public water supplies, navigation, flood control, and hydroelectric power. Over half of the world‘s large river systems (172 of 292) are affected by dams constructed in the past century (Nilsson et al. 2005). Approximately 800,000 dams have altered riverine habitats worldwide, with approximately 2 major dams constructed each day for the past 50 years (World Commission on Dams 2000). In the United States the total number of dams built during 1700- present is not known with certainty. The National Inventory of Dams (FEMA and U.S. Army Corps of Engineers 1994, 1996) listed approximately 76,000 dams including those deemed to be a threat to life and property downstream, those greater than 6 feet high with more than 50 acre-feet of storage, and those 25 feet or greater in height with more than 15 acre-feet of storage. The National Research Council estimated well over 2.5 million dams existed in the United States in 1992. All of the watersheds tributary to the South Atlantic Shelf Ecosystem are highly affected by large mainstem flood control and hydropower dams and many small dams constructed for various purposes. Bush, et al. (1998) in a review of existing dam location data identified 6,944 dams in South Atlantic watersheds (North Carolina to Florida).

Thousands of wetland acres have been impounded each year in the southeast for purposes such as waterfowl habitat creation, aquaculture, agriculture, flood control, and mosquito control. Historically, large areas of wetlands were impounded in South Carolina for rice production.

Projects range in size from minor, such as repair of existing embankments, to large-scale projects where constructing dikes and water- control structures may affect relatively large wetland tracts.

Numerous dams and other structures have been built on major rivers for industrial water uses, hydropower facilities, reservoirs, and as part of flood control projects. Those facilities near the coast can have an adverse effect by blocking fish passage, and modifying hydrology and sediment and nutrient flows to coastal waters. Dams affect or disrupt many natural processes including upstream and downstream movements of fish and other aquatic species, export of organic carbon, natural hydrological variability and seasonal flow patterns, seasonal temperature, dissolved oxygen and nutrient export patterns, and riverine, estuarine, and coastal geological processes (Freeman et al. 2003; World Commission on Dams 2000).

Potential Threats to EFH from Dams, Impoundments, and Other Barriers to Fish Passage
Direct effects of impoundments and other barriers are removal of habitat, conversion of habitat away from historic usage, alteration of hydrology, and modification of water quality by modification of temperature, salinity, and nutrient and sediment fluxes. Flow regimes often are controlled and differ substantially from pre-impoundment flows. This can adversely affect anadromous fish migration and spawning as well as food production for prey species needed by larvae and juveniles. Riverine, estuarine, and coastal marine ecosystems have evolved in synchrony with natural seasonal river flow variability and discharge patterns. Species life cycles, reproduction, and sustainable populations may be disrupted by man-made barriers and their many effects as described previously.

Large acreages of coastal wetlands have been impounded along the southeast Atlantic. Reasons vary, but include aquaculture, waterfowl production, mosquito control, and in the Old South prior to 1912, rice production. The overall amount of impounded coastal wetlands is not known, but probably exceeds 200,000 acres. Between 1981 and 1996, the NOAA Fisheries Service reviewed 721 proposals of varying sizes that blocked or impounded EFH (Tables 26-29). A review of 190 of these projects revealed that about 7,131 acres of EFH would be adversely altered through these projects. From 1996-2006, the NOAA Fisheries Service Habitat Conservation Division received 465 applications for barriers and impoundments.

A primary biological concern for barriers and impoundments is the impact on estuarine-dependent marine fisheries production. Most impoundments are managed for resources other than fish (e.g., waterfowl). The management regimes, based largely on seasonal consideration, may exclude or severely restrict access by fish and invertebrates. This decreases habitat area and proportionately, the production of fishery resources. Even if fisheries gain access, conditions within impoundments may not be hospitable and organisms may not be able to escape and enter harvestable and reproductively active populations found in surrounding waters. Other management regimes, such as marsh burning, may adversely affect fishery resources. Water quality and nutrient outflow also may be compromised.

However, it is important to note that existing impoundments can be managed to reduce their impacts on estuarine habitat, although some impacts may remain, (e.g., blockage of ingress-egress, reduction of carbon and nutrient export). New impoundments pose a potential risk to EFH and fish production and must be carefully evaluated. However, within the South Atlantic, some positive aspects are evident related to existing impoundments. Because wetlands have been extensively damaged, these areas (especially old rice fields) provide a wealth of available habitat. Further, production of fisheries organisms within these areas is often excellent. Crab production, for example, has been shown to be high in some areas and the production of many estuarine-dependent species has been observed.

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The effects of riverine dams and impoundments on riverine and coastal ecosystem processes, habitats, and health may be profound. Ecological functions of riverine ecosystems affected by dams may be grouped into five primary components: hydrology, biology, geomorphology, water quality, and connectivity (Instream Flow Council 2002). Each of the five components is strongly linked with physical habitat structure, important nutrient and carbon cycles, and health and productivity of estuarine and coastal marine ecosystems. Explained in simplest terms, the effects of dams are manifested through the broad impact categories of habitat fragmentation and flow regulation, in addition to alteration of morphological processes.

With respect to coastal ecosystems and managed fisheries, arguably the most critical effects of dams include blockage and consequent reduction in available reproductive habitat for sea-run diadromous fishes, and large-scale alteration of the distribution and periodicity of freshwater inflows.

Diadromous fishes including shad, herring and other alosines are important components of estuarine and marine food webs. Prior to construction of dams in Atlantic river basins large annual spawning runs of shad and herring and other diadromous species supported important coastal and river fisheries. Early accounts described annual spawning runs of shad and river herring in rivers including the Potomac, Susquehanna, Roanoke, and Savannah in the tens of millions (Baird 1887) with landings in individual river basins exceeding today‘s total Atlantic Coast managed fishery landings by a wide margin. Baird was among the first marine scientists to suggest the relationship between diadromous fish biomass and support for stocks of other commercially important marine species. Construction of dams in Atlantic Coast river basins began soon after European colonization in the early 1700s and continued in cycles through the early 1970s (Watson 1996). Nearly all large river basins in the South Atlantic were closed to significant diadromous fish spawning runs by mainstem dams by the 1960s and 1970s. Busch et al. (1997) estimated the reduction in Atlantic Coast riverine habitats for diadromous species due to construction of dams. In the North Atlantic region (Maine to Connecticut) stream access for diadromous species has been reduced by 91%, and the corresponding reduction for the South Atlantic Region (North Carolina to Florida) is 77%. As dam construction progressed, along with unregulated exploitation and increasing pollution, the Atlantic Coast shad fishery remained one of the most economically important fisheries into the 1940s prior to construction of the last major mainstem dams after the Second World War (Hightower 1997). Today the formerly large spawning runs of shad, river herring, striped bass and sturgeon are reduced to small remnant populations or have disappeared entirely in some rivers. Because of the drastic reductions in abundance of shad and other alosine species, their importance in food web support has also diminished and may represent a significant limiting factor in recovery of some federally managed species.

The timing, duration, and frequency of river flows are critically linked to the health and function of riverine, estuarine and coastal marine ecosystems and fisheries (Taylor et al., 1990). Estuarine and coastal marine wetlands and deepwater habitats are highly dependent upon inputs of freshwater and associated nutrients and sediments from rivers (Berkamp et al. 2000). Seasonal periods of increased river discharge and consequent inflow to estuaries and coastal waters may serve as biological triggers for fish and invertebrate migrations and reproductive cycles. More prominent examples include upstream spawning movements of shad, striped bass, and sturgeon to spawning habitats in river channels; and movements of spawning blueback herring and Atlantic menhaden into floodplain forested wetlands and deepwater sloughs (Rulifson 1982; Pardue 1983; Meador 1982). Natural seasonal patterns and variations in freshwater inflows to estuarine and coastal marine habitats provide suitable salinity and nutrient conditions for reproduction and growth of oysters, blue crabs, shrimp and many estuarine-dependent species. Regulation of river flows by dams, particularly for flood control and hydropower production, may significantly alter natural patterns of river discharge to which many species life cycles have adapted during their evolutionary history. River regulation may affect seasonal salinity patterns over large areas of estuarine and coastal marine habitats. Dams with large storage capacity can reduce downstream flows during critical late winter and spring diadromous fish migrations, resulting in reduced water level and duration and areal extent of inundation, severely limiting fish production.

Dams and reservoirs trap river-borne sediments, resulting in reduction of nutrient-rich sediment deposition in downstream floodplain wetlands and alluvial deltas. Resulting disruption of alluvial delta and wetland formation processes may cause large scale floodplain and wetland subsidence, adversely affecting habitat stability and productivity for estuarine and coastal marine fisheries.

Thermal stratification of large reservoirs during summer months often results in biological oxygen depletion of the cooler water of the hypolimnion, with consequent discharge of cooler water with low dissolved oxygen downstream of the dam. Fish and other aquatic life may be eliminated or adversely affected in riverine or estuarine areas downstream as far as the deoxygenation persists. Large, shallow impoundments lacking thermal stratification may result in solar warming with consequent release of water with elevated temperatures to downstream riverine and estuarine habitats. During warmer summer months, the resulting elevated water temperatures may exceed the tolerance levels for fish species adapted to naturally occurring seasonal temperature regimes.

Where dams and river regulation have been in place for many years, the continuing cumulative effects of habitat fragmentation, altered flows, water temperatures, and dissolved oxygen conditions may result in shifts in aquatic species community structure and composition. Populations of federally managed diadromous, estuarine and marine species may be limited by the continuing effects of dams and river regulation.

Dams and other barriers have been constructed on almost every major southeastern river. They serve multiple purposes including hydropower production, water supply, and flood attenuation. Dams located on the Roanoke and Neuse Rivers in North Carolina, the Cooper and Santee Rivers in South Carolina and on the Savannah River on the South Carolina-Georgia border are major impediments to anadromous fish migrations, as mentioned above. Most of these structures are old and were built either before their effects on fish and other wildlife were known, or at a time when environmental concerns were of lesser importance than economic and political factors. Considering the present level of knowledge of their effect on fish migration and production, water quality, and flow alteration, it is unlikely that major new structures will be built. The present challenge is to revisit older structures to determine their usefulness and where their negative impacts outweigh their benefits, they should be removed or modified. An example is removal of the Quaker Neck Dam on the Neuse River in North Carolina. Where removal is not feasible then consideration must be given to providing for, or improving fish passage and for modifying flow regimes to mimic pre-impoundment flows. These considerations will rely on new research and improvements in fish passage technologies.

6.1.1.11 Other sources of nonpoint-source pollution

Potential Threats to EFH from Other Sources of Nonpoint Source Pollution

Potential threats include reduced water quality, erosion, increased contaminants, increased sedimentation, and disease.

The more common sources of NPS pollution include runoff from agriculture, pasture lands, silviculture, mining, and developed areas as well as erosion created from modifying rivers, streams, and shorelines. These sources have separate sections in the Fishery Ecosystem Plan. Three additional sources of NPS runoff deserve brief mention and include construction sites, marinas, and septic systems

Runoff from construction sites can be considerable sources of NPS pollution (Carpenter et al. 1998). Construction sites occupy a relatively small percentage of land surface area, but rates of erosion from these sites can be high leading to a large amount of pollution coming from these small areas. Erosion rates from watersheds under development can approach 50 times the rate from agriculture lands and 500 times the rate from areas with undisturbed plant cover. Eroded material from construction sites contributes to siltation of water bodies as well as eutrophication. Best management practices for controlling runoff from construction sites are well known and should be followed to avoid impacting fishery resources.

Understanding NPS pollution associated with marinas can be difficult because marinas can be both a source of pollutants generated by activities occurring within the marina as well as the place where pollutants generated elsewhere collect (Flory 2005; USEPA 2001). Construction of the basins, docks, jetties, and bulkheads needed for marina operations typically reduce water circulation, and this reduced circulation promotes the settling of fine sediments that often have organic material, metals, or other pollutants attached to them. These materials concentrate in marina sediments and, at times, also can concentrate in marina waters. The pollutants that might be generated at a marina or accumulate within a marina basin include nutrients and pathogens (from pet waste and overboard sewage discharge), sediments (from parking lot runoff and shoreline erosion), fish waste (from dockside fish cleaning), petroleum hydrocarbons (from fuel and oil drippings and spills and from solvents), toxic metals (from antifouling agents and debris from boat maintenance), and liquid and solid wastes (from engine and hull maintenance and general marina activities).

Many contaminants generated from boat maintenance and general marina use (e.g., oil and grease drippings from cars) are insoluble in water. In the slow flowing, protected waters of the marina, the fine particles that these materials adhere to settle and accumulate in the sediments. While these sediments may then release their contaminants into the water in response to physical disturbance (such as dredging, propeller wash, or storms) or from changes in water chemistry (such as pH or dissolved oxygen concentration), effects upon benthic organisms and fishery resources are of greatest concern. Most benthic organisms either burrow into the sediment or feed by sorting through large volumes of sediment in search of prey items or detritus. Both behaviors bring benthic organisms into close contract with any contaminants that may be present and these contaminants can then accumulate in the bodies of the benthic animals. Fishery species that feed upon these benthic organisms are then exposed to concentrated doses of the contaminants, which may reduce the health or reproduction of the fishery individuals or make them unsuitable for consumption by humans.

Pollutants from marinas can cause pollution problems in the water column. These problems usually take the form of decreased levels of dissolved oxygen and increased levels of metals and petroleum hydrocarbons. Pollutants that cause these problems get into the water through storm water runoff, discharges from boats, and spills of fuel or bilge water. Low levels of dissolved oxygen can be a problem any place where organic material accumulates. The decay of organic material consumes oxygen from surrounding water. If the low circulation promotes accumulation of organic material while at the same time hindering exchange with oxygen-laden waters outside the marina, the result can be insufficient oxygen for fishery species.

In addition to pollutants that reduce the quality of sediment or the water, marinas often are associated with silt that can impair seagrass, oyster, or other habitats that support fishery resources. Increased boat traffic within and near a marina can erode shoals and the shoreline suspending large amounts of sediment into the water that fall upon fishery habitats. Waves generated by boat wakes can wash away seagrass that is loosely rooted in sediments and the benthic organism living at the sediment surface.

NPS pollution associated with marinas can be reduced by ensuring marinas are designed to flush regularly with adjacent waters; locating marinas close to tidal inlets and away from the headwaters of tidal creeks is part of these design decisions. Shorelines should be vegetated to reduce erosion. Stormwater runoff can be controlled by well designed and maintained stormwater management systems. Marina fueling and sewage collection stations should be maintained and designed to make cleanup of spills easier.

Septic systems include the underground system of pipes and tanks designed to use naturally occurring bacteria and microorganisms to treat bathroom, kitchen and laundry wastewater. In older homes, a septic system may be little more than a cesspool and a pipe that connects the cesspool to the house. In newer homes, a septic system usually includes a septic tank, distribution box, drain field, and pipes that connect these elements. Passing sewage and household wastewater through a septic system protects the environment from contamination. Microorganisms and insects living within the drain field help decontaminate waste materials by consuming leftover waste particles. Improperly maintained septic systems can allow nutrients and pathogens to enter ground waters and surface waters that flow into coastal ecosystems. The excess nutrients can lead to eutrophication and low levels of dissolved oxygen, both of which can impair habitats used by fishery species. The pathogens can spread disease that reduce the health of fishery species.

NPS pollution from septic systems can be reduced by ensuring the systems are inspected annually and pumped regularly. Pumping out every three to five years is recommended for a three-bedroom house with a 1,000-gallon tank; smaller tanks should be pumped more often. Storm drains should not be diverted into septic systems because the extra load on the systems will overwhelm its ability to process nutrients and eliminate pathogens. Any measure that decreases water use within a home can help a septic system protect coastal water quality by reducing the likelihood of overflow from the system.

6.1.1.12 Non-native or nuisance species

Update on Aquatic Invasive Species Management in the Southeast-March 2008
Marilyn Barrett-O'Leary Southeast Aquatic Resources Partnership (SARP)
Aquatic invasive species are a part of fisheries and wildlife management in all of the Southeast Aquatic Resources Partnership (SARP) states. Many of the states manage specific species cooperatively, but we do not have comprehensive regional management. For example, Texas and Louisiana partnered with some federal agencies to bring massive chemical control to reduce a giant salvinia infestation on Caddo Lake, a popular angling lake on the two states' shared border. Florida, a state with better funding resources than many of its neighbors, routinely shares research results and outreach products (on many invasive plants and animals) to promote regional control. All of the states are members of at least one regional Aquatic Invasive Species (ANS) panel, providing biannual meetings to share information and committees to work on problems regionally.
Every SARP state has developed an ANS management plan. Most have completed that process, which involves forming a task force, gathering information, identifying overlapping jurisdictions, setting priorities, and devising action plans. Most important, these activities lead to governor's buy in and signature, interagency agreements such as MOUs, and continuation of the task force in some form to facilitate management. As of this date, every SARP state has at least one agency person with ANS as part of his/her scope of work. Some have individuals with ANS as his/her exclusive scope of work. The states of Louisiana, Texas, Florida, Virginia and Missouri have officially accepted plans. Kentucky, South Carolina, Alabama, Tennessee and Mississippi are in the final, official stages of seeking national acceptance of their plans. They have effectively identified the problem and are already integrating solutions into their agency activities. Oklahoma, Georgia, North Carolina, and Arkansas are still developing their plans.

Both Georgia and North Carolina are developing plans that combine management of terrestrial and aquatic invasive species. All states are aware of the need to work in that direction. All of these states face similar issues. Below are a few of them:

1. Invasive species are not all bad or all good – they may cause problems in certain circumstances but actually benefit certain groups or situations. Management (treatment, regulation, education) requires ecological and economic evaluation on local, regional, and national levels and cooperation among state, local, and federal agencies.

2. Invasive species almost always alter the ecosystem; they seldom simply slip into an unfilled niche. They thrive in disturbed systems. Therefore, ecological management can contribute to invasive species prevention and control. Unfortunately, states are not funded or equipped to manage all state waters at that level, and every state has many water bodies that are managed privately or by federal agencies.

3. The general population has only fleeting knowledge of this problem, and often, unwittingly, contributes to it. Consistent, continuous education is needed over the long term. SARP agencies are trying to educate one of the most involved segments of the population – the recreational fisher – to clean off boats before leaving the dock, place unused live bait into the trash rather than dumping it into the water, and to refrain from moving live fishes in an attempt to ‗stock' for certain fish. Similar, targeted education efforts need to be made towards many other population segments. Tax dollars need to be earmarked for this management.

6.1.2 Marine/offshore processes

6.1.2.1 Navigation

Offshore maintenance dredging for navigation is mainly limited to inlet bar channels and other port entrances; (e.g., Port Canaveral, Florida). The sediments are typically coarse and the bottom communities are low diversity reflecting the dynamic nature of these areas. Bottom organisms occupying this zone are generally sparse and adapted to the dynamic nature of the habitat they occupy. As such, dredging in these locations generally does not pose the same magnitude or type of impact incurred when working in nearshore environments. The same is true for vessel operations, although to some degree the problems discussed in Section 4.1.1.6 also apply. Vessel operation impacts are mainly linked to sinking, grounding, routine disposal trash and wastes, and the accidental release or spillage of cargo and fuel.

However, offshore new dredging, namely widening and deepening existing port entrance channels to accommodate super-carriers, i.e., Post-Panamax vessels an impact complex hard bottom communities along channel walls in addition to reef trends. For example, the Jacksonville District COE in conjunction with Port Everglades is presently completing a feasibility study in part to evaluate the widening and deepening of the Port Everglades Outer Entrance Channel. The project could impact offshore marine habitats, including hard bottom and coral reef communities located offshore Fort Lauderdale, Florida (Broward County). In total, 11.9 acres of hard bottom habitat on the outer reef (Reef 3) may be removed during construction (COE 2006).

Potential Threats to EFH from Navigation

Potential threats include excavation and burial of EFH in connection with creation, expansion and maintenance of navigation channels; elevation of turbidity and resuspension of toxic and *harmful components of dredged materials (includes material that cause elevated sediment and dissolved oxygen demands); interruption of coastal sand movement and sub-adult fish migration through construction of channel stabilization structures such as jetties; potentially harmful vessel operations such as discharge or spillage of fuel, oil, grease, paints, solvents, trash, and cargo; grounding/sinking/prop scaring in ecologically/environmentally sensitive locations; exacerbation of shoreline erosion due to wakes; and transfer and introduction of exotic and harmful organisms through ballast water discharge.*

With a few exceptions, offshore dredging is performed using hopper dredges. Hopper dredges generally dump accumulated material through a split hull; however, the use of these dredges in connection with pipelines and vessel pump out is becoming more commonplace, especially where sand is needed for beach fill. Closer inshore, sidecast dredges may be used where wave amplitude is slight and dredging volumes are relatively minor. In protected waters pipeline dredges are almost always used since they provide the most effective and efficient means for removing and redepositing bottom sediments. On rare occasion, as in the case of the Cape Canaveral Ship Channel, pipeline dredges may be used in open waters but their vulnerability to wave damage generally precludes this. Bucket dredges and scows are employed in some locations, but such use is usually limited to situations where other dredges cannot operate due to water depth and pumping distances (for pipeline dredges).

In connection with offshore waters, threats to EFH are most significant in terms of possible burial of benthic communities in the vicinity of dump sites and in connection with turbidity from dumped materials. Contamination of the water column and bottoms is also possible if the dredged material is contaminated. Sediments may also be re-dispersed after being dumped in offshore sites and burial of productive bottoms is possible. On occasion, designated dump sites are not adequately studied or they change and high quality benthic habitat may be damaged or destroyed.

Although most ports are located in estuarine waters, navigation related threats can also be severe in offshore waters. As the shipping industry moves towards super containerships, the many eastern seaboard ports are evaluating the need to widen and deepen offshore entrance channels. Currently, only a limited number of ports can accommodate Post-Panamax vessels. The Port of New York/New Jersey is the only port along the Atlantic seaboard that is undergoing expansion work to support super-carriers.

Additional threats to EFH from offshore navigation occur through the overboard disposal of trash, cargo, and wastewater from ocean going vessels, and disposal of dredged material (see Section 7.4.2.1). Although comparisons are unavailable, it is likely that most vessel-related disposal occurs on the open ocean, rather than in estuarine and nearshore waters where such activities are likely to be observed.

Within Florida waters, particularly in the Florida Keys and Fort Lauderdale, vessel groundings represent a chronic threat to live coral habitat. Anchoring is also a problem, however, it has become less of a threat through wide spread use of single point mooring buoy systems. Vessel groundings can be broken into two broad categories: large vessel and ship groundings that often result in severe injury to live coral colonies and non-living reef framework; and small recreational boat groundings that result in numerous strikes to individual coral colonies in both inshore and offshore areas. Large vessel and ship groundings occur infrequently, but result in far more significant injury to coral reefs and other habitat types. Recreational boat groundings are much more frequent. Between 1993 and 1997, 2089 groundings were reported in the Florida Keys National Marine Sanctuary. Many more are likely unreported.

Table 6.1-1 reported Vessel Groundings in Florida Keys National Marine Sanctuary (FKNM) 1993-1997.

Accurate baseline data for live coral coverage exist mainly for reefs in the Florida Keys but not for the remaining habitat that contains stony corals that do not form reefs. In some cases though, sufficient data are available to allow calculation of the actual extent of a grounding incident. For example, on August 10, 1994, the R/V *Columbus Iselin*, a 154-foot research vessel, was conducting survey work for the University of Miami when it struck Looe Key, a spur and groove reef. Approximately 345 square meters of living coral and 338 square meters of non-living coral reef framework were destroyed.

Injuries to coral from groundings take several forms and include crushing, splitting and fragmentation, dislodging colonies, and depending on the severity of the incident, sedimentation and/or burial. In general, groundings occur on or near the reef crest where coral formations are closest to the water surface. Species commonly injured in the reef crest include elkhorn coral (*Acropora palmata*), staghorn coral (*A. cervicornis*), fire coral (*Millepora complanata*), starlet coral (*Siderastrea siderea*), mustard hill coral (*Porites astreoides*), and knobby zoanthidean (*Palythoa mammillosa*). Species that inhabit deeper Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 46 areas such as brain coral (*Diploria strigosa*), star coral (*Montastrea annularis*), and large star coral (*Montastrea cavernosa*) are at risk from deep draft vessels. Small individual groundings may recover over time, but the loss of live coral coverage is likely to take decades. Catastrophic groundings involving large ships or freighters may never fully recover.

Since 1994, there have been at least 10 reported large-scale groundings near the existing anchorage off Port Everglades (in Florida) that have collectively damaged over 3 acres of coral reef habitat. The existing shallow water anchorage is located between two lines of reef. Dozens of undocumented anchor and anchor chain drag impacts have also occurred damaging an undetermined amount of reef. The U.S. Coast Guard has proposed anchorage rulemaking to revise the existing anchorage locations to strengthen existing anchoring requirements and guidelines in order to provide a higher degree of protection to the reef resources.

6.1.2.2 Dumping

Dredged material disposal in ocean waters generally involves disposal of sediments dredged from inshore areas such as port facilities. Where navigation approaches from offshore and inlets are involved these materials may also be placed in offshore sites. Most of the sediments taken from inshore areas are fine, contain some degree of contamination, and produce at least short-term impacts such as turbidity plumes when removed or deposited. The overall effects of dumping on or near EFH can range from immeasurable to significant and are not well studied. Therefore, dredging and disposal are typically evaluated on a case-by-case basis. The SAFMC policy on dumping provides additional detail on the subject. The principal authority for designating ocean disposal sites for placement of dredged material is the Regional Administrator of the EPA. The EPA develops and publishes Environmental Impact Statements (EIS) and the rule making paperwork for ocean dredged material disposal site (ODMDS) designations. Corps of Engineer Districts provides the EPA with the necessary information to prepare the EIS and to identify significant issues to be addressed in the site designation process. Information required from the Districts includes: zone or siting feasibility data, justification for the need for ocean disposal, and alternatives to ocean disposal. The purpose of the EPA site designation process is to establish sites that minimize impacts to the environment, economize disposal site management and monitoring activities, and support multiple users (C. McArthur personal communication). Under provisions of the Marine Protection Research and Sanctuaries Act (MPRSA), ocean disposal of hazardous and toxic materials, other than dredged materials, is prohibited by U.S. flag vessels and by all vessels operating in the U.S. territorial sea and contiguous zone. The EPA may issue emergency permits for industrial waste dumping into ocean waters if an unacceptable human health risk exists and no other alternative is feasible. The MPRSA assigns responsibility the ocean disposal of dredged material to the EPA and the COE. This involves designating ocean sites for disposal of dredged material; issuing permits for the transportation and disposal of the dredged material; regulating times, rates, and methods of disposal and the quantity and type of dredged Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 47 material that may be dumped; developing and implementing effective monitoring programs for the sites; and evaluating the effect of dredged material disposed at the sites (C. McArthur, personal communication).

To date, offshore ocean dumping sites have been approved for ports at Wilmington, North Carolina; Brunswick and Savannah, Georgia; Georgetown, Charleston and Port Royal, South Carolina; and Miami, Palm Beach, Port Everglades, Fort Pierce, Jacksonville, and Fernandina Beach, Florida (C. McArthur, personal communication). The COE has identified Jacksonville Harbor as possibly needing a new or expanded ODMDS.

Table 6.1-2 Region IV of the U.S. Environmental Protection Agency identifies the following concerns in connection with existing South Atlantic Ocean Dredged Material Disposal Sites (ODMDS): Ocean Dredged Material Disposal Site	Site Specific Concerns
Charleston, SC ODMDS	Live bottom areas proximal to the site subject to possible impact.
Miami, FL ODMDS	Effect of disposal plumes on nearshore coral reefs are under investigation.
Port Everglades, FL ODMDS Palm Beach, FL ODMDS	Burial of deepwater hard bottoms and shelf edge zones that support managed species.
Fort Pierce, FL ODMDS	Conversion of sediment type could affect tilefish burrows.
	Possible presence of deepwater corals (e.g. *Oculina varicosa*).
	Burial of deepwater hard bottoms and shelf edge zones that support managed species.
	Conversion of sediment type could affect tilefish burrows.
	Possible presence of deepwater corals (e.g. *Oculina varicosa*).
	Offsite transport of disposed dredged material and subsequent burial of nearby hard bottom communities is of concern to local community.
Jacksonville, FL ODMDS	Lies within Northern Right Whale Critical Habitat and site may be undersized.
Fernandina, FL ODMDS	Lies within Northern Right Whale Critical Habitat.
Brunswick, GA ODMDS	Lies within Northern Right Whale Critical Habitat.
Wilmington, NC ODMDS	Wood debris in dredged material suspected of

Dumping of trash, wastewater, and unwanted cargo is more likely to occur on the open seas since it is less observable here than in inshore waters. Prior to passage of the Marine Plastic Pollution Research and Control Act (MPPRCA) of 1987 (PL 100-220), an estimated 14 billion lbs of garbage were being dumped into the ocean each year. More than 85% was believed to have come from the world's shipping fleet in the form of cargo-related wastes. See section 6.1.2.2.3 below.

Potential Threats to EFH from Dumping

Potential threats include burial of habitats and their flora and fauna, introduction of contaminants and toxic substances into waters and substrates, increased and harmful turbidity levels, and creation of hazards to fishing and navigation.

Threats associated with ocean dumping sites include covering of live bottom or hard bottom areas in or near a dump site; disposal of fish processing wastes; converting the sediment type in areas that support tilefish; impacts to nearshore coral reefs and live bottoms by disposal plumes; offsite transport of disposed dredged material and subsequent burial of nearby hard bottom communities; designated sites that are too small to handle the load; migration of debris (e.g., wood) to fishing grounds; derelict vessel disposal; and the location of dumping sites within critical habitat of endangered species such as the northern right whale.

Because monitoring of disposal activities is sometimes inadequate, there are reports of dredged material dumping outside of designated dump sites (short dumping). One recent example of a possible short dumping event involves the excavation associated with the Fort Pierce Harbor, Florida, expansion project. In this case, over 400,000 cubic yards of dredged material from this project was dumped at a mid-shelf site. Numerous complaints arose thereafter from fisherman and divers that the fill was short-dumped and large areas of reef habitat had been covered. These sites had previously served as productive snapper/grouper fishing locations. EPA Region IV undertook a number of studies into this issue. EPA monitoring reports are available at http://www.epa.gov/region4/water/oceans/sites.htm#ftpierce. Reed (1996) summarizes information available at the time regarding the mud deposits potentially derived from this event.

Another documented example of dumping occurring outside the designated ODMDS occurred during the Charleston Harbor Deepening Project. A total of 53 documented incidents of unauthorized disposal activity outside the ODMDS were reported subsequent to dredging for the Charleston Harbor Deepening Project. The unauthorized dumps were first detected during a routine assessment of the ODMDS and surrounding area using side scan sonar (Jutte et al. 2001). The documented dumps placed large quantities of mud and clay on sandy bottom habitat, with some located very near hard bottom reef habitat. Subsequent surveys over a four year period to determine whether movement of material from these sites or the ODMDS was having an adverse impact on nearby reef habitats did not identify clear loss of habitat with the exception of one site located closest to the ODMDS. The abundance finfish and large sessile invertebrates, such as sponges and corals also did not appear to be adversely affected during the survey period (Crowe et al. 2006).

In areas that have been suspect of short-dumping, such as the ODMDS located offshore the Port of Miami, the EPA Region IV and NOAA Fisheries Service habitat office have developed additional permit conditions that include:

1. The permittee shall use an electronic positioning system to navigate to and from the ODMDS;
2. The permittee shall certify the accuracy of the electronic positioning system proposed for use during disposal operations at the ODMDS;

3. The permittee shall not allow any water or dredged material placed in a hopper dredge or disposal barge or scow to flow over the sides or leak from such vessels during transportation to the ODMDS;
4. A disposal operations inspector and/or the captain of any tug boat, hopper dredge, or other vessel used to transport dredged material to ODMDS shall ensure compliance with disposal operation conditions defined in this permit;
5. If the disposal operations inspector or the captain detects a violation, he or she shall immediately report the violation to the relevant county Seaport Department, the Corps of Engineers District, and to NOAA Fisheries Service;
6. When dredged material is disposed, no portion of the hopper dredge or disposal barge or scow shall be farther than 500 feet of the center of the ODMDS;
7. The permittee shall use an automated disposal verification system that will continuously track (1 minute intervals) the horizontal location and draft condition of the disposal vessel (hopper dredge or disposal barge or scow) to and from the ODMDS;
8. The required digitally recorded data should include: date, time, vessel name, dump number, beginning and ending coordinates of the dredging area for each load, location at points of initiation and completion of disposal, description of material disposed (rock rubble, sand, clay or silt), volume of load, and disposal technique;
9. The permittee shall conduct a bathymetric survey of the ODMDS within 30 days following project completion;
10. The number and length of the survey transects shall be sufficient to encompass the ODMDS and a 0.25 nautical mile wide area around the site. The transects shall be spaced at 500_foot intervals or less;
11. Vertical accuracy of the survey shall be ±0.5 feet; and
12. At the dredge site, barges must be either lashed to dredges or cables must be floated to avoid impact to submerged resources.

Similarly, at the Charleston ODMDS site a number of constraints similar to those used in Miami were adopted, and it also included limiting the barge traffic to areas that were outside know hard bottom habitat.

Even with the use of approved practices and disposal sites, ocean disposal of dredged materials is expected to cause environmental harm since contaminants will continue to be released, productive bottoms will still be buried, and localized turbidity plumes and reduced oxygen zones will persist. Further, analyses are needed for use in dump site designation. For example, there have already been observed cases (e.g., at Charleston) where dump sites were designated and then, after dumping had been initiated, it was determined that valuable hard bottom habitats were located in or near the dump site. However, at the Charleston Harbor site, while it was determined that valuable hard bottom habitat is located adjacent to the dump site, monitoring has confirmed that construction of a berm along the edges of the disposal site is containing the majority of the dredged material, with the exception of occasional missed targeting and these are generally in the vicinity of the adjacent channel from which the vessel is traversing.

The effects of new disposal techniques such as creation of nearshore berms and ―beneficial uses‖ of dredged material such as creation of shallow water habitats and emergent wetlands are, in many cases, unclear and may cause long-term geomorphological and ecological change that is harmful to certain species and environments. In the Charleston ODMDS, the deepening project included the construction of large berms along the border of the ODMDS that were composed primarily of cooper-marl material that would stay in place. The logic for constructing these berms was to inhibit significant movement of the disposed material within the ODMDS to sensitive bottom habitats located nearby. This effort appeared to be successful based on subsequent monitoring activities (Crowe et al. 2006). The SAFMC recognizes offshore berm construction as a disposal activity. As such, its policies regarding disposal of dredged materials apply. The SAFMC also recommends that research should be conducted to quantify larval fish and crustacean transport and use of inlets prior to any consideration of placement of underwater berms. Until the impacts of berm creation in inlet areas on larval fish and crustacean transport are determined, the SAFMC recommends that disposal activities should be confined to an approved ODMDS. The SAFMC further believes that new offshore and near shore underwater berm creation activities should be reviewed under the most rigorous criteria and on a case-by-case basis.

In the absence of MPRSA and MPPRCA repeal or weakening, major dumping threats to EFH within federal waters should be limited mostly to illegal dumping and accidental disposal of material in unapproved locations. However, many agencies lack sufficient staff and funds to carry out mandated responsibilities and the opportunity for illegal and accidental dumping may be substantial. The effect of insufficient monitoring and enforcement is evident by the tons of debris, sometimes including hazardous materials such as syringes and medical wastes that are deposited along the nation's beaches every year.

As noted in Section 7.4.2.1 the SAFMC has developed Policies for disposal of dredged material in waters under its jurisdiction. With regard to use of ODMDSs, the policy provides that:
The ODMDS should be designated or re-designated so as to avoid the loss of live or hardbottom habitat and minimize impacts to all living marine resources.

Notwithstanding the fluid nature of the marine environment, all impacts from the disposal activities should be contained within the designated perimeter of the ODMDS.
The final designation of the ODMDS should be contingent upon the development of suitable management plans and a demonstrated ability to implement and enforce that plan.
The Council encourages EPA to press for the implementation of such management plans for all designated ODMDSs.
All activities within the ODMDS are required to be consistent with the approved management plan for the site. The Council's Habitat and Environmental Protection Advisory Panel when requested by the Council will review such management plans and forward comment to the Council. The Council may review the plans and recommendations received from the advisory sub-panel and comment to the appropriate agency.
ODMDS management plans should specify those entities/ agencies which may use the ODMDS, such as port authorities, the U.S. Navy, the Corps of Engineers, etc. Other potential users of the ODMDS should be acknowledged and the feasibility of their using the ODMDS site should be assessed in the management plan.
Feasibility studies of dredge disposal options should acknowledge and incorporate the ODMDS in the larger analysis of dredge disposal sites within an entire basin or project. For example, Corps of Engineers' analyses of existing and potential dredge disposal sites for harbor

maintenance projects should incorporate the ODMDS as part of the overall analysis of dredge disposal sites.

6.1.2.3 Marine Debris

One of the more conspicuous byproducts of commercial and recreational boating activities in coastal environments is the discharge of marine debris, trash, and organic wastes into coastal waters, beaches, intertidal flats, and vegetated wetlands. The debris ranges in size from microscopic plastic particles (Carpenter et al. 1972), to mile-long pieces of drift net, discarded plastic bottles, bags, aluminum cans, etc. In laboratory studies, Hoss and Settle (1990) demonstrated that larvae of estuarine-dependent fishes including Atlantic menhaden, spot, mullet, pinfish, and flounder consume polystyrene microspheres. Investigations have also found plastic debris in the guts of adult tuna, striped bass, and dolphin (Manooch 1973; Manooch and Mason 1983). Based on the review of scientific literature on the ingestion of plastics by marine fish, Hoss and Settle (1990) conclude that the problem is pervasive. Most media attention given to marine debris and sea life has focused on threatened and endangered marine mammals and turtles, and on birds. In these cases, the animals become entangled in netting or fishing line, or ingest plastic bags or other materials. Recently, a 35-foot- long sperm whale stranded and died in North Carolina due to ingestion of a plastic float, plastic jugs, a large piece of rubber, 50 feet of nylon rope, and a large plastic bag (D. Engel, personal communication). The production of plastic resin in the U.S. increased from 6.3 billion lbs in 1960 to 47.9 billion lbs in 1985. The increased production, utilization, and subsequent disposal of petro-chemical compounds known as plastics has created a serious problem of persistent marine debris. Marine ecosystems have, over the years, become the final resting place for a variety of plastics originating from many ocean and land-based sources including the petroleum industry, plastic manufacturing and processing activities, sewage disposal, and littering by the general public and government entities (commercial fishing industry, merchant shipping vessels, the U.S. Navy, passenger ships, and recreational vessels) (Department of Commerce 1988c).

Effective January 1, 1989, the disposal of plastic into the ocean is regulated under the Plastic Pollution Research and Control Act of 1987, implementing MARPOL Annex V. Recognizing worldwide concern for preservation of our oceanic ecosystems, the Act prohibits all vessels, including commercial and recreational fishing vessels, from discharging plastics in U.S. waters and severely limits the discharge of other types of refuse at sea. This legislation also requires ports and terminals receiving these vessels to provide adequate facilities for in-port disposal of non-degradable refuse, as defined in the Act.

The utilization of plastics to replace many items previously made of natural materials in commercial fishing operations has increased dramatically. The unanticipated secondary impact of this widespread use of plastics is the creation of persistent marine debris. Commercial fishing vessels have historically contributed plastics to the marine environment through the common practice of dumping garbage at sea before returning to port and the discarding of spent gear such as lines, traps, nets, buoys, floats, and ropes. Two types of nets are routinely lost or discarded drift gill nets and trawl nets (Department of Commerce 1988c). These nets are durable and may entangle marine mammals and endangered species as they continue to fish or when lost or discarded.

An estimated 16 million recreational boaters utilize the coastal waters of the United States (Department of Commerce 1988c). Disposal of spent fishing gear (e.g. monofilament fishing line), plastic bags, tampon applicators, six pack yokes, styrofoam coolers, cups and beverage containers, etc. is a significant source of plastic entering the marine environment.

In the mid 1970s, the National Academy of Science (NAS) estimated that approximately 14 billion lbs of garbage was disposed of annually into the world‘s oceans. Approximately 85% of total trash is produced from merchant vessels, with 0.7% of that total, or eight million lbs annually being plastic. The use of plastics has risen dramatically since the NAS study. In 1987, 20% of all food packaging was plastic and by the year 2000 this figure was expected to rise to 40% (CEE 1987).

The main contribution of plastic to the marine environment from cruise ships is the disposal of domestic garbage at sea. Ships operating today carry between 200 and 1,000 passengers and dispose of approximately 62 million lbs of garbage annually, of which a portion is plastics (CEE 1987). The U.S. Navy operates approximately 600 vessels worldwide, carrying about 285,000 personnel and discharging nearly four tons of plastic refuse into the ocean daily (Department of Commerce 1988a). The U.S. Coast Guard and NOAA operate 226 vessels which carry nearly 9,000 personnel annually and have internal operating orders prohibiting the disposal of plastic at sea. MARPOL Annex V does not apply to public vessels although the Plastic Pollution Research Control Act of 1987 requires all Federal agencies to come into compliance by 1994 (CEE 1987).

6.1.2.4 Offshore Sand and Mineral Mining and Beach Fill

To date, offshore mining for minerals has not been a significant issue in the South Atlantic region (oil and gas mining is discussed separately). However, several pending proposals are under regulatory consideration. Earlier consideration of mining for manganese nodules and removal of useable materials and metals from seawater have not materialized, probably due to market conditions. Recent discovery of large phosphate deposits in waters off North Carolina could eventually lead to requests to mine these deposits. As readily available upland sources of minerals and other materials are depleted, the extraction of marine deposits will become more feasible and likely to occur.

The mining of sand for beach nourishment presents a large, complex, and politically charged threat to EFH in the southeast. Between 1981 and 1996, the NOAA Fisheries Service reviewed more than 200 dredge proposals to nourish beaches. Between 1996 and 2006, NOAA Fisheries Service reviewed an additional 312 dredge proposals to nourish beaches. Most of these projects are large in scope and affect miles of coastline and nearshore habitats. Where sand is removed from nearshore environments, channels, and inlets, additional EFH alteration is possible due to a number of factors such as down drift erosion and removal of materials that eventually nourish shallow waters located behind barrier islands. A survey of 120 of the more than 200 beach nourishment projects received by the NOAA Fisheries Service showed that about 5,735 acres of aquatic sites were subject to excavation and filling.

The Federal Outer Continental Shelf (OCS) contains large sand deposits that MMS anticipates could serve as long-term sources of borrow material for beach nourishment projects. In the last few years, the potential for exploitation of these resources has rapidly grown with identification of suitable sand resource areas in some OCS regions. At the same time, the demand for high quality sand suitable for beach nourishment, coastal protection, and other public and private projects is anticipated to increase during coming years (Hammer et al. 2004). However, the SAFMC is concerned that excavation of the offshore shoals could have significant adverse consequences to the shoreline and living marine resources.

Potential Threats to EFH from Offshore Sand and Mineral Mining

Potential threats include: removal of substrates that provide habitat for fish and invertebrates; creation (or conversion) of habitats to less productive or uninhabitable sites such as anoxic holes or silt bottom; burial of productive habitats in the vicinity of the mine site or in nearshore disposal sites (as in beach nourishment); release of harmful or toxic materials either with actual mining, or in connection with machinery and materials used for mining; creation of harmful turbidity levels; and modification of hydrologic conditions that cause erosion of desirable habitats.

Offshore mining of sand for beach nourishment has steadily increased along the South Atlantic coast. Presently, sand mining and beach nourishment activities are performed along the entire South Atlantic coast from North Carolina to Florida. Major projects include those at Wrightsville Beach, North Carolina; Myrtle Beach and Folly Beach in South Carolina; and many of Florida‘s beaches such as Palm Beach, Boca Raton, Fort Lauderdale and Miami Beach. Large-scale beach nourishment has also been performed at Tybee Island in Georgia; however, the material for that project was obtained from the Savannah Harbor deepening project. In addition to the larger projects that can involve millions of cubic yards of material, a substantial number of smaller projects involving beach scraping and removal of nearshore and inlet sand deposits are performed annually. While most of the larger projects are publicly funded and performed by the COE, many of these smaller projects are paid for with local revenues and/or private funds.

Although some of the environmental effects of sand mining and beach nourishment are documented there is much that is not known or studied (National Research Council 1995). NOAA Fisheries Service and the FWS began raising questions over related effects as long as twenty years ago. In North Carolina and South Carolina concern over nearshore populations of mole crab (*Emerita talpoida*) and donax (*Donax spp.*) was raised with several projects. Although frequently requested, no long term studies on impacts to these and other beach fauna were ever performed. The fate of these species, from a population perspective, is of concern since they are important food items for transitory and resident fishes (e.g., Florida pompano, kingfishes, and spot) that are of economic and recreational importance (Hackney et al. 1996). Limited studies performed by Reilly and Bellis (1978) showed significant reductions in occurrence and biomass of mole crabs and Donax at nourished beaches. Considering that many miles of southeastern beach front are now filled and/or subjected to scraping and sand relocation each year the cumulative effect of this activity could be substantial. Reviews of numerous beach nourishment projects suggest that the overall infaunal communities recover relatively rapidly (months to less than 1yr) although some species may remain adversely affected (NRC 1995). Much depends on the compatibility of the material placed on the beach relative to what was present prior to the project.

In Florida, beach nourishment projects require the dredging and filling of millions of cubic yards of fine sediments among shallow cross-shelf habitats, repetition or these activities at 3-10 year intervals, and tens of millions of dollars in annual expenditures (ACOE 1996). A U.S. Fish and Wildlife report (2004) prepared pursuant to Resolution 4 from the 8th Coral Reef Task Force meeting held on October 2-3, 2002, in San Juan, Puerto Rico, concluded that projects involving filling and dredging for beach nourishment and port development have caused the most impacts to coral reef habitats in southeast Florida since 1985. Among mid-shelf sand plains, often having nearby reef habitats, dredges create large craters and increased turbidity. At both dredge and fill sites, acres of shallow water hard bottom, worm reef, seagrass, or other habitats can be directly buried or subjected to elevated turbidity. Nearshore reefs buried or indirectly affected by dredging in south and central Florida can be utilized by over 325 invertebrate species (Nelson 1989), 190 fish species, and serve as nursery habitats for many managed species (Lindeman et al. 2000). The timing of burial and anthropogenic turbidity spikes may have important effects upon the recruitment of settlement-stage fishes and invertebrates. Early spring through early fall dredge related burial of hard bottom may eliminate habitat required by larvae of many marine organisms during peak recruitment periods (Hackney et al. 1996; Lindeman and Snyder 1999).

Based primarily on summary tabulations of data for southeast Florida within ACOE (1996), Lindeman (1997) estimates that:
At least 47 large-scale offshore dredge and inshore fill projects have occurred since 1960.
Approximately 97 additional large-scale dredge projects are conservatively planned to occur between 1997 and 2046.
Over 48,000,000 cubic yards of offshore sediments have been dumped within an intertidal/subtidal corridor of approximately 500 feet x 110 miles in the last 36 years.
Over 80,000,000 additional cubic yards of excavated offshore material may be dumped within the same corridor of subtropical southeast Florida in the next 50 years.

Long-term estimates of mean turbidity values under natural conditions are not available for most areas. Therefore, the percentages of affected animals and algae that can tolerate repetitious (e.g., 2 to 4 hours to 4 to 6 times a day for three months) sedimentation and elevated turbidity events (that may approximate continuous three-month storms), are unknown. With exception of hurricanes, highly turbid nearshore conditions in southeast Florida are typically the product of winter storms and heavy runoff during the rainy season. Near Miami, Florida turbidity in the nearshore hard bottom habitat is highly variable, and affected by winds, longshore currents, swell condition and upland runoff. Summer-fall months normally show lower turbidity levels of 1-4 NTUs (Nephelometric Turbidity Units) and winter-spring months show higher average levels (3-7 NTUs) (Miami-Dade DERM unpublished). Direct effects of dredging activities on corals have been discussed by Marszalek (1981), Goldberg (1988) and Blair et al. (1990). Although sublethal effects of elevated turbidity are poorly known in tropical marine environments, some information is available. Bak (1978) showed that a relatively short period of dredge-induced turbidity stress created an abrupt decrease in growth in two species of hard corals (*Agaricia* and *Madracis*). From both the magnitude and duration of suppressed calcification, he concluded that such metabolic shock may have long-term consequences on reproduction. Long-term resuspension of bottom sediments has been shown to adversely affect an important hard coral, *Montastrea annularis* (Dodge et al. 1974). Teleniski and Goldberg (1995a; 1995b) have recently demonstrated negative effects of sediment loads on hard corals at turbidity levels of approximately 18 NTUs. This is noteworthy, as the Florida state administrative threshold for temporary shut-downs of dredge operations is substantially higher (29 NTUs). Such work is needed for other taxa and would provide a scientific basis for maximum turbidity thresholds (Goldberg 1988; Teleniski and Goldberg 1995b). Herrnkind et al. (1988) demonstrated that increased siltation can cause direct loss of critical habitat for spiny lobster recruitment. Enhanced resuspension of sediments over time and chronic turbidity may lower key growth and reproduction rates of some algal and invertebrate populations which are a basis for primary and secondary production on an ecosystem scale (Lindeman 1997b). The potential for management decisions to multiply over time and impact unintentionally large spatial scales is of concern (Odum 1982; Rothschild et al. 1994) and is particularly relevant when affected species are also over harvested (Ault et al. in press).

Adopting 15 NTU above background as a threshold level for turbidity in Florida and other areas where waters are naturally not turbid is supported by sound science and appropriate for the following additional reasons:
1. Research associated with investigations by Telesnicki and Goldberg (1995) examined the effects of turbidity measured as an absolute value. In southeast Florida, turbidity standards are based on relative conditions (i.e., above background conditions);
2. We do not have adequate statistical competency to conclude that turbidity monitoring stations would be positioned in a manner that would capture the densest portion of the turbidity plume. Inherent risks associated with this warrant adoption of a more conservative threshold level; and
3. Although elevated turbidity levels may not directly or instantaneously kill corals, construction-induced turbidity may have long-term adverse impacts on corals (e.g., reduced reproductive health) that cannot be detected without carefully designed long-term monitoring.

In other areas of the southeast where waters are more naturally turbid and sensitive bottom fauna such as reef habitat are not present, a higher NTU criteria may be desired. For example, the South Carolina Department of Health and Environmental Control has adopted a threshold of 25 NTU for impaired versus non-impaired estuarine and marine waters. While monitoring of turbidity plumes associated with beach nourishment operations in South Carolina have been limited, Van Dolah et al. (1994) monitored sediment plumes associated with a beach nourishment operation on Folly Beach, South Carolina to determine the both the amount and extent of turbidity. During calm seas, values of about 100NTU were measured in the surf zone at the pipeline outfall. Turbidity levels dropped to less than 50 NTU in the upcurrent direction over a fairly short distance (less than 200 m), and more slowly in the downcurrent direction 500-1000 m. Under more turbulent conditions of strong winds and rough seas, turbidity levels increased to over 200 NTU directly in front of the pipeline and higher turbidities were documented over a larger extent of the beach. However, turbidities in South Carolina‘s surf zone are naturally turbid, and turbidity values of about 100NTU were occasionally recorded at a reference beach in the Folly Beach study. In addition, resource management agencies are examining the value of integrating Acoustic Current Doppler Current Profile (ADCP) technology into water quality monitoring protocols. ADCP is an instrument with capability of collecting acoustic backscatter data through the full depth of the water column and has demonstrated utility in other projects, especially in areas that are characterized by shifting currents (e.g., a project in Long Island Sound in which ADCP was utilized in the turbidity monitoring program in order to accurately locate the plume so that targeted water column sampling could be accomplished). We note that the nature of a plume in open water can be highly variable both spatially and temporally and can be further complicated by winds and seas. Therefore, to overcome these challenges and position the monitoring in the right place at the right time, full depth profiling with ADCP may be essential to the integrity of the monitoring performed. Use of third party environmental inspectors for water quality monitoring has also been included in recent large scale offshore construction project Corps of Engineers permits.

The SAFMC is concerned that excavation of the offshore shoals could have significant adverse consequences to the shoreline and living marine resources. Between 1995 and 2006, the Minerals Management Service (MMS) provided approximately 14 million cubic yards of material from the Outer Continental Shelf (OCS) for 9 coastal projects in Florida (8) and South Carolina (1). Although many offshore shoals have not been thoroughly studied with respect to fish utilization, SAFMC believes the shoals serve as a benthic nursery area, refuge, and feeding ground for a variety of fishery resources. The SAFMC identifies sandy shoals as EFH for migratory pelagic fish, including king mackerel, Spanish mackerel, cobia, and dolphin. Clarke et al. (1988) and Michel et al. (2001) note the geomorphology of offshore shoals provide a unique assembly of micro-habitats that facilitate high biological productivity.

The MMS and Corps of Engineers are evaluating the St. Lucie Shoal (located offshore St. Lucie and Martin Counties, Florida) as a potential excavation site for beach renourishment in Dade and St. Lucie Counties. Anecdotal evidence suggests that the shoal is biologically unique and diverse, supporting fisheries that are economically and recreationally important, such as the migratory species listed above, sailfish, and prey species consumed by these fishery species.

In South Carolina, a survey of multiple sites dredged for beach nourishment purposes identified that most sites were slow to refill (average of 7 yrs among 5 sites) and generally refilled with non-beach compatible material (Van Dolah et al. 1998).

The SAFMC is concerned that mining shoals for sand may alter the local wave climate bringing about erosion that could affect EFH. Through an evaluation of the potential impacts from dredging linear shoals in the U.S. Gulf and Atlantic continental shelves, Hayes and Nairn (2004) concluded that the deflation of a shoal feature could change wave patterns between the shoal and the shoreline. In turn, such dredging could change longshore and cross-shore sand-transport patterns and erosion and accretion rates along the shore. Kelley et al. (2004) verified this conclusion in their examination of a borrow site offshore Martin County (depths were approximately 8 to 10 m), and recommend application of wave transformation numerical modeling tools that recognize the random nature of incident waves as they propagate onshore when examining incremental and cumulative changes from sand dredging on the continental shelf.

Furthermore, the SAFMC is concerned that excavation of nearshore borrow areas in addition to the placement of fill in nearshore areas could adversely affect hardbottom reefs in the area that are known to support corals and worm reefs colonized by *Phragmatopoma lapidosa*. Nearshore hardbottoms and worm reefs are also identified as EFH and HAPC by the SAFMC. These reefs reduce wave energy and stabilize shorelines (Kirtley 1967; Kirtley and Tanner 1968) and provide structural habitat for hundreds of fishery organisms (Gore et al. 1977; Nelson 1989; Lindeman and Snyder 1999). Avoidance and minimization of impacts to hardbottom resources is needed.

Due to the importance of these concerns, SAFMC recommends that MMS and the COE continue to coordinate closely with the NOAA Fisheries Service Habitat Conservation Division to ensure the EFH assessments and NEPA documents contain sufficient detail to support federal decision making.

Other offshore mineral and mining does not presently occur along the South Atlantic coast. Extensive phosphate deposits have been located in Onslow Bay in North Carolina and large quantities of mineral nodules containing manganese and other metals are abundant along the continental shelf floor. It is reasonable to conclude that mining of these and other materials could become economically feasible. If initiated, mining of marine bottoms would cause substantial bottom disturbance that could impact productive hard bottom communities, shellfish beds, and wintering grounds for demersal fish. Since related port and processing facilities do not presently exist, new mooring and dockside facilities would be needed and related secondary impacts would be expected. These impacts are discussed in detail in Section 7.4.2.1 of this document.

6.1.2.5 Oil and Gas Exploration, Development, and Transportation

Extensive areas of the South Atlantic have been designated and blocked off for oil and gas development. Prior to 2003, this activity had been relatively dormant, unlike the pipelines and liquefied natural gas (LNG) facilities that proliferate in the Gulf of Mexico. Initial exploration in the vicinity of Cape Hatteras several years ago did not advance due to environmental and other concerns including consistency issues associated with North Carolina‘s Coastal Zone Management Program. As of this writing, interest in the potential for renewed oil and gas exploration off North Carolina is again being considered. Environmental Impact Statements have been prepared for Mid-Atlantic Sale 121 and South Atlantic Sale for the exploration of oil and gas offshore of Cape Hatteras, North Carolina. Should gas or oil be found, the laying of pipe to North Carolina‘s shoreline facilities would likely have to traverse barrier islands and associated wetlands. As oil and gas levels decline, exploration will undoubtedly resume and if economically viable reserves are located, this activity could expand and inshore and offshore EFH could be at risk. There are currently three natural gas pipeline proposals in Florida that propose to construct pipelines from the Bahamas to southeast Florida. Between 1996 and 2006, NOAA Fisheries Service reviewed 548 applications and support documents associated with pipelines in the South Atlantic area. The NOAA Fisheries Service Southeast Region Habitat Conservation Division (HCD) office is engaged in three separate EFH consultations for natural gas pipeline projects proposed to be constructed from southeast Florida to the Bahamas. One of three projects (AES Ocean Express) has received Department of the Army (DA) authorization and a Federal Energy Regulatory Commission (FERC) license to proceed with construction. However, to our knowledge, all of these projects are still awaiting the necessary approvals from the Bahamian government.

One pipeline company (Calypso), recently filed an application with the U.S. Coast Guard to construct a deepwater port located approximately 5 to 10 off the eastern coast of Florida to the northeast of Port Everglades in a water depth of approximately 640 to 950 feet.

Potential Threats to EFH from Oil and Gas Exploration, Development, and Transportation
Potential threats include elimination or damage to bottom habitat due to drill holes and positioning of structures such as drilling platforms, pipelines, anchors, etc., water intake and

impacts to ichthyoplankton, release of harmful and toxic substances form extracted muds, oil, and, gas and from materials used in oil and gas recovery; discharges of potentially large volumes of drilling fluids (muds) used during the well drilling process and produced (brine) water from the extraction phase; damage to organisms and habitats due to accidental spills; damage to fishing gear due to entanglement with structures and debris; and damage to fishery resources and habitats including deep water habitats, due to anchoring and effects of blasting (used in platform support removal); and indirect and secondary impacts to nearshore aquatic environments affected by product receiving, processing, and distribution facilities.

The various threats to EFH that would result from natural gas pipeline installation and construction depend on project location and construction methods proposed. Horizontal directional drilling was one of the primary nearshore construction methods evaluated, but eventually ruled out due to concerns that pertain to frac-outs, which are generally caused when the drill head moves through an area of unconsolidated sediments. Frac-outs are typically monitored through monitoring the hydrostatic pressure differential. Considering that frac-outs can occur anywhere along or near the pipeline route, pressure monitoring alone was not sufficient in areas that support reef. Frac-outs can occur as a slight release of mud or an uncontrolled flow of drilling muds.

According to Stauber et al. (2003), with sufficient geotechnical information it is possible to calculate a maximum allowable borehole pressure curve for a given HDD bore profile. Using this information, preliminary bore plans could be developed that provide reasonable assurance that the bore could be completed without incident. Therefore, SAFMC recommends that pipeline applications include an HDD Risk Analysis to ensure that the bore paths identified are the least likely to contribute to a frac-out.

Other threats to EFH could occur as a result of offshore dredging of exit pits and direct burial of resources through the pipeline placement, movement, and/or articulated concrete mats which are typically proposed for use in water depths of less than 200 feet for pipeline stabilization. In addition, drilling muds and the use of additives, such as Envis (a mixed metal hydroxide) or StaFlo (a polyanionic cellulose) are commonly used during drilling operations to control drilling mud flow and fluid loss. Another potential threat is hydrostatic testing which is typically proposed to verify that the pipeline was properly installed and structurally sound. Chemicals may be proposed for use in hydrostatic testing and can include corrosion inhibitors, biocides, oxygen scavengers, and leak detection dye that would be used for pipe treatment and as seawater additives.

Another nearshore construction approach involves tunneling, which is preferred over HDD but has not been tested yet in nearshore areas of southeast Florida. Tunneling poses less risk to the marine environment because it may be possible to conduct operations independent of weather and it reduces or eliminates the risk of frac-outs because the operation is conducted under much less pressure and at greater depths. However other issues are still being evaluated, such as the potential for localized slumping or heave, tunnel failure, a higher probability of a frac-out near the tunnel exit location, and hydrostatic testing, as mentioned above.

To date, only one deepwater LNG port has been proposed in the South Atlantic. However, the Federal Energy Regulatory Commission has received three applications (including Calypso) to construct pipelines from southeast Florida to the Bahamas. To date, none of the applications has received approval from the Bahamian government to construct regassification facilities. Therefore, SAFMC is concerned about the potential for multiple deepwater ports to be proposed offshore southeast and east-central Florida.

The September 2006 Calypso application states that approximately 273 acres of deepwater habitats could be impacted as a result of anchoring activities. Benthic organisms may be adversely affected from direct crushing and disturbance of sediments in the immediate vicinity of the anchors. The Calypso LNG terminal is proposed to be located on or adjacent to the Miami Terrace, which is a proposed deepwater coral HAPC. Hardbottom and coral resources found along the Miami Terrace and Escarpment are identified as EFH and HAPC by the SAFMC. Reed et al. (2006) characterized the fauna on the Miami Terrace and Escarpment as consisting of gorgonacean octocorals, colonial scleractinian corals (including thickets of *Lophelia pertusa*, *Madrepora oculata*, and *Enallopsammia profunda*), stylasterine hydrocorals, and Antipatharia. Diverse populations of the sponges Hexactinellida and Demospongia also occur along the Miami Terrace and Escarpment. In addition, based on studies conducted for the Calypso Pipeline Final Environmental Impact Statement, side-scan sonar results from the area show highly reflective signatures, which suggests the substrate is hardbottom mixed with medium carbonate sands and silty sands. Unlike the open loop LNG facilities proposed and in operation in the Gulf of Mexico, the Calypso LNG facility is proposed to be a closed loop system (it should be noted, however, that Calypso could have chosen to use open loop regasification technologies and, given cost considerations, so might any other LNG company that looks at the Atlantic coast off Florida). Open loop systems use seawater for the regasification of LNG and water intakes can exceed 100 million gallons of water per day. However water intake associated with closed loop systems is only for engine cooling and can range from approximately 30-60 million gallons per day depending on the number, type, and duration of vessels at Port. With the closed loop system proposed in the South Atlantic, the discharge water would be approximately 13 degrees Fahrenheit warmer than the intake water.

Applications for LNG facilities should adequately consider potential impacts to fishery resources and the project‘s proximity to the Gulf Stream. The conditions and flow of the Gulf Stream are variable on time scales ranging from two days to entire seasons. Important spawning locations can occur along the Gulf Stream front (e.g., *Coryphaena*, *Xiphius*) (SAFMC 1998). Movement of the Gulf Stream front also affects the distribution of adult fishes (Magnuson et al. 1981); hook-and-line fishermen and longliners target much of their fishing effort in these frontal zones.

Biological and economic analyses of impacts related to impingement and entrainment of the various life stage histories of fishery resources are needed to allow the SAFMC, public, and NOAA to assess the costs of lost fisheries production from the water intake/discharge component of the Calypso LNG deepwater port. Such examinations should include detailed comparisons of the environmental impacts and environmental costs of alternative closed-loop regasification technologies to understand more fully the potential impacts to fishery resources. Analyses should be based on an assumption of 100% zooplankton mortality that would result from water intake, unless the applicant can show applicable studies demonstrating otherwise. In addition, surveys of the ichthyoplankton communities within project areas are needed because in many areas, including water off Fort Lauderdale, there are no site-specific data regarding ichthyoplankton resources. Such surveys should be designed to provide a quantitative assessment of the impacts to fishery resources. In addition, the surveys should be designed to support the monitoring of impacts from port operations on fishery resources so that adjustments to those operations can be made in a timely manner. Although the continental shelf of the South Atlantic Bight has been the focus of moderate interest for exploration of oil and gas resources, there are presently no ongoing related activities in the region with exception of that mentioned above.

In addition to what is presented above and considering the current status of the industry, a brief overview of the facilities that might be emplaced on the Outer Continental Shelf (OCS) to facilitate oil and gas exploration, development, and production is also presented. This includes drilling vessels (jack-ups, semi-submersibles, and drill ships), production platforms, offshore moored terminals, and pipelines. Oil and gas related activities are inherently intrusive and pose a considerable level of threat to marine and estuarine ecosystems, including EFH. As discussed below, exploration and recovery operations may cause substantial localized bottom disturbance. Where large scale development is undertaken the area of impact may be greatly expanded and become regional in scale. The toxic nature of hydrocarbon products and certain drilling materials (e.g., drilling muds), spill cleanup chemicals, and the large volume of unrefined and refined products that must be moved within the coastal zone places large areas and resource bases as risk.

Structure emplacement can be expected to disturb some bottom area and, if anchors are deployed, the area of disturbance could be expanded. Jack-up rigs and semi-submersibles are generally used in water depths not exceeding 400 meters and disturb about 1.5 ha (3.7 ac) of bottom each. Conventional fixed platforms are also employed where water depths are less than 400 meters and they disturb about 2 ha (4.9 ac). Where water depths exceed 400 meters, dynamically-positioned drill ships may be used and sea floor disturbance is usually limited to the well site. Tension leg platforms may also be employed at these depths and the potential bottom disturbance area associated with these structures is about 5 ha (10.25 ac).

Each exploration rig, platform, terminal, and pipeline emplacement on the OCS can be expected to disturb surrounding areas. Exploration rigs, platforms, and pipe laying barges use an array of eight 9,000 kg anchors to position a rig and barge, and to move the barge along the pipeline route. These anchors are continually moved as the pipe laying operation proceeds and the total area actually affected by the anchors will depend on water depth, wind, currents, anchor chain length, and the size of the anchors and chain (MMS 1996). With conventional, fixed multi-leg platforms, which are anchored to the sea floor by steel pilings, explosives are generally used to sever conductors and pilings. These support structures are substantial in size since they must withstand hurricane conditions and have an average lifespan of about 20 years. The Minerals Management Service requires severing support structures at five meters below the sea floor surface so as to preclude interference with commercial fishing operations.

Possible injury to biota from use of explosives extends horizontally to 900 meters from the detonation site, and vertically to the surface. Based on MMS data, it is assumed that approximately 80% of removals of conventional fixed platforms in the Gulf of Mexico, in water less than 400 meters in depth, will be performed with explosives (MMS 1996). Alternative methodologies such as mechanical cutting and inside burning are often ineffective and are hazardous to workers.

Associated bottom debris commonly associated with over water oil and gas operations includes cable, tools, pipe, drums, assorted trash, and structural parts of platforms. The amount of bottom debris deposited around a site may vary and may be measured in tons. Extensive analysis of remotely-sensed data within developed lease blocks indicates that the majority of ferromagnetic bottom debris falls within a 450 meter radius of the site. The Fisherman's Contingency Fund, which was established by the oil and gas industry, provides recourse to commercial fishing interests for recovery of equipment losses due to shrimp net entanglement (MMS 1996).

Blowouts occur when improperly balanced well pressures result in sudden, uncontrolled releases of petroleum hydrocarbons. Blowouts can occur during any phase of development: exploratory drilling, development drilling, production, or workover operations. About 23% of all blowouts will have associated oil spills, of which 8% will result in oil spills greater than 50 barrels, and 4% will result in spills greater than 1000 barrels. In subsurface blowouts, sediment will be resuspended and bottom disturbance will generally occur within a 300 meter radius. Whereas larger grain sediment will settle first, fined grained material may remain in suspension for periods of up to thirty days or longer. Fine grained material may be redistributed over a significantly large area depending on the volume of sediment disturbed, bottom morphology, and currents (MMS 1996).

The major operational wastes associated with offshore oil and gas exploration and development include drilling fluids and cuttings, and produced waters. Other important wastes include: from drilling--waste chemicals, fracturing and acidifying fluids, and well completion and workover fluids; from production--produced sand, deck drainage, and miscellaneous well fluids; and from other sources--sanitary and domestic wastes, gas and oil processing wastes, ballast water, storage displacement water, and miscellaneous minor discharges (MMS 1996). Major contaminants or chemical properties of materials used in oil and gas operations may include those that are highly saline; have a low ph.; contain suspended solids, heavy metals, crude oil compounds, organic acids, priority pollutants, and radionuclides; and those which generate high biological and chemical oxygen demands. Pierce et al. (1980) documented that wild fish have been injured by petroleum pollutants. Grizzle (1983) suggested that larger liver weights in fish collected in the vicinity of production platforms versus control reefs could have been caused by increased toxicant levels near the platforms. He also suspected that severe gill lamella epithelium hyperplasia and edema in red snapper, vermilion snapper, wenchman, sash flounder, and creole fish were caused by toxicants near the platforms. These types of lesions are consistent with toxicosis.

Accidental discharge of oil can occur during almost any stage of exploration, development, or production on the OCS or in near shore base areas. Oil spills may result from many possible causes including equipment malfunction, ship collisions, pipeline breaks, human error, or severe storms. Oil spills may also be attributed to support activities associated with product recovery and transportation. In addition to crude oil spills, chemical, diesel, and other oil-product spills can occur with OCS activities. Of the various potential OCS-related spill sources, the great majority are associated with product transportation activities (MMS 1996).

As of this writing, only test wells have been drilled in the South Atlantic Bight area and these have been confined to inshore areas. All of these wells were capped immediately after drilling. No production or transportation facilities such as offshore terminals and pipelines have been built, nor are any such facilities currently planned in Bight waters. Despite this, millions of barrels of crude oil and refined product transit South Atlantic Bight waters by tank vessel every year and the potential exists for the discharge of thousands of barrels of oil due to vessel collision or sinking. Discharge of untreated ballast water from transiting vessels is also a chronic low level source of petroleum-based pollution.

6.1.2.6 Commercial and Industrial Activities

Direct physical encroachment into offshore environments by industrial activities is relatively limited along the South Atlantic seaboard. Notable exceptions include thermal intake and outfall structures associated with power plants in North Carolina and Florida, and sea walls that are used to protect commercial and industrial development. Several municipal sewage outfalls which discharge commercial and possibly light industrial wastes also exist. Although direct physical impacts may be minor on a regional scale, water quality effects are largely unknown. Indirect effects, such as those associated with point and nonpoint-source discharges are thought to be substantially greater since it has been shown that discharges, including trash and debris, from land based activities may reach coastal waters and food webs.

Commercial development for hotels, motels, and related infrastructure along the South Atlantic shoreline has been extensive. Because many of these developments are located on unstable and shifting coastlines, maintaining associated buildings, revetments, bridges, causeways, beaches etc. has, and will continue to have an adverse effect on nearshore and offshore processes and environments.

Potential Threats to EFH from Commercial and Industrial Activities

Potential threats include: direct and/or non-point-source discharge of chemicals, placement of intake structures, and protective sea walls (often used in connection with commercial establishments), and cumulative and synergistic effects caused by these and other industrial and non-industrial related activities.

Future exploration and recovery of marine resources and placement of offshore mooring and unloading facilities could substantially threaten offshore EFHs. Although none of these activities or facilities are presently being planned, it is likely that continued economic growth, depletion of limited natural resources, and use of limited coastal lands will eventually lead to greater exploitation of offshore resources.

Electric power generation is needed for commercial and industrial development, and for residential purposes (See Section 4.1.1.4). Between 1996 and 2006, NOAA Fisheries Service evaluated 85 proposals to construct new or expand existing electric generation facilities. When located in coastal waters, power generation facilities may adversely affect EFH and associated biota. Potential threats include direct displacement of wetlands, submerged bottoms, and vegetated upland buffer areas for generation facilities and ancillary uses such as fossil fuel storage, cooling towers, and water intake and outfall structures; construction of navigation channels and docks for unloading coal, oil, and other materials needed for operation of generators and equipment; discharge of toxic substances from air emissions; cooling waters (e.g., chlorine); and from point and nonpoint-source discharges emanating from impervious surfaces and coal and slag piles; discharge of thermal discharges that may be lethal to flora and fauna, or that serve as attractants that subject fish, invertebrates, and marine mammals to thermal stress when changes in plant operation or weather occur; and entrainment and impingement of living marine resources in which organisms succumb to or are damaged as a result of entrapment in intake structures or capture on screens.

An example of an electric power generation plant and threats to EFH is the Florida Power and Light‘s Turkey Point Power Plant, located along Biscayne Bay in Dade County Florida, which directly impacted over 24 acres of estuarine emergent wetlands, including mangrove wetlands, seagrass, and open water habitat in order to construct a natural gas-fired electric generating facility to provide electricity to meet the projected 2007 demand in southeast Florida. An additional 10.7 acres of wetlands were impacted through secondary effects. The wetlands at the subject site are high quality, uncommon, and provide direct benefits to the fishery resources of Biscayne Bay. The bay‘s extensive seagrass beds, mangrove wetlands, and hardbottom communities support a diverse array of fishes and invertebrates including over 512 species of fishes and over 800 species of invertebrates which have widely variable environmental requirements for growth and reproduction.

Although relatively minor in its present scale, the commercial harvest of *Sargassum* from coastal waters off North Carolina is of concern. *Sargassum* weed lines and associated frontal zones provide cover, trophic, and other attributes needed to sustain endemic fish and invertebrates of the pelagic *Sargassum* community and associated fauna. The weed lines may be especially important during early life stages of sea turtles and certain fish and they are important sites for the North Carolina and South Carolina offshore recreational fishery.

The occurrence of methyl mercury in the flesh of the large piscivorous fish such as king and Spanish mackerel and other large pelagic and demersal species such as amberjack, wahoo, snapper, and grouper has been documented and is of concern largely with respect to human consumption of these species (D. Engel, personal communication). The probable source of these contaminants is atmospheric input from worldwide inventories associated with emissions from incinerators, fossil fueled power plants, automobiles, and industry. As such, the regulation of surface water contamination from atmospheric pollution may require local, regional, and international efforts.

Effects related to commercial development are similar to those from urban and suburban development and the discussions in Section 4.1.1.4 apply. Further, effects of shoreline modifications such as beach nourishment are found in Section 4.1.2.3.

6.1.2.7 Artificial Reefs
Artificial reef construction in the South Atlantic has substantially increased over the last 10 years. Project scales range from single family homeowners applying to place reef balls under docks for lobster recruitment to 3,000 acre areas located in offshore areas. Project applications

typically state that the purpose of the project is to ―further develop three artificial reef sites to increase the marine flora and fauna within the area for local fishermen and SCUBA divers without detriment to the existing reef structures or fish populations.‖ However, artificial reefs are also constructed to replace natural reef habitats. Construction at the larger scale sited typically involves the placement of a variety of materials including concrete, limestone boulders, submerged vessels, and other approved items.

Potential Threats to EFH from Artificial Reefs
Potential threats to EFH include permanent conversion of one habitat type to another, introduction of predators, possible increased fishing activity and relic gear on structures.

Although the SAFMC recognizes and appreciates applicant‗s efforts to provide additional marine habitat, information regarding the level of impact this project would have on EFH resources is needed in the application process. This information need includes a thorough assessment of environmental impacts and details concerning its design and specifications.
The type of information that should be contained in an artificial reef application includes:
 It should be demonstrated that the project will provide enhanced marine fisheries habitat. This may be achieved through (but not limited to):
o Identifying the specific fisheries and life history stages that will be enhanced by the proposed work.
o Demonstrating a clear link between the structural design and the fisheries the artificial reef will support.

 The applicant should demonstrate full consistency with NOAA‗s *National Artificial Reef Plan* (1985) and the draft plan revision (2001)1, including, but not limited to, the following provisions:
o Demonstrated consistency with the applicable state‗s artificial reef plan (e.g., the State of Florida‗s Artificial Reef Plan). Through this, the applicant should:
o Have a specific objective for fisheries management or other purpose stated in the goal of the statewide, or site-specific plan;
o Have biological justification relating to present and future fishery management needs;
o Have minimal negative effects on existing fisheries, and/or conflicts with other uses;
o Have minimal negative effects on other natural resources and their future use;
o Use materials that have long-term compatibility with the aquatic environment;

1 National Artificial Reef Plan (revised 2001). National Marine Fisheries Service. Available on-line at: http://www.nmfs.noaa.gov/irf/Revised_PLAN_11_16.pdf Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 67

o Conduct monitoring during and after construction to determine whether the reef meets permit terms and conditions and is functioning as anticipated.
 The applicant should ensure that the proposed artificial reef structure will not threaten the integrity of natural habitats in the area, including live/hardbottoms, corals, seagrasses, and macroalgae;
 The application should verify that any vessels deployed have been cleaned in accordance with Environmental Protection Agency Guidelines;

The constructed reef should remain stable during a 100-year storm event;
The applicant should identify the most extreme sea state and wave surge conditions under which work will be undertaken; and
An entity should be identified to demonstrate the capability of assuming long-term financial liability for the deployment, biological and stability monitoring, and maintenance of the artificial reef.

Artificial reefs can serve as effective fishery management tools (when coupled with additional fishery management measures, for example the designation of no-take zones) to attract fish and, in some situations, mitigate for anthropogenic and natural damage to coral and hardbottom reefs. The SAFMC concurs with the leading artificial reef researchers in this region (see Bohnsack 1989) that artificial reefs are unlikely to benefit heavily exploited or overfished populations without other management actions. Conversely, if not properly sited they may have only minimal habitat value and could even degrade existing reef resources if placed on or in close proximity to such habitats. Artificial reefs are also constructed as mitigation reefs. A U.S. Fish and Wildlife report (2004) prepared pursuant to Resolution 4 from the 8th Coral Reef Task Force meeting held on October 2-3, 2002, in San Juan, Puerto Rico, concluded that projects involving filling and dredging for beach nourishment and port development have caused the most impacts to coral reef habitats in southeast Florida since 1985. The 26 Florida projects (16 completed; 10 pending) reviewed in this report impacted 217 acres of reef, and mitigated with 113 acres of artificial reef. However, a study is needed that would provide information as to impacts to hard bottom communities of shoreline projects, including whether proposed mitigations are adequate to offset the environmental impacts of the activities. General practice in Florida is to permit mitigation for shallow hard bottom communities in deeper waters is contributing to a substantial net loss of the shallow communities and related functions.

6.1.2.8 Alternative Energy Technologies

Sections below excerpted from MMS Alternative Energy Synthesis report: Michel, J., Dunagan, H., Boring, C., Healy, E., Evans, W., Dean, J.M., McGillis, A. and Hain, J. 2007. Worldwide Synthesis and Analysis of Existing Information Regarding Environmental Effects of Alternative Energy Uses on the Outer Continental Shelf. U.S. Department of the Interior, Minerals Management Service, Herndon, VA, MMS OCS Report 2007-038. 254 pp. See Report for references in following sections.

Offshore wind turbines

An offshore wind farm is a set of turbines that generate electricity from the mechanical force that the wind imparts upon an object and are specifically designed for their oceanic location. Each modern oceanic turbine is capable of producing up to 4.5 megawatts of power (some older turbines installed in the 1990s produced less than 1 megawatt, newer turbines under development may produce 5 to 10 megawatts), and the hub of the turbine is 180 feet or more above the sea surface. Present proposals include systems with blades that will reach more than 510 feet above the sea surface. The number of turbines in a farm varies and will be affected by economics, space, and demand for the electricity generated. The number of turbines in proposed farms ranges from three units in a proposed research setting to over 150. The turbines need to be separated from each other by a distance of 0.25 mile or more in order to reduce the effect one turbine has upon the wind field experienced by adjacent turbines. Wind farms include a distribution platform that serves as a hub for the cables that collect power from each turbine and the fewer, but larger, cables that carry the power to shore.

A recent study conducted by the Minerals Management Service (MMS 2007) cites the following as the current primary economic and technical feasibility determinants that affect the choice of sites for offshore wind parks:
 Availability of a substantial, relatively constant wind resource
 Shallow water (less than 30 meters deep)
 Proximity to an area of high electricity consumption
 Distance to shore

Water depth is a critical design element that currently limits installation in deeper waters because of technology and economic constraints. Existing wind parks in Europe are installed in very shallow water (up to 15 m deep). Most North American wind resources are in water greater than 30 m deep, requiring development of economically feasible new technologies for wind turbine structures that can withstand wave and wind action in deeper areas (MMS 2007).
In addition to the water depth limitations of technology as of 2007, significant economic concerns are associated with the distance from shore and the length of subsea electrical cable required to reach the onshore electrical grid. Although available wind turbine designs allow installation in waters less than 30 m deep, wind parks operating in Europe are in shallower coastal areas (water depths of approximately 15 m). In the United States, wind parks are likely to be developed along the Atlantic seaboard and the Gulf of Mexico (MMS 2007).
The Cape Wind project offshore of Massachusetts and the Long Island Offshore Wind Park (LIOWP) offshore New York are in the environmental impact statement (EIS) stage, and other projects are planned along the northern and central U.S. coast. In addition, two leases have been granted by the State of Texas to develop wind parks off the coastline of Padre Island and Galveston Island. Additional projects are in the early planning stages along the U.S. east coast and Gulf of Mexico (MMS 2007).

Potential threats to EFH from Offshore Wind Turbines
Operational characteristics of each turbine design and its size are influenced by the minimum sustained winds occurring in an area and needed to make the wind farm profitable. Studies from the northeastern U.S. conclude a minimum wind speed of 16 mph or more is needed, studies from the southeastern U.S. conclude wind speeds of 11 to 13 mph are sufficient (Stewart 2005). Analyses are not simple; wind persistence, direction and natural turbulence can limit a turbine's ability to produce electricity even though its blades are spinning. Analyses must also consider the efficiency of the turbines and the number of days in a year when the wind reaches or exceeds the minimum speed required to produce electricity. Other factors that influence the feasibility of establishing a wind farm include proximity to an established electrical grid and water depth, because market availability and water depth affect construction cost. Some authorities suggest 180 feet is a maximum depth; developers of the wind farm off Cape Cod, MA, actively sought waters less than 50 feet deep. Lloyd's Insurance has set 12 fathoms as their insurance risk limit. The occurrences of high winds are an issue since they can damage the turbine systems. Wind speeds that cause the blades to rotate above 14 revolutions per minute trigger most systems to shutdown.

In the United States, there are no offshore wind farms in operation, although six projects are currently being considered (Dennehey 2006; Ludwig 2006). Two are off the coast of Cape Cod, MA, two of the coast of Texas, one off the coast of Long Island, NY, and one is being considered off the Pacific Northwest Coast. Evaluations of the general environment off North Carolina and Georgia by universities conclude that wind farms warrant further investigation in these areas (Halks et al. 2005; Stewart 2005). Offshore wind farms have been established in Europe, especially in Denmark, and business forecasts indicate additional farms are likely due to tax and business incentives that focus on renewable energy (Danish Energy Authority 2005, see also http://home.planet.nl/~windsh/offshoreplans.html).

There are three general designs currently in use for anchoring turbines to the sea bottom, and the design chosen affects the extent of the environmental impacts (Danish Energy Authority 2005). A gravity foundation uses a large base (much broader than the pylon) with supplemental mass being placed on the base structure to anchor the pylon on the seafloor. A monopile base is a piling driven deep into the sea floor to create the stable anchor and is similar in diameter to the pylon itself; monopiles are currently used in water depths up to 60 feet. Multi-pole bases consist of piling systems similar to those used in small offshore oil and gas platforms; pilings are driven into the sea bottom over an area that is broader than the pylon that supports the turbine and the pylon is attached to a framework and platform that links the pilings. When commenting on a proposal by Cape Wind Associates for a wind farm on Nantucket Sound, MA, NOAA Fisheries Service indicated a preference for the 46-ft diameter, monopole design because it impacts less sea bottom and fishing gear is less likely to snag on this type of structure. Research being done in Europe is examining the feasibility of floating foundations and hybrids between monopile and gravity foundations that will allow farms to be located in deeper water without requiring a foundation that occupies a large amount of sea bottom. One of the wind farms proposed for New York plans to investigate the stability of a jack-up barge as its base, and the wind farms proposed for Texas are exploring use of oil and gas platforms that are no longer needed by the petroleum industry.

Long-term impacts to coastal ecosystems from wind farms are unclear because only a few offshore wind farms have existed for more than 10 years. However, all the wind farms recently constructed or authorized in Europe include substantial monitoring programs, so lack of data should not remain a problem for long. U.S. Army Corps of Engineers (2004) and the Danish Energy Authority (2005) provide initial lists and summaries of the impacts that can be expected from an offshore wind farm and the latter also provides Internet links to Web sites planned for distributing future study results.

Direct impacts to coastal ecosystems include usurpation of seafloor habitat(s) by the pilings, distribution platforms, and cables that connect the turbines to the onshore power grid. Especially when the monopile design is used, the cumulative area impacted is small; for example, Cape Winds Associates estimates the pylons from their farm of 130 turbines would occupy less than one acre of sea bottom. Construction equipment impacts during cable and system installations would add to this acreage. Direct effects to the sea bottom also may occur from alteration of current fields moving past the foundations, but these impacts to be manageable in most circumstances.

The most obvious affect of the pilings on marine biota will be from the structures serving as fish habitat. Many fish are attracted to any structure that provides relief from the otherwise featureless sea floor. Benthic organisms, which may adhere to a pylon or its base, depending on local conditions and construction materials, may add to the attractiveness of the structure to fish. Although unlikely to be an issue, there is some concern that electromagnetic fields (EMF) may disrupt the movements of sharks and other aquatic resources that navigate by sensing the earth's electromagnetic fields. Wind farms can transmit direct current, which has a greater capacity than alternating current to create localized EMF. Recent research indicates the severity of this impact may be small. Vibrations transmitted from the structures and systems to the water column and affecting the behavior of fish is a concern but not much is known about the severity of this impact. Monitoring in Europe has not found evidence of either EMF or vibration impacting aquatic resources (Ludwig 2006). Indirect impacts to marine biota may result from wind farms shifting navigation away from preferred routes into areas where marine mammals or fishery resources are more concentrated. The Federal Aviation Administration and military have recently identified that wind farms create a shadow effect on near ground, tracking radars.

Socioeconomic impacts have been controversial. Many members of the public object to the expected deteriorations in the vistas caused by the wind farms as well as wind farms occupying preferred fishing grounds. However, the Europeans have experienced a sharp increase in eco-tourism at their wind farm sites. The public also has been focused on impacts to seabirds, although impacts to birds seem uncommon based on preliminary evidence (Danish Energy Authority 2005).

Ocean current technology

(Excerpted from MMS 2007 report)
Ocean current technology is similar to wind technology, only underwater. Instead of wind, ocean current pushes turbine blades to transfer kinetic energy. Similar to wind turbines, the blades of the current turbines move at a very slow speed. For example, one type of design has vertical turbine rotors that rotate 10 to 30 revolutions per minute, which is approximately 10 times slower than ship propellers. Although the rotors move slowly, they produce a significant amount of energy because of the density of water moving them.

In the United States, no operating commercial systems using ocean current technology are connected to an electrical grid at this time (MMS 2006). However, the technology to harness ocean current energy as an alternative energy source is in the developmental stage. Demonstration and pilot studies of different prototypes are taking place throughout the world. Marine current velocities are lower than those of wind, but because water is 835 times denser than air, a 3-knot current has the kinetic energy of 161 km/h wind. The total potential energy contained in marine currents worldwide is estimated at approximately 5,000 GW (MMS 2006).

Available data indicate that current velocities between 2 and 5 meters per second (m/s) would be required to make ocean current energy technology economically viable at a particular site (MMS 2006).

In the United States, the most promising sources of ocean current energy include the Florida Current (part of the Gulf Stream) and the California Current (MMS 2006). These ocean current resources are located relatively close to shore and near centers of high electricity demand, making ocean current energy an attractive resource. In addition, ocean currents tend to be significantly more constant than wind resources, which can fluctuate greatly over relatively short periods of time.

A number of turbine designs exist, some of which have been through field testing while others are still in the development phase (MMS 2007). Florida Hydro is testing a disk-like design called the Open Center Turbine for use in the Florida Current (**Figure 6.1-1**).The moving parts of this technology are encased within the unit. Designed to produce 2.5 MW, the turbine was tested off Palm Beach, FL.

Figure 6.1-1. Open center current/tidal turbine with encased moving parts (Source: Open Hydro Group Limited).

Several other ocean current technologies are being developed. Those designs are tethered to the seabottom using anchors or on poles that extend from seabed foundations (ABP, 2004). These technologies are in the very early stages of development; however, they may be the most promising design for deeper, offshore applications on the OCS.

Solar technology

(Excerpted from MMS 2007 final report)

Solar energy technology has been producing useable energy from land-based, full-scale, grid connected power plants for more than a decade, but use of solar energy technology on the OCS is very limited. Economically feasible installation of full-scale solar energy projects on the OCS will depend on producing significant amounts of transmittable energy.

The possibilities for solar technology are not limited to large offshore solar plants; solar energy technology could be collocated with other alternative energy technologies. For example, solar collectors could be installed near the base of a wind turbine, and then used to augment energy output. Solar technology also could be installed as an alternative use for decommissioned oil and gas platforms on the OCS. Already some small, unmanned oil and gas platforms use solar panels for electricity needs. Solar panels are also used on buoys, platforms, and meteorological stations. The potential for annual average solar power varies greatly by latitude and cloud cover; solar radiation is significantly greater in the lower latitudes. In the United States, solar radiation is greatest in southern parts of the country. A literature review yielded no information on solar radiation levels offshore and along the OCS (MMS 2006). However, unpublished solar radiation data may exist as shipboard information collected during routine or research operations.

Solar energy is converted into useable energy through two basic technologies: thermal and photonic. Thermal technologies convert solar energy to heat. Photonic technologies absorb solar photons, which are then converted into electricity through photovoltaic (PV) cells. Technology is also in the early stages of development to store the photonic energy as hydrogen for later use, rather than convert it directly to electricity (MMS 2006).

Some solar technologies use concentrating mechanisms to focus heat or photonic solar energy into a collector. Technology and application of concentrated PV are not as advanced as concentrated thermal technology, but it is under development. Concentrated PV and thermal systems use mirrors or lenses configured to concentrate solar radiation on receiving panels.

Current solar energy technology has limited application on the OCS. It is distributed only to power buoys, weather stations, and small, unmanned oil and gas platforms. A literature review revealed no solar energy projects on the OCS at any stage of planning or development. Any offshore solar energy project would need to be mounted onto some sort of large floating or fixed structure (MMS 2006). The number of solar panels, and therefore, the size of the structure necessary to support an offshore commercial solar energy facility would vary depending on the solar radiation level at the location, the orientation of the panels, and weather conditions. Thermal solar technologies require dry, warm locations, and thus, current technologies likely would not be feasible on the OCS where humidity is high. PV solar technology surface area requirements also limit their application at OCS locations, where a floating platform would be required. Approximately 8 to 12 square meters is required for each kilowatt of capacity, meaning 0.8 to 1.2 hectares (0.008 to 0.012 km2) of PV cells would be required for each 1 MW of power output (MMS 2006). Concentrated PV systems developed for thermal solar projects are in early development. Efficient concentrated PV technologies may increase the economic feasibility of OCS solar applications because PV is more effective in humid environments.

Hydrogen Technology

Hydrogen technology would be used on the OCS as a transport or storage mechanism for energy produced by one of the other alternative energy technologies (wind, wave, current or solar). No projects were identified at any stage of planning or implementation for this type of technology. The best source of information on the possibilities of using hydrogen technology for storage or transport of energy on the OCS is the MMS (2006) white paperSince the application of hydrogen technology is so undefined at this stage, and because there are no current plans or prototypes for OCS application, the potential impacts were not included in the MMS report (2007).

6.1.2.9 Non-native or nuisance species

Indo-Pacific Lionfish

Lionfish (*Pterois volitans/miles* complex) are venomous coral reef fishes from the Indian and western Pacific oceans, that are now found in the western Atlantic Ocean (Whitfield et al. 2002; Hare and Whitfield 2003; Meister et al. 2005; Ruiz-Carus et al. 2006; Whitfield et al. 2006). Adult lionfish have been observed from the Turks and Caicos Islands throughout the northern Bahamas and from Florida to Cape Hatteras, North Carolina, including Bermuda. There is also recent evidence to suggest that lionfish have been found near Tampa Bay, Florida in the Gulf of Mexico (Ramon Ruiz-Carus, pers. Comm.). Juvenile lionfish have been observed in increasingly high numbers off New Jersey, New York and Rhode Island, generally in the fall of the year. Lionfish reports from the public (beginning in 2000) combined with quantitative surveys conducted from Florida to North Carolina (2004-2006) suggest that the number of lionfish continues to increase along the east coast and their distribution is expanding both in the northern (juveniles in northeast) and southern range (Whitfield et al. 2006; Whitfield unpublished data). Due to the large geographic range now inhabited by lionfish this invasion is likely irreversible as removal of this invader across this region would be expensive and take unprecedented resources.

Introductions of marine species occur in many ways. Ballast water discharge is a very common method of introduction for marine invertebrates, and is responsible for many freshwater fish introductions. In contrast, most marine fish introductions have resulted from intentional stocking for fishery purposes. In the case of lionfish, all evidence points to an unintentional or intentional aquarium release (Hare and Whitfield 2003).

Currently no management actions have been taken to limit the effect of lionfish on the southeast United States continental shelf ecosystem. Under this scenario we predict that; 1) the lionfish population and geographic range will continue to increase; 2) as a result of this increasing abundance, the impacts of lionfish on the southeast United States continental shelf ecosystem will become more noticeable; 3) eventually, human impacts from lionfish _stings' will occur along the southeast United States coast (Hare and Whitfield 2003; Whitfield et al. 2006).

The introduction and success of lionfish along the east coast may change the long-held perception that marine fish invasions are a minimal threat to marine ecosystems. The magnitude of this invasion as a stressor on marine ecosystems presently has not been quantified, but NOAA scientists have made a great deal of progress in understanding the lionfish introduction into the Western Atlantic. We have also made significant inroads in our understanding of many aspects of lionfish biology and ecology including reproduction, diet, population demographics and genetics. This section summarizes the current state of knowledge regarding the Atlantic lionfish population within five main topic areas: 1) Description and Distribution, 2) Reproduction, 3) Development, growth movement patterns and genetics, 4) Ecological relationships/Potential Impact and 5) Abundance and status of the stock.

Description and Distribution
The Indo-Pacific lionfish (*Pterois volitans/miles* complex, Scorpaenidae) is a venomous predator (Halstead 1970) native to the sub-tropical and tropical regions of the South Pacific, Indian Oceans and the Red Sea (Schultz, 1986). Lionfish are generally well known and recognized as a popular aquarium fish. Lionfish have venomous dorsal, anal, and pelvic spines, similar to other members of the family Scorpeanidae. The venomous spines are not known to be used in prey capture but are generally thought to be for self-defense and male/male agonistic displays during spawning (Fishelson 1975).

The present distribution (October 2006) of Indo-Pacific lionfish within the Atlantic is from southeast Florida to North Carolina, including Bermuda, the Bahamas, Turks and Caicos and along the northeast U.S. shelf as juveniles. Lionfish may have originated off the east coast of Florida in the early 1990's, but the actual source of the lionfish invasion remains unknown. In 2000, lionfish were first reported in North Carolina and Bermuda. In 2004, lionfish were first reported in the Bahamas, and in 2006 they were reported in the Turks and Caicos. Public reports combined with quantitative surveys suggest that both the number and geographic extent of the population continues to grow (Whitfield et al. 2006).

Within their native range lionfish are found on coral reefs and rocky outcrops from the surface to 50 meters (Schultz 1986). Within the South Atlantic Bight lionfish are widespread in abundance, found on all types of habitat (low relief hard bottom to high relief artificial structures) within water depths from 115 to 300 ft deep (Whitfield et al. 2006). By all accounts lionfish were already established (reproducing and dispersing) by the time the first surveys were conducted in 2004 and lionfish captures by hook and line are also on the rise within the past two years but these captures still vastly under-represent the extent of the lionfish population within the Atlantic. The large geographic extent of the lionfish distribution and the speed with which they occupied this area (since 2000) suggest they are very successful colonizers and competitors within their _new‘ ecosystem (Atlantic).

At present the primary factors that can potentially limit their distribution are available habitat, availability of prey and winter bottom water temperatures. Both habitat and prey appear to be plentiful, especially with the potential increase in prey resources made available through overfishing of many grouper species (likely competitors for prey) (Huntsman et al. 1999; NMFS 2004). Thus the minimum bottom water temperatures remain the single most important factor in controlling the present lionfish distribution within the Atlantic. This is not only evidenced by the shift in depth distribution from their native habitat (shallower) to the Atlantic (deeper) but also by winter bottom water temperature data collected in both nearshore (colder) and offshore (warmer; Gulf Stream influenced) locations (Whitfield et al. 2002; Whitfield et al. unpublished data). Minimum winter bottom water temperatures collected from locations where lionfish are known to over-winter support the thermal minimums found in laboratory studies (Kimball et al. 2004). Based on laboratory thermal minimums, lionfish would not survive water temperatures that dip below 10º C (Kimball et al. 2004). In North Carolina, this equates to an inshore depth limit of approximately 80 to 90 ft, depending on winter temperatures overall. Nevertheless, lionfish can still recruit into shallower areas but they are not expected to over-winter in shallow water (less than 80-90ft) north of Florida (see Figure 5, Kimball et al. 2004). However, since the thermal tolerance of fishes is known to change with changes in fish size and age (Wootton 1992), a series of mild winters could interact with the advancing size and age of Atlantic lionfish, eventually establishing subpopulations inshore of those currently surveyed. Therefore, the actual inshore limit remains unresolved off the Mid-Atlantic states. At their southern limit (southeast Florida, Bahamas, Turks and Caicos and Gulf of Mexico) there are no such depth or temperature constraints as water temperature remains warm year round. Thus lionfish have been reported in water depths as shallow as 3 ft in the Bahamas and Jacksonville, FL (Ruiz-Carus et al. 2006).

It is important to mention that although connectivity between the Bahamas and the Caribbean is low, there are certain locations such as the Turks and Caicos where connectivity is higher (Cowen et al. 2006). Since lionfish have free-floating eggs and larvae even minimal larval connectivity from the southeast U.S. and Bahamas could lead to invasion of the Caribbean and the Gulf of Mexico through a stepping-stone effect (Carr & Reed 1993; Cowen et al. 2006).

Reproduction

Lionfish can be characterized as gonochoristic, iteroparous, asynchronous, indeterministic batch spawners. This mode of reproduction is consistent with other members of the *Pterois* and *Dendrochirus* genera. Lionfish appear to be summer spawners off North Carolina with a resting period lasting throughout the winter. The lionfish spawning season is likely to increase at the southern range of their distribution (i.e., Florida/Bahamas).

From observations in the Red Sea, Fishelson (1975) has reported that lionfish are pair-spawners exhibiting a complex courtship during mating. Laboratory and shipboard observations indicate that lionfish release two buoyant egg balls during each spawning event consisting of a batch fecundity of approximately 30,000 eggs. Lionfish eggs are released while encased in a gelatinous mucus which breaks apart releasing the developing embryos within 48 hours. Lionfish do not exhibit sexual dimorphism; however, males do grow significantly larger than females. Sex ratio of lionfish in the Atlantic is approximately 1:1. Female lionfish appear to be sexually mature within two years of age corresponding to approximately 150 mm standard length (Morris, J.A., Jr., pers. comm.).

In their native range lionfish are reported as being solitary defending their home range against conspecifics; groups were typically observed only during mating (Fishelson 1975). In contrast within the Atlantic, lionfish are regularly found in groups, but, to our knowledge no mating behavior has been observed (Whitfield, pers. obs.).

Development, growth, age, movement patterns and genetics
The early life history stages of lionfish are poorly known. Mito and Uchida (1958) and Fishelson (1975) describe the development and early larval stages of congenerics, while Imamura and Yabe (1996) describe five *P. volitans* larvae collected in the water column off of northwestern Australia. Lionfish settle from the water column to benthic habitats at about 10-12 mm. Laidig and Sakuma (1998) reported a larval growth rate of 0.3 mm d-1 for *Scorpaena*, a genus in the same family as lionfish, Scorpeanidae. Using this growth rate, the estimated planktonic larval duration (PLD) of lionfish is 25 to 40 d, which means that larvae may be in the water column and susceptible to transport by ocean currents for approximately one month. However, confirmation of PLD specific to *P. volitans* is needed as PLD can vary widely, even within members of the same genus (Victor 1986).

In 2004, a total of 149 lionfish were collected off North Carolina for life history analyses, These ranged in length from 5 to 45 cm (average length = 30.5 cm) and in weight from 25 to 1380 grams (3 lbs) with average wt of 480 grams. Several lionfish collected in this study were larger (45 cm) than the reported maximum length from their native range (38 cm) (Schultz 1986; Randall et al. 1997; Myers 1999), suggesting that lionfish growth along the southeast U.S. is not resource limited (Elton 1958). The growth rate of lionfish in the Atlantic or in their native habitat remains unknown.

Although preliminary, analyses of annual zones on sagittal otoliths suggest that the lionfish population off North Carolina is relatively young, (max. age 7 years old; 43 cm specimen). If confirmed, these results would support our general timeline of the invasion which we believe began around the year 2000, off North Carolina. However, age validation is still required to confirm this result.
As in most reef fishes, the major dispersal phase of lionfish probably occurs while eggs and larvae are in the plankton. The northward dispersal (i.e., from Florida to NC) of lionfish is thought to be greatly facilitated by the strong northerly flowing Gulf Stream currents. Dispersal further into the northeast is most likely facilitated by Gulf Stream eddies (e.g., cross shelf transport, Hare and Cowen 1996). Once settled to the benthos, observations from their native habitat suggest that lionfish exhibit site fidelity and do not migrate (Fishelson 1975, 1997; McBride and Able,1998.) In the Atlantic, however, the question of lionfish movement or migration, especially in response to cold water incursions, remains an important area of research

but to date is unknown. If lionfish did move offshore in the winter in response to cold bottom water temperatures, this may increase their ability to survive thereby decreasing their natural mortality.

Genetics analyses of the Atlantic lionfish specimens revealed the presence of two closely related sister species *Pterois volitans* and *P. miles* within the Atlantic but 93.5% of collected specimens were *P. volitans*. We also found that the complexity of the haplotype network for Atlantic specimens was greatly simplified when compared to specimens in their native range. Twenty-eight different haplotypes were found within 43 native range *P. volitans* as opposed to 3 haplotypes within the 160 Atlantic *P. volitans*

specimens. In addition, 95% of the Atlantic *P. volitans* shared the same haplotype. These data indicate a large decrease in genetic diversity within the Atlantic population most likely caused by a small founder population, but of no less than 3 female specimens. These data may indicate that a small release in the right environment can result in an invasion of impressive proportions.

Ecological relationships – Potential Impact

Within their native habitat the ecology of lionfish is not well known. A few studies on lionfish found they consumed a wide variety of smaller fishes, shrimps and crabs (Fishelson 1975), and occupy the upper levels of the food chain (Fishelson 1997). Moreover, few predators of lionfish have been reported in their native range (but see, Bernadsky and Goulet 1991; Moyer and Zaiser 1981). Although, potential lionfish predators along the southeast United States have no experience with the venomous spines of the lionfish (Ray and Coates 1958; Halstead 1967) there are other native venomous fishes such as scorpionfishes (same family as lionfish) which are consumed by native predatory fishes (Randall 1967; Ebert et al. 1991; Roel and Macpherson 1998; Bowman et al. 2000). However, the potential role of predation in decreasing the number of lionfish is unknown, as is the effect of lionfish on predators.

Lionfish could impact native ecosystems through direct predation, competition and overcrowding. Preliminary data on the diet of Atlantic lionfish specimens suggest that they are primarily generalist piscivores, similar to their native counterparts. The Atlantic lionfish diet is comprised mainly of prey from a variety of fish families including members of the Serranidae, Pomacentridae, Labridae, Scaridae, Blenniidae, Bothidae, Carangidae, and Monacanthidae. Ninety eight percent of stomachs examined contained fishes, and other prey items (decapod crustaceans, cephalopod and bivalve mollusks) make up only a fraction of prey contents by volume (approx. 0.5 % or less). The small serranids (sea basses) were substantially more important in terms of volume than other families of fishes (41% vs. 15% and lower for other prey families) (Munoz et al. in prep). Since lionfish are opportunistic predators feeding primarily on smaller fishes, there is potential for trophic overlap with native fishes (Sano et al. 1984; Naughton 1985; Matheson et al. 1986; Fishelson 1997) such as groupers in the genus Mycteroperca. Groupers comprising this genus feed almost exclusively on fishes (Dodrill et al. 1993). In particular, gag (*Mycteroperca microlepis*) and scamp (*M. phenax*) groupers are present in significant numbers off the North Carolina coast and scamp occur at size classes that appear to overlap size classes of lionfish. Serranids form one of the most important food items in the scamp diet (Matheson et al. 1986) so similarly sized scamp and lionfish may be targeting similar prey. In addition, lionfish have been confirmed to prey upon scad (Carangidae), one of the dominant fish species in the diet of gag (Naughton & Saloman 1985). If these prey fishes are already or became a limiting resource, a growing lionfish population could negatively impact the scamp and gag populations via competition for food resources. The style of lionfish predation, (i.e., ambush predator) is not unique on southeast United States reefs and wrecks (e.g., red grouper, frog fish, scorpion fish), but the lack of experience of prey species may increase the predation efficiency of lionfish. Moreover, continued mortality of groupers and other native predators through overfishing (Huntsman et al. 1999; NMFS 2004) may open niche space and further increase resources for lionfish (Davis 2000).

Lionfish may also affect the use of habitat by other species through physical overcrowding and aggressive tendencies. Lionfish are often described as _standing their ground' and male-male aggression is extremely high prior to and during reproductive activities, during which lionfish will even threaten divers (Thresher 1984; Myers 1991). If this behavioral characteristic was extended towards other organisms in their introduced range, the threat might be expected to increase with lionfish abundance and potentially cause native species displacement into sub-optimum habitats (Schumacher and Parrish 2005; Taylor et al. 1984).
Abundance and status of the stock

The total population abundance of lionfish in the Atlantic is currently unknown. Quantitative surveys combined with public reports suggest the population is growing in number and increasing in geographic extent and may potentially colonize the entire Caribbean and Gulf of Mexico (Whitfield et al. 2006). Within the last two years quantitative surveys at the same nineteen locations off North Carolina (95 to 150 fsw) indicate that lionfish densities have doubled. Moreover, yearly surveys from the same nineteen locations, off North Carolina, suggest lionfish densities may be similar to many native fish species (i.e., *Cephalopholis cruentatus*, *Epinephelus guttatus*, *E. adscensonis*, *Mycteroperca interstitialis*, *M. microlepis*) (Whitfield et al. 2006). At this point there is every expectation that the total population and geographic extent of lionfish will continue to increase. More information is clearly needed to determine the status of the entire population, but traditional fishery sampling methods are not appropriate because lionfish are not captured effectively in this manner. More detailed information on the amount and type of benthic habitat within the southeast region combined with a random program of quantitative visual surveys over a broad geographic area (Bahamas to NC) will assist in estimating the total population size of lionfish.

Summary

The southeast United States continental shelf ecosystem is already undergoing change. Many important reef fish predators are overfished (Huntsman et al. 1999). In the Snapper-Grouper Management Unit of the South Atlantic Fisheries Management Council, approximately half of the stocks for which the status is known are classified as overfished. The reef fish fauna of the southeast United States continental shelf is also becoming more tropical (Parker and Dixon 1998). From the 1970's to the 1990's, the number of tropical species and the abundance of individual tropical species increased off the coast of North Carolina. Both of these large-scale changes favor the continued growth and dispersal of the lionfish population along the southeast United States. The effect of climate change, overfishing and invasive species have been implicated in ecosystem decline and collapse in several marine ecosystems, (Harris & Tyrrell 2001; Stachowicz et al. 2002; Frank et al. 2005). Along the southeast U.S. shelf the high number of stressors acting in synergism may eventually have unexpected and irreversible consequences for the native communities and economically valuable fisheries in this region.

6.1.3 Natural Events and Climate Change

Potential Threats to EFH from Natural Events and Climate Change

Potential threats: Coastal and inland storms can cause severe acute and chronic perturbations including habitat erosion, burial of habitat and organisms by sediment deposition; creation of strong currents that alter habitats and remove biota; damage by wind and waves; creation of turbidity levels that can cause physiological damage and disrupt feeding, spawning migration, and other vital processes; and abrupt changes in salinity and other water quality characteristics such as fecal coliform levels and harmful algal blooms. Long-term climatological changes, such as, changes in weather patterns and ocean currents, can bring about similar changes by increasing storm activity, changing fresh water inputs and salinity in coastal systems, increasing ocean acidification which affects coral reef building, and changing water column productivity that can affect certain fish population. For example, the Atlantic Multidecadal Oscillation can cause large scale ecological changes called regime shifts where temperature alterations favor or harm a particular species or group. Changes that cause relocation of frontal boundaries, weed lines, and stratification and temperature boundaries may also cause substantial and undesirable environmental change.

Coastal processes may be dramatically altered by natural events. These include short term events such as severe storms, hurricanes, floods, etc. Effects vary from potentially positive to catastrophic. For example, a moderate storm may provide needed freshwater, flush and recharge stagnant water bodies, and transfer nutrients from uplands and high marsh surfaces to tidal waters. On the other hand, shoreline erosion, wetlands destruction and subsidence and substantial changes in the structure of coral communities (e.g., Bythyell et al. 1993) are possible.

Hurricanes and other severe climatological events and change can drastically alter shorelines and associated environments including wetlands. Some changes may be positive such as the flushing of stagnant systems. However, wind induced erosion and overwash can remove and fill large areas of SAV and emergent wetlands. In overwash areas, newly created ―uplands‖ are often quickly developed and stabilized and geomorphological processes that lead to rebuilding of wetlands and shallow water areas may be precluded. As storm activity increases in severity and regularity, emergency shoreline protection response threatens coastal nearshore habitats primarily through burial by beach restoration efforts. Littoral sand drift has interrupted by the development of stabilized inlet jetties, which has reduced sand budgets. Decreased sand budgets coupled with increased severe storm activity (a known result of increased rates of global warming) necessitate an increase in large-scale beach dredge and fill projects. The direct, secondary and cumulative effects of these activities are known to have a profound effect on EFH through burial of nearshore hard bottom, worm reef, coral reef and sand bottom habitat areas. Loss of habitat areas utilized by various life stages of federally managed species and their prey species will continue to have a negative effect. As the need for such projects increases and the time between projects decreases adverse effects will be amplified. Hurricanes also cause vertical mixing in coastal waters that results in cooling and nutrient enrichment of surface water and stimulation of algal growth. In estuaries, hurricanes suspend sediment and increase terrestrial runoff that can result in algal blooms and hypoxia in bottom waters (NOAA 2005). Algal blooms and hypoxia can cause fish die-offs and spread disease to other plants and animals.

Climate Change
This section was excerpted from the *Summary Report for Policymakers* based on the assessment carried out by the three Working Groups of the Intergovernmental Panel on Climate Change (IPCC). It provides an integrated view of climate change as the final part of the IPCC's Fourth Assessment Report, released in fall 2007. A complete elaboration of the topics covered in this summary can be found in this Synthesis Report and in the underlying reports of the three Working Groups available online at (http://www.coastalclimate.org/).
Observed changes in climate and their effects
Warming of the climate system is unequivocal, as is now evident from observations of increases in global average air and ocean temperatures, widespread melting of snow and ice, and rising global average sea level (**Figure 6.1-2).**
Eleven of the last twelve years (1995-2006) rank among the twelve warmest years in the instrumental record of global surface temperature (since 1850). The 100-year linear trend (1906-2005) of 0.74 [0.56 to 0.92]°C 1 is larger than the corresponding trend of 0.6 [0.4 to 0.8]°C (1901-2000) given in the Third Assessment Report (TAR) (**Figure 6.1-2**). The temperature increase is widespread over the globe, and is greater at higher northern latitudes.
Land regions have warmed faster than the oceans (**Figure 6.1-3**). Rising sea level is consistent with warming (**Figure 6.1-2**). Global average sea level has risen since 1961 at an average rate of 1.8 [1.3 to 2.3]mm/yr and since 1993 at 3.1 [2.4 to 3.8]mm/yr, with contributions from thermal expansion, melting glaciers and ice caps, and the polar ice sheets. Whether the faster rate for 1993 to 2003 reflects decadal variation or an increase in the longer-term trend is unclear.
Observed decreases in snow and ice extent are also consistent with warming (Figure SPM.1). Satellite data since 1978 show that annual average Arctic sea ice extent has shrunk by 2.7% (2.1 to 3.3) per decade, with larger decreases in summer of 7.4% (5.0 to 9.8) per decade. Mountain glaciers and snow cover on average have declined in both hemispheres.
From 1900 to 2005, precipitation increased significantly in eastern parts of North and South America, northern Europe and northern and central Asia but declined in the Sahel, the Mediterranean, southern Africa and parts of southern Asia. Globally, the area affected by drought has likely increased since the 1970s. Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 82
It is very likely that over the past 50 years: cold days, cold nights and frosts have become less frequent over most land areas, and hot days and hot nights have become more frequent. It is likely that: heat waves have become more frequent over most land areas, the frequency of heavy precipitation events has increased over most areas, and since
1975 the incidence of extreme high sea level has increased worldwide.
There is observational evidence of an increase in intense tropical cyclone activity in the North Atlantic since about 1970, with limited evidence of increases elsewhere. There is no clear trend in the annual numbers of tropical cyclones. It is difficult to ascertain longer-term trends in cyclone activity, particularly prior to 1970.
Average Northern Hemisphere temperatures during the second half of the 20th century were very likely higher than during any other 50-year period in the last 500 years and likely the highest in at least the past 1300 years. Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 83
Figure 6.1-2. Observed changes in (a) global average surface temperature; (b) global average sea level from tide gauge (blue) and satellite (red) data and (c) Northern

Hemisphere snow cover for March-April. All differences are relative to corresponding averages for the period 1961-1990. Smoothed curves represent decadal averaged values while circles show yearly values. The shaded areas are the uncertainty intervals estimated from a comprehensive analysis of known uncertainties (a and b) and from the time series (c). Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 84 Observational evidence from all continents and most oceans shows that many natural systems are being affected by regional climate changes, particularly temperature increases. Changes in snow, ice and frozen ground have with high confidence increased the number and size of glacial lakes, increased ground instability in mountain and other permafrost regions, and led to changes in some Arctic and Antarctic ecosystems.

There is high confidence that some hydrological systems have also been affected through increased runoff and earlier spring peak discharge in many glacier- and snow-fed rivers, and effects on thermal structure and water quality of warming rivers and lakes.

In terrestrial ecosystems, earlier timing of spring events and poleward and upward shifts in plant and animal ranges are with very high confidence linked to recent warming. In some marine and freshwater systems, shifts in ranges and changes in algal, plankton and fish abundance are with high confidence associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels and circulation.

Of the more than 29,000 observational data series, from 75 studies, that show significant change in many physical and biological systems, more than 89% are consistent with the direction of change expected as a response to warming. However, there is a notable lack of geographic balance in data and literature on observed changes, with marked scarcity in developing countries.

There is medium confidence that other effects of regional climate change on natural and human environments are emerging, although many are difficult to discern due to adaptation and non-climatic drivers. They include effects of temperature increases on:

agricultural and forestry management at Northern Hemisphere higher latitudes, such as earlier spring planting of crops, and alterations in disturbance regimes of forests due to fires and pests

some aspects of human health, such as heat-related mortality in Europe, changes in infectious disease vectors in some areas, and allergenic pollen in Northern Hemisphere high and mid-latitudes

some human activities in the Arctic (e.g., hunting and travel over snow and ice) and in lower-elevation alpine areas (such as mountain sports).

Causes of change

Changes in atmospheric concentrations of greenhouse gases (GHGs) and aerosols, land-cover and solar radiation alter the energy balance of the climate system.

Global GHG emissions due to human activities have grown since pre-industrial times, with an increase of 70% between 1970 and 2004. Carbon dioxide (CO2) is the most important anthropogenic GHG. Its annual emissions grew by about 80% between 1970 and 2004. The long-term trend of declining CO2 emissions per unit of energy supplied reversed after 2000. Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 85 Global atmospheric concentrations of CO2, methane (CH4) and nitrous oxide (N2O) have increased markedly as a result of human activities since 1750 and now far exceed pre-industrial values determined from ice cores spanning many thousands of years.

Atmospheric concentrations of CO2 (379ppm) and CH4 (1774 ppb) in 2005 exceed by far the natural range over the last 650,000 years. Global increases in CO2 concentrations are due

primarily to fossil fuel use, with land-use change providing another significant but smaller contribution. It is very likely that the observed increase in CH4 concentration is predominantly due to agriculture and fossil fuel use. Methane growth rates have declined since the early 1990s, consistent with total emissions (sum of anthropogenic and natural sources) being nearly constant during this period. The increase in N2O concentration is primarily due to agriculture.
There is very high confidence that the net effect of human activities since 1750 has been one of warming.
Most of the observed increase in globally-averaged temperatures since the mid-20th century is very likely due to the observed increase in anthropogenic GHG concentrations. It is likely there has been significant anthropogenic warming over the past 50 years averaged over each continent (except Antarctica) (Figure SPM.4).
During the past 50 years, the sum of solar and volcanic forcings would likely have produced cooling. Observed patterns of warming and their changes are simulated only by models that include anthropogenic forcings. Difficulties remain in simulating and attributing observed temperature changes at smaller than continental scales. Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 86

Figure 6.1-3. Comparison of observed continental- and global-scale changes in surface temperature with results simulated by climate models using either natural or both natural and anthropogenic forcings. Decadal averages of observations are shown for the period 1906-2005 (black line) plotted against the centre of the decade and relative to the corresponding average for the period 1901-1950. Lines are dashed where spatial coverage is less than 50%. Blue shaded bands show the 5-95% range for 19 simulations from 5 climate models using only the natural forcings due to solar activity and volcanoes. Red shaded bands show the 5-95% range for 58 simulations from 14 climate models using both natural and anthropogenic forcings.
Advances since the TAR show that discernible human influences extend beyond average temperature to other aspects of climate.
Human influences have:
very likely contributed to sea level rise during the latter half of the 20th century
likely contributed to changes in wind patterns, affecting extra-tropical storm tracks and temperature patterns
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likely increased temperatures of extreme hot nights, cold nights and cold days
more likely than not increased risk of heat waves, area affected by drought since the 1970s and frequency of heavy precipitation events.

Anthropogenic warming over the last three decades has likely had a discernible influence at the global scale on observed changes in many physical and biological systems. Spatial agreement between regions of significant warming across the globe and locations of significant observed changes in many systems consistent with warming is very unlikely to be due solely to natural variability. Several modeling studies have linked some specific responses in physical and biological systems to anthropogenic warming.
More complete attribution of observed natural system responses to anthropogenic warming is currently prevented by the short time scales of many impact studies, greater natural climate

variability at regional scales, contributions of non-climate factors and limited spatial coverage of studies.
Projected climate change and its impacts
There is high agreement and much evidence that with current climate change mitigation policies and related sustainable development practices, global GHG emissions will continue to grow over the next few decades.
The IPCC Special Report on Emission Scenarios (SRES, 2000) projects an increase of global GHG emissions by 25-90% (CO2-eq) between 2000 and 2030, with fossil fuels maintaining their dominant position in the global energy mix to 2030 and beyond. More recent scenarios without additional emissions mitigation are comparable in range.
Continued GHG emissions at or above current rates would cause further warming and induce many changes in the global climate system during the 21st century that would very likely be larger than those observed during the 20th century. For the next two decades a warming of about 0.2°C per decade is projected for a range of SRES emission scenarios. Even if the concentrations of all GHGs and aerosols had been kept constant at year 2000 levels, a further warming of about 0.1°C per decade would be expected. Afterwards, temperature projections increasingly depend on specific emission scenarios.
For an explanation of SRES emission scenarios, see Box _SRES scenarios' in Topic 3 of this Synthesis Report. These scenarios do not include additional climate policy above current ones; more recent studies differ with respect to UNFCCC and Kyoto Protocol inclusion.
There is now higher confidence than in the TAR in projected patterns of warming and other regional-scale features, including changes in wind patterns, precipitation, and some aspects of extremes and sea ice. Regional-scale changes include:
warming greatest over land and at most high northern latitudes and least over Southern Ocean and parts of the North Atlantic Ocean, continuing recent observed trends **(Figure 6.1-4)**;

contraction of snow cover area, increases in thaw depth over most permafrost regions, and decrease in sea ice extent; in some projections using SRES scenarios, Arctic late-summer sea ice disappears almost entirely by the latter part of the 21st century;
very likely increase in frequency of hot extremes, heat waves, and heavy precipitation;
likely increase in tropical cyclone intensity; less confidence in global decrease of tropical cyclone numbers;
poleward shift of extra-tropical storm tracks with consequent changes in wind, precipitation, and temperature patterns; and
very likely precipitation increases in high latitudes and likely decreases in most subtropical land regions, continuing observed recent trends.

There is high confidence that by mid-century, annual river runoff and water availability are projected to increase at high latitudes (and in some tropical wet areas) and decrease in some dry regions in the mid-latitudes and tropics. There is also high confidence that many semi-arid areas (e.g., Mediterranean basin, western United States, southern Africa and northeast Brazil) will suffer a decrease in water resources due to climate change.

Figure 6.1-4. Projected surface temperature changes for the late 21st century (2090- 2099). The map shows the multi- AOGCM average projection for the A1B SRES scenario. All temperatures are relative to the period 1980-1999.
Studies since the TAR have enabled more systematic understanding of the timing and magnitude of impacts related to differing amounts and rates of climate change. Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 89
Figure 6.1-5. Illustrative examples of global impacts projected for climate changes (and sea level and atmospheric CO2 where relevant) associated with different amounts of increase in global average surface temperature in the 21st century.
The black lines link impacts; broken line arrows indicate impacts continuing with increasing temperature. Entries are placed so that the left hand side of text indicates the approximate level of warming that is associated with the onset of a given impact. Quantitative entries for water scarcity and flooding represent the additional impacts of climate change relative to the conditions projected across the range of SRES model scenarios. Adaptation to climate change is not included in these estimations. *Confidence levels for all statements are high.*
Examples of some projected regional impacts in North America:
 Warming in western mountains is projected to cause decreased snowpack, more winter flooding, and reduced summer flows, exacerbating competition for over-allocated water resources.
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 In the early decades of the century, moderate climate change is projected to increase aggregate yields of rain-fed agriculture by 5-20%, but with important variability among regions. Major challenges are projected for crops that are near the warm end of their suitable range or which depend on highly utilized water resources.
 During the course of this century, cities that currently experience heat waves are expected to be further challenged by an increased number, intensity and duration of heat waves during the course of the century, with potential for adverse health impacts.
 Coastal communities and habitats will be increasingly stressed by climate change impacts interacting with development and pollution.

Moreover, some systems, sectors and regions are likely to be especially affected by climate change.
 Particular ecosystems
o terrestrial: tundra, boreal forest and mountain regions because of sensitivity to warming; Mediterranean-type ecosystems because of reduction in rainfall; and tropical rainforests where precipitation declines
o coastal: mangroves and salt marshes, due to multiple stresses
o marine: coral reefs due to multiple stresses; the sea ice biome because of sensitivity to warming
 Water resources in some dry regions at mid-latitudes and in the dry tropics, due to changes in rainfall and evapotranspiration, and in areas dependent on snow and ice melt agriculture in low-latitudes, due to reduced water availability.
 Low-lying coastal systems, due to threat of sea level rise and increased risk from extreme weather events.
 Human health in populations with low adaptive capacity.

Altered frequencies and intensities of extreme weather, together with sea level rise, are expected to have mostly adverse effects on natural and human systems.
Examples for selected extremes and sectors are shown in **Table 6.1-3.** These do not take into account any changes or developments in adaptive capacity. The likelihood estimates in column two relate to the phenomena listed in column one. Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 91

Table 6.1-3. Examples of possible impacts of climate change due to changes in extreme weather and climate events, based on projections to the mid- to late 21st century.
Notes:
a) See WGI Table 3.7 for further details regarding definitions.
b) Warming of the most extreme days and nights each year.
c) Extreme high sea level depends on average sea level and on regional weather systems. It is defined as the highest 1% of hourly values of observed sea level at a station for a given reference period.
d) In all scenarios, the projected global average sea level at 2100 is higher than in the reference period. The effect of changes in regional weather systems on sea level extremes has not been assessed.
Anthropogenic warming could lead to some impacts that are abrupt or irreversible, depending upon the rate and magnitude of the climate change. Partial loss of ice sheets on polar land could imply meters of sea level rise, major changes in coastlines and inundation of low-lying areas, with greatest effects in river deltas and low-lying islands. Such changes are projected to occur over millennial time scales, but more rapid sea level rise on century time scales cannot be excluded. Climate change is likely to lead to some irreversible impacts. There is medium confidence that approximately 20-30% of species assessed so far are likely to be at increased risk of extinction if increases in global average warming exceed 1.5-2.5°C (relative to 1980-1999). As global average temperature increase exceeds about 3.5°C, model projections suggest significant extinctions (40-70% of species assessed) around the globe.

Based on current model simulations, the meridional overturning circulation (MOC) of the Atlantic Ocean will very likely slow down during the 21st century; nevertheless temperatures over the Atlantic and Europe are projected to increase. The MOC is very unlikely to undergo a large abrupt transition during the 21stcentury. Longer-term MOC changes cannot be assessed with confidence. Impacts of large-scale and persistent changes in the MOC are likely to include changes in marine ecosystem productivity, fisheries, ocean CO2 uptake, oceanic oxygen concentrations and terrestrial vegetation. Changes in terrestrial and ocean CO2 uptake may feedback on the climate system.

The five ―reasons for concern‖ identified originally in the IPCC‗s Third Assessment Report (TAR) remain a viable framework to consider key vulnerabilities. These ―reasons‖ are assessed here to be stronger than in the TAR. Many risks are identified with higher confidence. Some risks are projected to be larger or to occur at lower increases in temperature.
Understanding about the relationship between impacts (the basis for ―reasons for concern‖ in the TAR) and vulnerability (that includes the ability to adapt to impacts) has improved. This is due to more precise identification of the circumstances that make systems, sectors and regions

especially vulnerable, and growing evidence of the risks of very large impacts on multiple century time scales.

Risks to unique and threatened systems. There is new and stronger evidence of observed impacts of climate change on unique and vulnerable systems (such as polar and high mountain communities and ecosystems), with increasing levels of adverse impacts as temperatures increase further. An increasing risk of species extinction and coral reef damage is projected with higher confidence than in the TAR as warming proceeds. There is medium confidence that approximately 20-30% of plant and animal species assessed so far are likely to be at increased risk of extinction if increases in global average temperature exceed 1.5-2.5°C over 1980-1999 levels. Confidence has increased that a 1-2°C increase in global mean temperature above 1990 levels (about 1.5-2.5°C above pre-industrial) poses significant risks to many unique and threatened systems including many biodiversity hotspots. Corals are vulnerable to thermal stress and have low adaptive capacity. Increases in sea surface temperature of about 1-3°C are projected to result in more frequent coral bleaching events and widespread mortality, unless there is thermal adaptation or acclimatization by corals. Increasing vulnerability of indigenous communities in the Arctic and small island communities to warming is projected.
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Risks of extreme weather events. Responses to some recent extreme events reveal higher levels of vulnerability than the TAR. There is now higher confidence in the projected increases in droughts, heat waves, and floods as well as their adverse impacts.

Distribution of impacts and vulnerabilities. There are sharp differences across regions and those in the weakest economic position are often the most vulnerable to climate change. There is increasing evidence of greater vulnerability of specific groups such as the poor and elderly in not only developing but also developed countries. Moreover, there is increased evidence that low-latitude and less-developed areas generally face greater risk, for example in dry areas and megadeltas.

Aggregate impacts. Compared to the TAR, initial net market-based benefits from climate change are projected to peak at a lower magnitude of warming, while damages would be higher for larger magnitudes of warming. The net costs of impacts of increased warming are projected to increase over time.

Risks of large-scale singularities. There is high confidence that global warming over many centuries would lead to a sea level rise contribution from thermal expansion alone which is projected to be much larger than observed over the 20th century, with loss of coastal area and associated impacts. There is better understanding than in the TAR that the risk of additional contributions to sea level rise from both the Greenland and possibly Antarctic ice sheets may be larger than projected by ice sheet models and could occur on century time scales. This is because ice dynamical processes seen in recent observations but not fully included in ice sheet models assessed in AR4 could increase the rate of ice loss.

There is high confidence that neither adaptation nor mitigation alone can avoid all climate change impacts; however, they can complement each other and together can significantly reduce the risks of climate change.

Adaptation is necessary in the short and longer term to address impacts resulting from the warming that would occur even for the lowest stabilization scenarios assessed. There are barriers, limits and costs, but these are not fully understood. Unmitigated climate change would, in the long term, be likely to exceed the capacity of natural, managed and human systems to adapt. The time at which such limits could be reached will vary between sectors and regions. Early mitigation actions would avoid further locking in carbon intensive infrastructure and reduce climate change and associated adaptation needs.

Many impacts can be reduced, delayed or avoided by mitigation. Mitigation efforts and investments over the next two to three decades will have a large impact on opportunities to achieve lower stabilization levels.

Delayed emission reductions significantly constrain the opportunities to achieve lower stabilization levels and increase the risk of more severe climate change impacts. In order to stabilize the concentration of GHGs in the atmosphere, emissions would need to peak and decline thereafter. The lower the stabilization level, the more quickly this peak and decline would need to occur.

Sea level rise under warming is inevitable. Thermal expansion would continue for many centuries after GHG concentrations have stabilized, for any of the stabilization levels assessed, causing an eventual sea level rise much larger than projected for the 21st century. The eventual contributions from Greenland ice sheet loss could be several meters, and larger than from thermal expansion, should warming in excess of 1.9-4.6°C above pre-industrial be sustained over many centuries. The long time scales of thermal expansion and ice sheet response to warming imply that stabilization of GHG concentrations at or above present levels would not stabilize sea level for many centuries.

Ocean Acidification

another global change issue relates to changes in the earth‘s carbon budget and cycle. Carbon cycles through the earth‘s ecosystems in organic and inorganic forms. Recent increasing trends in carbon dioxide in the earth‘s atmosphere is shifting the cycle of carbon in the ocean and increasing carbonic acid and a gradual decrease in ocean pH and calcium carbonate. Experimental evidence suggests that if these trends continue, key marine organisms, such as corals and some plankton, will have difficulty maintaining their external calcium carbonate skeletons (Orr et al. 2005).

According to the Intergovernmental panel on Climate Change (2007), the uptake of anthropogenic carbon since 1750 has led to the ocean becoming more acidic with an average decrease in pH of 0.1 units. Increasing atmospheric CO2 concentrations lead to further acidification. Projections based on SRES scenarios give a reduction in average global surface ocean pH of between 0.14 and 0.35 units over the 21st century. While the effects of observed ocean acidification on the marine biosphere are as yet undocumented, the progressive acidification of oceans is expected to have negative impacts on marine shell-forming organisms (e.g., corals) and their dependent species.

6.2 Adverse impacts of fishing activities under South Atlantic Council Fishery Management Plans

(excerpted from Barnette 2001)
All fishing has an effect on the marine environment, and therefore the associated habitat. Fishing has been identified as the most widespread human exploitative activity in the marine environment (Jennings and Kaiser 1998), as well as the major anthropogenic threat to demersal fisheries habitat on the continental shelf (Cappo et al. 1998). Fishing impacts range from the extraction of a species which skews community composition and diversity to reduction of habitat complexity through direct physical impacts of fishing gear.

The nature and magnitude of the effects of fishing activities depend heavily upon the physical and biological characteristics of a specific area in question. There are strict limitations on the degree to which probable local effects can be inferred from the studies Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 95
of fishing practices conducted elsewhere (North Carolina Division of Marine Fisheries 1999). The extreme variability that occurs within marine habitats confounds the ability to easily evaluate habitat impacts on a regional basis. Obviously, observed impacts at coastal or nearshore sites should not be extrapolated to offshore fishing areas because of the major differences in water depth, sediment type, energy levels, and biological communities (Prena et al. 1999). Marine communities that have adapted to highly dynamic environmental conditions (e.g., estuaries) may not be affected as greatly as those communities that are adapted to stable environmental conditions (e.g., deep water communities). While recognizing the pitfalls that are associated with applying the results of gear impact studies from other geographical areas, due to the lack of sufficient and specific information within the Southeast Region it is necessary to review and carefully interpret all available literature in hopes of improving regional knowledge and understanding of fishery-related habitat impacts.

In addition to the environmental variability that occurs within the regions, the various types of fishing gear and how each is utilized on various habitat types affect the resulting potential impacts. For example, trawls vary in size and weight, as well as their impacts to the seabed. Additionally, the intensity of fishing activities needs to be considered. Whereas a single incident may have a negligible impact on the marine environment, the cumulative effect may be much more severe. Within intensively fished grounds, the background levels of natural disturbance may have been exceeded, leading to long-term changes in the local benthic community (Jennings and Kaiser 1998). Collie (1998) suggested that, to a large extent, it is the cumulative impact of bottom fishing, rather than the characteristics of a particular gear, that affects benthic communities. Unfortunately, a limitation to many fishing-related impact studies is that they do not measure the long term effects of chronic fishing disturbance. Furthermore, one of the most difficult aspects of estimating the extent of fishing impacts on habitat is the lack of high-resolution data on the distribution of fishing effort (Auster and Langton 1999).

The effects of fishing can be divided into short-term and long-term impacts. Short-term impacts (e.g., sediment resuspension) are usually directly observable and measurable while long-term impacts (e.g., effects on biodiversity) may be indirect and more difficult to quantify. Even more difficult to assess would be the cascading effects that fishery-related impacts may have on the marine environment. Additionally, various gears may indirectly impact EFH. Bycatch disposal and ghost fishing are two of the more well-documented indirect impacts to EFH. While recognizing that these are serious issues that pertain to habitat, this review does not attempt to discuss these due to the secondary nature of the impacts.

The majority of existing gear impact studies focus on mobile gear such as trawls and dredges. On a regional scale, mobile gear such as trawls impact more of the benthos than any other gear. However, other fishing practices may have a more significant ecological effect in a particular area due to the nature of the habitat and fishery. Yet there are few studies that investigate other gear types, especially static gear. Rogers et al. (1998) stated that there are few accounts of the physical contact of static gear having measurable effects on benthic biota, as the area of sea bed affected by each gear is almost insignificant compared to the widespread effects of mobile gear. Regardless, static gear may negatively affect EFH and, therefore, must be considered.

The exact relationship that particular impacts have on the associated community and productivity is not fully understood. While it is clear that fishing activities impact or alter EFH, the result of those impacts or the degree of habitat alteration that still allow for sustainable fishing is unknown (Dayton et al. 1995; Auster et al. 1996; Watling and Norse 1998). Hall (1994) noted that not all impacts are negative. A negative effect at one level may sometimes be viewed as a positive effect at a higher level of biological organization – particular species may be removed in small-scale disturbances yet overall community diversity at the regional scale may rise because disturbance allows more species to coexist.

6.2.1 Fishing Gear Regulations under Council FMPs

The following is a list of gear currently in use (or regulated) in fisheries managed under the South Atlantic Council fishery management plans. In general, if gear is not listed it is prohibited or not commonly used in the fishery:

Snapper Grouper Fishery

Vertical hook-and-line gear, including hand-held rod and manual or electric reel or ―bandit gear‖ with manual, electric or hydraulic reel (recreational and commercial).

Spear fishing gear without rebreathers (recreational and commercial).

Powerheads, except where expressly prohibited in Special Management Zones (SMZs). In addition, the use of explosive charges, including powerheads, is prohibited in the EEZ off South Carolina (recreational and commercial).

Bottom longlines (commercial). Prohibited south of a line running east of St. Lucie Inlet, Florida (27° 10‘ N. lat.) and in depths less than 50 fathoms north of that line. May not be used to fish for wreckfish.

Sea bass pots (commercial). May not be used or possessed in multiple configurations. Pot size, wire mesh size and construction restrictions. May not be used in the EEZ south of a line running due east of the NASA Vehicle Assembly Building, Cape Canaveral, Florida (28° 35.1‘ N. lat.).

Special Management Zones (created under the Snapper Grouper FMP). Sea bass pots are prohibited in all Special Management Zones. Fishing may only be conducted with hand-held

hook-and-line gear (including manual, electric, or hydraulic rod and reel) and spearfishing gear in specified Special Management Zones; however, in other specified Special Management Zones a hydraulic or electric reel that is permanently affixed to a vessel (―bandit gear‖) and/or spear fishing gear (or only powerheads) are prohibited.
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Penaeid shrimp trawls (commercial). The Shrimp Fishery Management Plan allows North and South Carolina, Georgia and east Florida to request a closure in federal waters adjacent to closed state waters for brown, pink or white shrimp following severe cold weather that results in an 80% or greater reduction in the population of white shrimp (whiting, royal red and rock shrimp fisheries are exempt from a federal closure for white shrimp). During a federal closure, a buffer zone is established extending seaward from shore to 25 nautical miles, inside of which no trawling is allowed with a net having less than 4" stretch mesh. Vessels trawling inside this buffer zone cannot have a shrimp net aboard (i.e., a net with less than 4" stretch mesh) in the closed portion of the federal zone. Transit of the closed federal zone with less than 4" stretch mesh aboard while in possession of penaeid (white, brown and pink) species will be allowed provided that the nets are in an unfishable condition, which is defined as stowed below deck. Specified areas are closed to trawling for rock shrimp.

Rock shrimp trawls (commercial). The minimum mesh size for the cod end of a rock shrimp trawl net in the South Atlantic EEZ off Georgia and Florida is 1-7/8 inches (4.8 cm), stretched mesh. This minimum mesh size is required in at least the last 40 meshes forward of the cod end drawstring (tie off strings), and smaller mesh bag liners are not allowed. A vessel that has a trawl net on board that does not meet these requirements may not possess a rock shrimp in or from the South Atlantic EEZ off Georgia and Florida.

Bycatch Reduction Devices (BRDs). On a penaeid shrimp trawler in the South Atlantic EEZ, each trawl net that is rigged for fishing and has a mesh size less than 2.5", as measured between the centers of opposite knots when pulled taut, and each try net that is rigged for fishing and has a headrope length longer than 16.0 ft. must have a certified BRD installed. The following BRDs are certified for use by penaeid shrimp trawlers in the South Atlantic EEZ: extended funnel, expanded mesh and fisheye.
As of January 12, 2007, on a vessel that fishes for or possesses rock shrimp in the South Atlantic EEZ, each trawl net or try net that is rigged for fishing must have a certified BRD installed.
Turtle Excluder Devices (TEDs). TEDs are required for the penaeid and rock shrimp fisheries.

Red Drum
No harvest or possession is allowed in or from the EEZ (no gear specified).

Golden Crab
Crab traps (commercial). May not be fished in water depths less than 900 feet in the northern zone and 700 feet in the middle and southern zones. Rope is the only allowable material for mainlines and buoy line. Max. trap size equals 64 cubic feet in volume in the
Northern zone and 48 cubic feet in volume in the Mid and Southern zones. Traps must have at least 2 escape gaps or rings and an escape panel. Traps must be identified with a permit number.

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Coral, Coral Reefs, and Live/Hard Bottom Habitat

Hand harvest only for allowable species (recreational and commercial). A toxic chemical may not be used or possessed in a coral area in the EEZ. A power-assisted tool may not be used to take prohibited coral, allowable octocoral or live rock.

Oculina Bank Habitat Area of Particular Concern: Fishing with bottom longlines, bottom trawls, dredges, pots or traps is prohibited. Fishing vessels may not anchor, use an anchor and chain, or use a grapple and chain.

Coastal Migratory Pelagics

Hook and line gear, usually rod and reel or bandit gear, hand lines, flat lines etc. (recreational and commercial).

Run-around gillnets or sink nets (commercial). A gillnet must have a float line less than 1,000 yards in length to fish for coastal migratory pelagic species. Gillnets must be at least 4-3/4 inch stretch mesh.

Purse seines for other coastal migratory species (commercial) with an incidental catch allowance for Spanish mackerel (10%) and king mackerel (1%).

For Atlantic king mackerel (commercial) north of the Cape Lookout, NC Light (34° 37.3‘ N. lat.) all gear is authorized except for drift gillnets and long gillnets. South of the Cape Lookout Light the following gear is authorized: **automatic reel, bandit gear, handline, rod & reel.**

For Spanish mackerel (commercial) **automatic reel, bandit gear, handline, rod & reel, cast net, run around gill net and stab net**. Minimum size of 3.5‖ stretch mesh required for all run around gill nets.

Spiny Lobster

Traps, hand harvest, dip nets and bully nets (recreational and commercial). No poisons or explosives are allowed. No spear, hooks or piercing devices are allowed.

A degradable panel is required on non-wooden traps. Traps may not be tended at night. Buoy and trap identification is required.

Dolphin and Wahoo

Pelagic longline, hook and line gear including manual, electric, or hydraulic rod and reels, bandit gear, handline and spearfishing gear (including powerheads). Surface and pelagic longline gear for dolphin and wahoo is prohibited within any ―time area closure‖ in the Atlantic EEZ which is closed to the use of pelagic gear for highly migratory pelagic species (HMS) (commercial).

Sargassum

Nets used to harvest Sargassum be constructed of 4‖ stretch mesh or larger fitted to a frame no larger than 4 x 6 feet. Fishery Ecosystem Plan of the South Atlantic Region Volume IV Threats and Recommendations 99

6.2.2 Gear Descriptions

6.2.2.1 Mobile Gear

(excerpted from Barnette 2001)

Crab Scrape

A crab scrape is composed of a net bag attached to a rigid frame with short teeth (Figure 1). This gear, used exclusively in state waters, is dragged in the shallow water of bays and estuaries to catch crabs. There are no studies available that document potential damage to habitat. However, due to their design, their use in SAV would likely result in the potential uprooting of some plants, as well as leaf shearing (Barnette, personal observations). However, crab scrapes are not typically employed in vegetated areas due to the amount of plant litter that would fill the net. Penetration of the benthos by the teeth would result in sediment resuspension.

Frame Trawl

Roller frame trawls are primarily utilized to harvest bait shrimp in the State of Florida. They consist of a frame that holds open a net and supports slotted rollers that grip the bottom and turn freely. This motion prevents the scouring and scraping impacts primarily associated with otter trawls. Because participants in the fishery usually operate in shallow water, 9.14 m (30 ft) or less, frame trawls are typically limited to state waters.

A study by Futch and Beaumariage (1965) found that while frame trawls gathered large amounts of unattached algae and deciduous *Thalassia testudinium* leaves, no SAV with roots attached were found in the trawl catch.

Trawls with larger rollers (20.3cm; 8 in diameter) reduced the amount of bycatch material, with most drags uprooting SAV. Damage to SAV beds was noted on several occasions when the boats ran aground. The study concluded that side frame trawls do negligible damage to SAV beds. This conclusion was supported by Meyer et al. (1991; 1999), who found no significant trawl impacts on shoot density, structure, or biomass with increased trawling on turtlegrass (*Thalassia testudinium*). However, these studies did not evaluate the effects of repetitive trawling.

Woodburn et al. (1957) noted that the roller on the bottom of the trawl does cause the leaves ripe for shedding to break off, though this would not negatively impact the plant itself. Higman (1952) concluded that frame trawling is not sufficient to denude vegetated areas permanently or to damage the ecology of such locations. Additionally, Tabb and Kenny (1967), while not explicitly investigating habitat impacts, believed that roller frame trawls do no significant damage to habitat. In contrast to studies that assessed impacts to SAV, Tilmant (1979) found a high incidence of damage to stony corals in a study that investigated frame trawl impacts to hard bottom habitat in Biscayne Bay. Frame trawls turned over or crushed 80% of *Porites porites* and *Solenastrea hyades* and damaged over 50% of sponges and 38% of gorgonians in the trawl path. Macroalgae, including *Halimeda* and *Sargassum*, were heavily damaged. The primary impact on *Sargassum* was that it was torn loose from the bottom resulting in an early release to the free floating state. Tilmant (1979) found it doubtful that this action was harmful to *Sargassum* unless it occurred during early column formation. It was concluded that frame trawls have a significant impact on certain benthic organisms (Tilmant 1979).

Furthermore, within dense SAV communities, removal of epibenthic algae, tunicates, sponges, and other primary producers may also be significant. Futch and Beaumariage (1965) recommended that the diameter of the rollers be no less than 15.2cm (6in) and that the teeth of the rakes on the trawls should not extend below the roller. Furthermore, they recommend that boats employed in the frame trawl fishery that operate in shallow water should be of tunneled construction to prevent damage to SAV from propeller scarring. Tabb (1958) recommended that strainer bars should be rigid and aimed into the roller so that regardless of how far forward the net frame tips, the bars cannot dig into the bottom. The results from Tilmant (1979) indicated that extensive damage occurs to hardbottom habitat from frame trawls.

A logical recommendation that can be extrapolated from this study is the prohibition of frame trawling in areas where hardbottom habitat exists. Frame trawls, while causing negligible damage to SAV, are not compatible with hardbottom areas due to the damage it causes to complex vertical habitat (e.g., sponges, corals, gorgonians).

Prohibition on the use of bottom trawls

The use of trawl gear to harvest fish in the directed snapper grouper fishery south of Cape Hatteras, North Carolina (35°15' N. Latitude) and north of Cape Canaveral, Florida (Vehicle Assembly Building, 28°35.1' N. Latitude) is prohibited (SAFMC 1987). A vessel with trawl gear and more than 200 lbs of fish in the snapper grouper fishery on board will be defined as a directed fishery. The amendment also establishes a rebuttable presumption that a vessel with fish in the snapper grouper fishery on board harvested its catch of such fish in the Exclusive Economic Zone.

The Council based the trawl prohibition on habitat destruction and the desire to prevent overfishing of vermilion snapper. Fishes present in live bottom areas are described by Grimes et al. (1982) and include 113 species representing 43 families of predominantly tropical and subtropical fishes. Vermilion snapper were more abundant on the shelf edge than on the open shelf (Grimes et al. 1982). Miller and Richards (1980) described the distribution of live bottom habitat in the South Atlantic Bight and reported the most productive area of the shelf for commercial reef fish as being in the open shelf zone between 33 and 40 meters. Parker et al. (1983) reported on a survey of the areas from Cape Canaveral, Florida to Cape Fear, North Carolina and from Cape Fear to Cape Hatteras, North Carolina. From Cape Hatteras to Cape Fear 14,486 square km between 27 and 101 m were surveyed and contained 2,040 square km (14%) of reef habitat of which only 204 square km (10%) had one meter or more relief (distance from the highest point of the live bottom to the ocean floor). In the area from Cape Fear to Cape Canaveral, 24,826 square km between 27 and 101 m were surveyed and contained 7,403 square km (30%) of reef habitat of which 1,743 square km (7%) had one meter or more relief. The Oregon II cruise report (Anon. 1978) supports the scattered nature of live bottom in the South Atlantic from Cape Canaveral, Florida to Cape Hatteras, North Carolina. The Fishery Management Plan reported that in terms of the entire shelf area, current data suggest that from three to 30% of the shelf is suitable bottom for snapper grouper species (SAFMC 1983a).

The report on effects of a research trawl on live bottom (Van Dolah et al. 1987) documents that habitat damage does occur from the use of trawl gear even in the case of one pass through an area in a controlled study. The abstract is as follows:

―The effects of a research trawl on several sponge and coral species was assessed in a shallow-water, hard-bottom area located southeast of Savannah, Georgia. The study entailed a census of the numerically dominant species in replicate 25-square meter quadrants located along five transects established across a trawling alley. The density of undamaged sponges and corals was assessed in trawled and non-trawled (control) portions of each transect immediately before, immediately after, and 12 months after a 40/54 roller-rigged trawl was dragged through the alley once. Some damage to individuals of all target species was observed immediately after trawling, but only the density of barrel sponges (*Cliona* spp.) was significantly reduced. The extent of damage to the other sponges (*Ircinia campana, Haliclona oculata*), octocorals (*Leptogorgia virgulata, Lophogorgia hebes, Titanideum frauenfeldii*) and hard corals (*Oculina varicosa*) varied depending on the species, but changes in density were not statistically significant. Twelve months after trawling, the abundance of specimens counted in the trawled quadrants had increased to pre-trawl densities or greater, and damage to the sponges and corals could no longer be detected due to healing and growth. Trawl damage observed in this study was less severe than the damage reported for a similar habitat in a previous study. Differences between the two studies are attributed to (1) differences in the roller-rig design of the trawls used, and (2) differences in the number of times the same bottom was trawled.

The authors point out that in a study by Tilmant (1979) looking at the effects of commercial bait shrimping with roller-frame trawls in a shallow-water area of Biscayne Bay, Florida damage was much more severe: ―Tilmant observed severe damage (specimens crushed or torn loose) to more than 80% of the stony corals, 50% of the sponges and 38% of the soft corals along the trawl path.‖ It should be noted however, that this frame trawl consists of a solid, rectangular frame to which a net is attached and is used to fish grass bed areas; it was not designed to ―roll over‖ live bottom and would be expected to cause significant damage to corals, etc.

Importantly, habitat damage described by Van Dolah et al. (1987) resulted from one tow of trawl gear through the study area. That study was designed to evaluate the effects of a research trawl that does not typically cross the same bottom area more than once. Commercial trawling does not operate in this manner. Under commercial fishing conditions, a live bottom area would be fished over and over until the catches from such an area become unprofitable. Under such conditions, habitat damage would be expected to be much greater than is indicated from the above study. The *Oregon II* cruise report (Anon. 1978) indicated that drags with a trawl yielded a total catch of 476 lbs which included 424 lbs of finfish and 46 lbs of sponges and corals (10% of the total catch). This area was reported to have been on a mud bottom but turned out to be a low profile live bottom of sand ridges, clumps of sponges and scattered corals. Further indication of habitat damage is reported by Wenner (1983):

―The 3/4 Yankee trawl net effectively covers a much wider area of the bottom than the measured sweep (8.7 m) due to the configuration of the otter doors, ground cables, and bottom leg lines. Although this arrangement cannot increase the actual spread of the net beyond the headrope length, the passage of these cables over the substrate creates a disturbance that serves to herd fish in the path of the net (Baranov 1969). This net does, however, damage the sponge-coral habitat by shearing off sponges, soft corals, bryozoans, and other attached invertebrates. The 56 trawl tows made in the sponge-coral habitat for this study collected 2,351 kg of attached invertebrates (including sponges, soft corals, tunicates, bryozoans, and hydroids) yielding an average 42 kg/tow. This is only the amount of bottom material actually removed from the habitat. An estimate of the total amount of bottom destroyed by the doors, ground cables, and leg lines cannot be ascertained from the current study.

Personal observations and interviews with commercial fishermen attest to the productivity of the sponge-coral habitat. Most studies indicate the importance of habitat availability and space in determining the abundance and diversity of reef fishes (Emery 1978). With this in mind, and given the knowledge that 1) the use of the 3/4 Yankee trawl net reduces the amount of attached invertebrate growth (the amount damaged by doors and ground cables is presently not quantifiable); 2) the places where the invertebrates had been attached may be sanded over and rendered unsuitable for recolonization; and 3) the removal of these attached invertebrates reduces refugees for decapods, polychaetes, etc., that are food items for *Centropristis striata* and other benthic feeders, one must conclude that the continued use of this trawl net reduces the amount of productive fish habitat. For these reasons, in addition to the ineffectiveness of the gear in sampling commercially important species, alternate nondestructive methods, such as direct observations or the use of mark-recapture techniques with trap catches, should be employed in assessment surveys of the commercially important species of this habitat.

Results of trawl survey work in Australia provide some insight into what can happen to catches in an area after the continued use of commercial trawl gear. Young and Sainsbury (1985) report that "At moderate to low levels of fishing effort, the main effect of fishing on the relative abundance of bottom shelf fishes is by alteration of the relative frequency and spatial distribution of habitat types. In particular this refers to the conversion of areas with dense epibenthos (sponge, corals, hydroids, gorgonians) to areas with sparse epibenthos. (It may be noted that even at the relatively low intensity of trawling of the past few years the fishing effort exerted on the main trawl grounds is sufficient to sweep 50 to 100 per cent of the area of those grounds per year.)." These results are from trawling conducted in 1982 as compared to trawl catches in 1966 from the same locations and at the same time of year. The catch composition shifted from species associated with sponges, soft corals, etc. (during 1966) to those associated with open sandy bottom (during 1982).

A similar type of scenario for the South Atlantic was suggested by Bob Low (pers. comm.): ―Parker et al. (1983) estimated that, in the area they surveyed between Cape Fear and Cape Canaveral, there were 7,403 square km of reef habitat. Of this, 1,743 square km had an average profile exceeding 1 m. Assuming that such ground could not be trawled, this leaves about 5,660 square km (1,398,000 acres) of trawlable reef habitat. The average boat might pull a net with a footrope of 120 feet, giving an effective sweep of the roller gear of about 72 feet maximum. A typical tow over open bottom is perhaps 3 hours at 2 knots. The area swept by the roller gear per tow is then about 20 acres/hour or 60 acres/tow. Assume that 20 boats participate for 4 months (January-April) each year. [Note: The actual number of vessels during 1987 was seven.] The average vessel makes 3 trips/month, with 3 days of fishing each trip. The average (24 hr) fishing day includes perhaps 4 tows. A typical trip therefore consists of 12 tows or 36 hr of fishing. The 20 boats make an aggregate of 240 trips. This equates to 2,880 tows, covering around 172,800 total acres. If each tow was over a previously unswept area, the total area covered by the roller gear would then amount to about 12% of the trawlable reef habitat estimated by Parker et al. (1983). Under one set of assumptions, the area affected by the doors, bridles, and warps would add to this. Under a second set, repetitive trawling over identical areas would reduce the total area impacted. Van Dolah et al. (1987) noted a substantial renewability within a year. There are likely to be 8 months of recovery time between trawling seasons. Doesn't that allow for significant restoration in many of the trawled areas?

The above scenario indicated that about 12% of available habitat between Cape Fear and Cape Canaveral would be impacted annually by trawling, whereas in the Australian work the area impacted was between 50 and 100%. The Council has concluded that the level of damage to the live-bottom habitat in the South Atlantic is significant and that our available knowledge is not sufficient to risk impacting the long-term abundance of snapper and groupers by reducing their habitat. The results shown by Van Dolah et al. (1987) indicated that regeneration of tissue sufficient to have rounded off the tops of partially severed sponges and to have closed wounds on other sponges occurs within a year but that additional growth is limited as indicated by some of the sponges being obviously shorter than before the trawling damage. This supports the Council‘s concern because in a four month trawling season there would be a net loss of habitat (i.e., more damage than regrowth) with the effects being cumulative over time. By destroying habitat we destroy the productivity of the resource being harvested and we are in essence drawing on the principal, not just taking the interest so that next year the same amount of trawling will represent more than 12% of the habitat and the year after even more. Given this information, the South Atlantic Fishery Management Council concluded that over the long-term there would be a net loss of existing habitat, which is counter to the Council's habitat policy and the Magnuson-Stevens Act. Indirect evidence of habitat damage is provided in Christian et al. (1985) where they report on attempts to use crab nets rigged with light chain and plastic mud rollers. These nets proved to be inadequate for offshore fish trawling on broken bottom because the light molded plastic mud rollers were not durable and did not prevent net damage. They further reported that captains who tried crab nets soon switched to nets with heavy netting, properly rigged sweep systems and steel vee-doors for trawling over rough bottom. Further indication of habitat damage was presented in Section II of Snapper Grouper Amendment 1 with the numerous references to gear damage, gear loss and the need to use rollers and modified doors to be able to trawl in rough and broken areas.

An additional reference concerning potential habitat damage is provided by Moore and Bullis (1960) when they reported on the discovery of a deep water reef in the Gulf of Mexico. The MV *Oregon* was cruising over the continental slope about 40 nautical miles due east of the Mississippi Delta and observed an unusual tracing on the depth recorder. They sampled this bottom area using a shrimp trawl and reported the following: ―A drag, made over the area with a shrimp trawl, contained a large mass of coral, other invertebrates, and fish. The netting of the trawl was torn and most of its contents were lost, but about three hundred lbs of coral remained in the bag. A sample was brought back to the laboratory where it was identified by Moore as *Lophelia prolifera*.

Invertebrates associated with sponges and corals occur in disproportionately high densities which suggest that they may use sponges and corals as a food source or a refuge from predation (Wendt et al. 1985). These invertebrates in turn serve as a food source for various snapper and grouper species. In addition, corals are very slow growing with some such as *Oculina* sp. only growing between 11 and 16 mm per year (Reed 1981). Damage to these areas can negatively affect the food and shelter available to snappers and groupers. Further, Grimes et al. (1982) note the importance of the live bottom and shelf edge habitats in serving as reservoirs for recruits in shallow areas (less than 30 m).

The best estimate of the number of boats operating in the fishery during the winter of 1986/87 was four boats (one South Carolina boat fishing in South Carolina and three North Carolina boats fishing in South Carolina, Georgia and Florida). The number of vessels increased to seven during the winter of 1987/88. These vessels fished during the slow period for shrimp which is normally January to March/April. Even though the actual number of boats is small, the amount of habitat damage is significant when one realizes that these boats fish directly on the limited live bottom habitat in these areas. Productive snapper grouper habitat on the continental shelf is limited and trawl gear is fished repeatedly in these areas over this three to four month period. Most, if not all, fishermen use Loran which allows them to return to the exact spot and trawl a particular rock out-cropping repeatedly. The data previously described from Australia points out the changes to bottom habitat and catches resulting from such a fishery.

Vermilion snapper in the early 1980s were experiencing growth overfishing (see SAFMC 1983a p. 44-58 for a more detailed discussion). Yield per recruit (or yield per individual) analysis indicated that a 12 inch minimum size will increase yield per recruit from 132 g to 177 g which is equivalent to a 34% increase in yield if recruitment is constant. Confidential data available to the South Atlantic Council indicated that the minimum mesh size of 4 inches is not being adhered to and as a result the Council's prior action establishing the mesh restriction has not been effective in releasing small vermilion (less than 12 inches). The trawl prohibition will result in an increase in yield for vermilion snapper. Catch data from South Carolina (Bob Low, pers. comm.) show a slight negative correlation between trawl landings and hook & line landings ($r = -0.13$). A good fishery independent index of abundance would allow us to examine the affect of trawl catches on abundance of vermilion snapper. Given the available information, the South Atlantic Fishery Management Council concluded that the trawl prohibition would increase yield; however, our ability to measure this increase is lacking.

The potential existed for more vessels to enter the fishery particularly if the calico scallop, shrimp and sea scallop fisheries have not been productive or are not active during this time period. The actual number of vessels during 1987/88 was seven, greater than the number expected. This further supported the Council's concern that effort could have increased rapidly.

Impacts on affected vessels from prohibiting use of trawl gear in the snapper grouper fishery were not significant. Input from public hearings, committee and Council meetings indicated that income from fish trawling made up a small portion of total income. No trawl fishermen came forward with information during the public hearing process indicating that impacts would be significant. Fishermen used this fishing method primarily as a fill-in activity and had the ability to utilize other gear (e.g., electric & hydraulic reels, black sea bass traps, longlines, etc.) to fish snappers and groupers. These general conclusions are supported by the following in Christian et al. (1985): ―The major seafood industry in the South Atlantic Bight is based on shrimp, and this dependence on one crop has made the industry financially precarious....Therefore, fishermen have looked to other activities such as bottom trawling for finfishes to supplement their income. This is not the single salvation for the whole industry. Although fish trawling can offer an alternative which may aid some shrimpers in maintaining year-round income, suitable trawling bottom in this area is limited, and target species of such a fishery (snapper, grouper, and porgies) are relatively long-lived, slow-growing, and can sustain only limited fishing pressure.

Hydraulic Escalator Dredge
Hydraulic escalator dredges have been utilized since the 1940s to harvest shellfish such as clams and oysters and are designed expressly for efficient commercial harvest (Coen 1995). The dredge consists of a water pump supplying a manifold with numerous water jets mounted in front of a conveyer belt that dislodges buried organisms from the sediment (Figure 3). Hydraulic escalator dredges are currently only employed in a limited shellfish fishery in South Carolina state waters. Hydraulic escalator dredges may penetrate the benthos approximately 45.7cm (18in), thus disturbance to the sediment may be substantial (Coen 1995). Increased turbidity, burial/smothering, release of contaminants, increased nutrients, and removal of infauna were offered as potential effects from dredging activities (Coen 1995). Turbidity was found to be elevated only in the immediate vicinity of the harvester operation and downcurrent of the study area to a distance of between 1.5-1.75km. Turbidity values returned to baseline levels within a few hours (Maier et al. 1998). Manning (1957) stated that hydraulic clam dredging can result in severe damage to oysters within a distance of 7.6m (25ft) downcurrent from the site of dredging. Enough sediment was displaced and redeposited to a distance of at least 15.2m (50ft), but not more than 22.9m (75ft) downcurrent, to cause possible damage to oyster spat. Beyond about 22.9m (75ft) there was no visible or measurable change in the experimental area. Sediment plumes caused by dredge activity were found by Ruffin (1995) to range from less than 1 to 64 hectares. Although sediment plumes increased turbidity and light attenuation at all depths, plumes in shallow water (less than 1.0 m) caused greater increase in turbidity and light attenuation over background than did plumes in deeper waters. Plume decay is based largely on sediment size, with sand particles settling quickly while the silt/clay particles remain in suspension longer. Sites were monitored for storm disturbance to compare against dredge impacts. Storm events increased turbidity and light attenuation com pare d to calm days but not to the extremes obtained in sediment plumes.

Storm events affect a large area at a low intensity while dredging intensely affects a more localized area. SAV subjected to decreased light penetration will inhibit reproduction, reduce propagule abundance, and structurally weaken SAV due to the need of plants growing higher into the water column (Ruffin 1995). Ruffin (1995) concluded that clam dredging increased light attenuation to the point of inhibiting SAV growth. As may be expected, hydraulic clam dredges are highly destructive t o SAV within the immediate area of intensive dredging (Manning 1957; Godcharles 1971). Due to the capability of the water jets to penetrate the substrate to a depth of 45.7cm (18in), virtually all attached vegetation in its path is uprooted (Godcharles 1971). As the use of this gear is limited to a fishery in South Carolina where SAV does not exist, discussion of SAV impacts are included only to provide information on potential impacts should this gear type be considered in the future for other geographic area s where SAV may be found. Although there may be physical impacts associated with escalator dredge activity, the chemical effects apparently are not as dramatic. Dissolved oxygen, pH, and dissolved hydrogen sulfide were measured throughout the harvesting process at varying distances. No consistent patterns of depression or release were noted. Only in the direct plume of the harvester did they measure even a temporary reduction in dissolved oxygen and pH (Coen 1995). While it is recognized that there is infaunal and epifaunal species mortality associated with escalator dredge activity, based on all evidence, these community impacts appear to be short-term (Godcharles 1971; Peterson et al. 1987a; Coen 1995). Coen (1995) noted that the escalator possibly provides a tilling effect of the bottom that has been observed to be beneficial to subtidal oyster and clam populations.

Typically, shellfish dredging operations have typically not been considered to have deleterious results, since its effects are perceived to be negligible compared to natural environmental variation (Godwin 1973). Coen (1995) concluded that based on all direct and indirect evidence, the short-term effects of subtidal escalator harvesters are minimal, with no long-term chronic effects, even under worst case scenarios. Observed effects were often indistinguishable from ambient levels or natural variability. Recovery of the benthos may vary greatly depending on sediment composition. Shallower trenches with shorter residency times are typical of coarse sediments (i.e., sand), whereas trenches generated in muddy, finer sediments are typically deeper, often persisting for greater than 18 months (Coen 1995). Godcharles (1971) observed that trenches had filled in between 1 to 10 months, depending on bottom type. In regard to SAV, no trace of *Thalassia testudinium* recovery was evident after more than 1 year, though *Caulerpa prolifera* began to re-establish itself in dredge areas within 86 days (Godcharles 1971).

Otter Trawl

Perhaps the most widely recognized and criticized type of gear employed in the southeast region is the otter trawl. Utilized in both state and Federal waters of the Gulf of Mexico and South Atlantic, otter trawls pursue invertebrate species such as shrimp and calico scallops, as well as finfish species such as flounder and butterfish. As the most extensively utilized towed bottom-fishing gear (Watling and Norse 1998), trawls have been identified as the most wide-spread form of disturbance to marine systems below depths affected by storms (Watling and Norse 1998; Friedlander et al. 1999).

Jones (1992) broadly classified the way a trawl can affect the seabed as: scraping and ploughing; sediment resuspension; and physical habitat destruction, and removal or scattering of non-target benthos. The following discussion attempts to group documented impacts into either physical-chemical (e.g., sediment resuspension, water quality) or biological impact categories. In many instances documented habitat impacts overlap these categories.

Physical-Chemical Repercussions

The degree to which bottom trawls disturb the sediment surface depends on the sediment type and the relationship between gear type, gear weight, and trawling speed (ICES 1991). Various parts of trawl gear may impact the bottom including the doors, tickler chains, footropes, rollers, trawl shoes, and the belly of the net. While the components of trawl gear are similar, trawl design may vary greatly. Potential impacts may be shared by all otter trawls, but differences in the weight of trawl doors, footrope design, and operation (tow times), will result in a broad spectrum of impact severity. Furthermore, the number and weight of tickler chains vary the degree of disturbance. Margetts and Bridger (1971) concluded that the cumulative effect of tickler chains is likely to emulsify the sediment to a depth proportional to the number of chains. Additionally, the cumulative effect of intense otter trawling is as important as gear weight and design in impacting the benthos (Ball et al. 2000). Although the effect of one passage of a fishing (trawl) net may be relatively minor, the cumulative effect and intensity of trawling may generate long-term changes in benthic communities (Collie et al. 1997). Trawl gear disturbs the benthos as it is dragged along the bottom. Otter trawl doors, mounted ahead and on each side of the net, spread the mouth of the net laterally across the sea floor. The spreading action of the doors results from the angle at which they are mounted, which creates hydrodynamic forces to push them apart and, in concert with the trawl door‘s weight, also to push them toward the sea bed (Carr and Milliken

1998). The doors, due to their design and function, are responsible for a large proportion of the potential damage inflicted by a trawl. The footrope runs along the bottom of the net mouth and may be lined with lead weight and rollers. On relatively flat bottom, it is expected that the footrope would not have a major effect on the seabed and its fauna (ICES 1995). However, in areas of complex benthic habitat the footrope would likely have more impact with the benthos.

The South Atlantic Draft Calico Scallop FMP noted that during the early years of the calico scallop fishery, large quantities of benthic material were removed by trawlers. Reports were received during numerous meetings about entire ―rocks‖ being removed. One individual provided a print-out from a depth sounder which indicated a large amount of bottom relief in a particular area prior to the calico scallop fishery. Similar bottom plots after the calico scallop fishery operated in that area indicated a relatively flat bottom (SAFMC 1998b). Additionally, while the footrope generally causes little physical substrate alteration aside from smoothing of bedforms and minor compression on relatively flat bottoms (Brylinsky et al. 1994), these minor compressions can lead to sediment ―packing‖ after repeated trawling activity on the same general areas (Schwinghammer et al. 1996; Lindeboom and de Groot 1998). Further compression can result from the dragging of a loaded net (cod end) along the bottom. The remaining path of the trawl is influenced by the ground warps which, while not in direct contact with the seabed, can create turbulence that resuspends sediment (Prena et al. 1999).

Trawl gear, particularly the trawl doors, penetrates the upper layer of the sediments which liquefies the affected sedimentary layers and suspends sediment in the overlying water column. This sediment ―cloud‖ generated by the interaction of the trawl gear with the benthos and the turbulence created in its wake contributes to fish capture (Main and Sangster 1979; 1981). The appearance of the sediment cloud, but not its size, is governed by the type of seabed. Brief observations on different seabed types show that soft, light-colored mud produces the most opaque and reflective type of cloud and the fine mud remains in suspension much longer than coarse sand. Studies of sediment disturbance by trawls vary greatly, though it can be concluded that benthic habitat areas composed of fine sediments (e.g., clay, mud) are affected to a greater degree than those with coarse sediments (e.g., sand). In sandy sediments, otter boards cannot penetrate deeply due to the mechanical resistance of the sediment, and the seabed in sandy areas is more rapidly restored by waves and currents (DeAlteris et al. 1999). Short-term alterations to sediment size distribution result from the various rates of redeposition of suspended sediments; as noted before, coarse grains (i.e., sand) settle out rapidly while fine grains (i.e., silt) settle out relatively slowly. In general, resuspended sediments settle out of the water column at a rate inversely proportional to sediment size (Margetts and Bridger 1971). Transport of fine-grained sediments away from trawled area s due to this slow settling period may result in permanent changes to the sediment grain size of a trawled area. Again, this effect will be more pronounced in mud/silt habitats than in habitat areas consisting of heavier sand. For example, suspended sediment concentrations of 100-500mgl-1 were recorded 100m astern of shrimp trawls in Corpus Christi Bay, Texas (Schubel et al. 1979), an estuary dominated by muddy sediments. The same study estimated that the total amount of sediment disturbed annually as a result of shrimp trawling was 25-209,000,000m3, which is 10-100 times greater than the amount dredged during the same period for maintenance of shipping channels in the same area.

ICES (1973) concluded that the physical effects of trawling in tidal waters cannot be permanent. However, it is possible that frequently repeated trawling of one ground with a mixed sediment type bottom in strongly tidal waters might ultimately alter the nature of the bottom towards being predominantly coarse sand because the finer particles are carried away to settle elsewhere. In deeper waters, impacts may be more profound and longer lasting. Engel and Kvitek (1998) investigated two adjacent areas in 180m of water to determine the differences between a heavily trawled site and a lightly trawled site. The data indicated that intensive trawling significantly decreased habitat heterogeneity. Rocks and mound s were less common and sediments and shell fragments were more common in the highly trawled area. Rocks and mounds were more abundant in the lightly trawled area, as well as the amount of flocculent matter and detritus. They theorized that less trawling most likely results in an area with more topographical relief and allows for the accumulation of debris, whereas consistent trawling removes rocks, smoothes over mounds, and resuspends and removes debris. Likewise, Kenchington (1995) found that sand ripples were flattened and stones were displaced after a trawl passage. Churchill (1989) modeled sediment resuspension by trawling and found that this may be a primary source of suspended sediment over the outer shelf where storm-related bottom stresses are weak.

Otter trawl doors were found to have a maximum cutting depth of 50 - 300mm (Drew and Larsen 1994) and, according to Schubel et al. (1979), the footropes of shrimp trawlers in Texas disturbed approximately the upper 50mm of the sediment. Schwinghamer et al. (1996) observed that while the trawl doors may leave scours or depressions, trawling activity reduces the overall surface roughness. Ripples, detrital aggregations, and surface traces of bioturbation are smoothed over by the mechanical action of the trawl and the suspension and subsequent deposition of the surface sediment. In general, the passage of an otter trawl was found to have a minor physical and visual impact on the soft sedimentary sea bed, represented by a flattening of the normally mounded sediment surface and some disturbance of the sessile epifauna (Lindeboom and de Groot 1998). The potential to suspend sediments varies greatly, in large part due to the type of sediment a trawl is working on. Regardless, the suspension of sediments, whether fine silt or coarse sand, impacts the chemical and physical attributes of water quality. The resuspension of sediments may influence the uptake or release of contaminants and, depending on the frequency of disturbance, the nature of the contaminant(s). Clearly, such effects may be more significant where contaminant burdens are relatively high, (e.g., near areas affected by major industrialization,) (ICES 1995). Repetitive trawling on the same ground may enhance nutrient release from sediments and that estimates of average trawling effort for large areas may be unsuitable for estimating these effects (ICES 1995). This has important implications on nutrient cycling in areas that are regularly trawled. Pilskaln et al. (1998) found that impacts include burial of fresh organic matter and exposure of anaerobic sediments; large nutrient delivery to the water column, possibly impacting primary production; increase in nitrate flux out of the sediments; and reduced denitrification (conversion of remineralized nitrogen into N2 gas). All of these may have desirable or undesirable ecosystem impacts. An increase in nitrate fluxes to the water column may alter primary production (phytoplankton), potentially benefiting fisheries, or stimulating deleterious phytoplankton growth that results in harmful algal blooms (Pilskaln et al. 1998).

Increased water turbidity as a result of trawling activity has the potential to compress the width of the euphotic zone, wherein light levels are sufficient to support photosynthesis (North Carolina Division of Marine Fisheries 1999). The magnitude of this effect depends on sediment size, duration and periodicity of the trawling event, gear type, season, and site-specific hydrographic and bathymetric features (Paine 1979; Kinnish 1992).

Dredging studies would indicate that the effect of turbidity is greatly dependent on local conditions. Windom (1975) found that sediment resuspension caused by dredging operations significantly reduced phytoplankton growth in a naturally clear estuary (south Florida) but not in a naturally turbid estuary (Chesapeake Bay). Additionally, increased turbidity resulting from trawling activities may reduce primary production of benthic microalgae. This may have serious consequences as benthic microalgae support a variety of consumers and can be a significant portion of total primary production (Cahoon and Cooke 1992; Cahoon and Tronzo 1992; Cahoon et al. 1990; 1993). Increased turbidity also has may reduce the foraging success of visual predators (Minello et al. 1987) and contribute to the mortality of organisms by impeding the normal functioning of feeding and respiratory structures (Sherk et al. 1975). Sediment resuspension may increase the amount of organic matter resulting from enhanced primary production and may stimulate heterotrophic microbial production. If the amount of resuspended organic material is copious, sustained proliferation of heterotrophic microflora will reduce the dissolved oxygen content within the water, and widespread hypoxia or anoxia could ensue to the detriment of benthic and pelagic fauna (West et al. 1994). Conversely, oxygen penetration into the sediment might be enhanced through trawling activity, resulting in shifts in mineralization patterns and redox-dependent chemical processes. Among other consequences, a change from anaerobic to aerobic conditions facilitates the degradation of hydrocarbons. As Kaiser (2000) pointed out, bottom trawls are designed to stay in close contact with the seabed and an inevitable consequence of their design is the penetration and resuspension of the seabed to some extent. While it is possible to reduce the direct physical forces exerted on the seabed by modifying fishing practices, the benefits are questionable and catches would most certainly suffer. Despite attempts to improve gear design, as long as bottom dwelling species are harvested using towed gear, there will be inevitable sediment resuspension.

Biological Repercussions

The physical disturbance of sediment, such as the ones previously discussed, can also result in a loss of biological organization and reduce species richness (Hall 1994). In general, the heavier the gear and the deeper its penetration of the sediment, the greater the damage to the fauna. Impacts also will vary depending on type of habitat the gear is working. Gibbs et al. (1980) determined that shrimp trawling occurring within a sandy estuary had no detectable effect on the macrobenthos. After repeated trawls the sea bottom appeared only slightly marked by the trawl‘s passage. However, Eleuterius (1987) noted that scarring due to shrimp trawls in Mississippi SAV was common, especially in deeper water. Trawling activities left tracks and ripped up the margins of the beds, and great masses of seagrass were often observed floating on the surface following the opening of shrimp season. Furthermore, Wenner (1983) noted that the use of an otter trawl on hardbottom habitat may inflict considerable damage. The net damages the sponge-coral habitat by shearing off sponges, soft corals, bryozoans, and other attached invertebrates. Therefore, it is not necessarily that trawl gear is doing a constant level of damage, but rather particular habitats are more vulnerable to impacts than others.

Numerous studies cite specific, direct biological impacts to habitat such as the reduction of algal and SAV biomass (Tabb 1958; Fonseca et al. 1984; Bargmann et al. 1985; Peterson et al. 1987a;

Sánchez-Lizaso et al. 1990; Guillén et al. 1994; Ardizzone et al. 2000). Gelatinous zooplankton and jellyfish, which provide habitat to juvenile and other fish species, are greatly impacted as they pass through the mesh of mobile gear (Auster and Langton 1999). Fishing activity may reduce the size and number of zooplankton aggregations and disperse associated fishes. Furthermore, there is a directed trawl fishery for cannonball jellyfish in the Gulf of Mexico.

While this fishery removes jellyfish which may provide habitat for juvenile fish, otter trawls utilized in this fishery do not interact with the benthos. Trawls in the Gulf of Mexico and South Atlantic have been noted to impact coral habitat, damaging and destroying various colonies (Moore and Bullis 1960; Gomez et al. 1987; Bohnsack, personal observation). Loss of sponges and associated cnidarian benthos has been documented to lead to a reduction in fish catch (Sainsbury 1988; Hutchings 1990). Sponges are particularly sensitive to disturbance because they recruit periodically and are slow growing in deeper waters (Auster and Langton 1999). Bradstock and Gordon (1983) observed that trawling virtually destroyed large areas dominated by encrusting coralline growths (bryozoans), reducing colony size and density. Probert et al. (1997) documented the bycatch of benthic species that occurs in a deep-water trawl fishery and noted the vulnerability of pinnacle communities and deepwater coral banks such as the Oculina habitat area of eastern Florida. Van Dolah et al. (1983; 1987) conducted experimental trawl surveys over hard bottom habitat consisting of coral and sponge off the coast of Georgia. A single pass of an otter trawl on this habitat damaged all counted species (Van Dolah et al. 1983; 1987).

However, only the density of barrel sponges was significantly decreased by trawling activities. It should be noted that these studies did not investigate the cumulative impacts of trawls. The repetitive effects of trawling over the same area can be expected to have more severe consequences to benthic habitat. While Moran and Stephenson (2000) estimated that a demersal otter trawl reduced benthos (greater than 20cm in maximum dimensions) density by 15.5% in a single pass, Cappo et al. (1998) estimated that complete denuding of the sea bottom structure occurs after 10-13 trawl passes over the same area. Of equal importance are the observations of Moran and Stephenson (2000), who noted variations among trawl studies, possibly due to differences of employed ground ropes. These variations are a warning against generalizations about the impact of otter trawls on attached benthos. As many epifaunal and infaunal organisms create structures which provide habitat for other species, summaries of these studies and their findings are included. For example, many infauna species and other bioturbators have an important role in maintaining the structure and oxygenation of muddy sediment habitats. Consequently, any adverse effects on these organisms would presumably lead to changes in habitat complexity and community structure (Jennings and Kaiser 1998). Furthermore, the loss of biogenic epifaunal species (epibenthic habitat) increases the predation risk for juveniles of other species, thereby lowering subsequent recruitment to adult stocks (Bradstock and Gordon 1983; Walters and Juanes 1993; Jennings and Kaiser 1998). Therefore, reduction in biomass of epifaunal species may be considered a reduction or degradation of habitat in certain instances and trawling has been documented to decrease mean individual biomass of epibenthic species (Sainsbury et al. 1993; Prena et al. 1999). While it may be hard to quantify the impact this loss presents to habitat-dependent organisms, it should be noted nonetheless. In a long-term study of Corpus Christi Bay, Texas, Flint and Younk (1983) noted that the continual minor and random disturbance, both in time and space, of channel sediments by large tanker traffic and shrimp trawling probably was sufficient to keep these communities in a state of constant disruption. This allowed the opportunists to persist more successfully than other species. The disturbed channel sites of the study, though viable, consistently had lower densities, lower numbers of species and corresponding low diversities contrasted to the lesser impacted shoal sampling sites (Flint and Younk 1983). Engel and Kvitek (1998) investigated two adjacent areas in 180 m of water to determine the differences between a heavily trawled site and a lightly trawled site.

They concluded that high-intensity trawling apparently reduces habitat complexity and biodiversity while simultaneously increasing opportunistic infauna and the prey of some commercial fish. The data indicated that intensive trawling significantly decreased habitat heterogeneity. All epifaunal invertebrates counted were less abundant in the highly trawled area. Bergman and Santbrink (2000) estimated direct mortality on various species of benthic megafauna from a single pass of an otter trawl (sole fishery) at between 0-52% for silty sediments and between 0-30% for sandy sediments. In general, small-sized species tend to show lower direct mortalities, when compared with larger sized species and smaller individuals of megafaunal species tend to show lower mortalities than larger-sized ones. Krost and Rumohr (1990) noted damage directly resulting from otter trawl doors. Benthic organisms were found to be reduced in number by 40 to 75% in otter board tracks, as compared to control sites. Biomass was also generally reduced. However, they found almost no differences in epibenthic species such as crustaceans. In shallow areas with densely packed sediments, inhabitants of the upper sediment layer were found to suffer most by the trawling impact.

In contrast to the above studies, there are several studies that document no significant habitat impact. Van Dolah et al. (1991) found no long-term effects of trawling on an estuarine benthic community; five months of shrimp trawling in areas previously closed to fishing were found to have no pronounced effect on the abundance, diversity, or composition of the soft bottom community when compared to nearby fished areas. They concluded that seasonal reductions in the abundance and numbers of species sampled had a much greater effect than fishing disturbance. In a power analysis of their sampling strategy, Jennings and Kaiser (1998) noted that Van Dolah et al. (1991) only considered changes in the abundance of individuals and the number of species. This assumes that the response of the infauna to trawling disturbance was unidirectional, whereas a consideration of changes in partial dominance might have been more sensitive to subtle changes in the fauna. Yet, Jennings and Kaiser (1998) stated that the results of Van Dolah et al. (1991) were plausible and that light shrimp trawls probably do not cause significant disturbance to communities in poorly sorted sediments in shallow water. Sanchez et al. (2000) determined that sporadic episodes of trawling in muddy habitats may cause relatively few changes in community composition. They found similar infaunal community changes in both fished and unfished control areas through time. Sanchez et al. (2000) also noted that the decrease in the abundance of certain species in the unfished control areas may indicate that the natural variability at the experimental site exceeds the effect s of fishing disturbance. Regardless, Ball et al. (2000) commented that epifauna are generally scarce in muddy sediment habitats, and detection of fishing effects on such species has therefore been limited.

While the passage of a trawl may damage or destroy macroinfauna, Gilkinson et al. (1998) suggested that smaller infauna are resuspended or displaced by a pressure wave preceding otter trawl doors and are redeposited to the sides of the gear path. Due to a buffer effect caused by a displacement field of sediment (sand), bivalves incur a low level of damage (5%) by the passing of a trawl door. In contrast to coarse sediment communities where the infauna are found within the top 10 cm, organisms in soft mud communities can burrow up to two meters deep (Atkinson and Nash 1990). Due to their depth, it is likely that these organisms are less likely impacted by passing trawls (Jennings and Kaiser 1998), though it should be noted that the energetic costs of repeated burrow reconstruction may have long-term implications for the survivorship of individuals.

Studies documenting impacts to habitat from successive trawling are not prevalent. However, a few studies suggest that shifts in species abundance and diversity are a result of the cumulative effects of trawling. Over a longer time scale (i.e., 50 years), Ball et al. (2000) suggested that fishing disturbance may ultimately lead to an altered, but stable, community comprising a reduced number of species, and hence, diversity. Sainsbury et al. (1993; 1997) noted that composition of a multispecies fish community in Australia was at least partially habitat-dependent and that historical changes in relative abundance and species composition in this region were at least in part a result of the damage inflicted on the epibenthic habitat by the demersal trawling gear. In summary, trawling has the potential to reduce or degrade structural components and habitat complexity by removing or damaging epifauna; smoothing bedforms which reduces bottom heterogeneity; and removing structure producing organisms. Trawling may change the distribution and size of sedimentary particles; increase water column turbidity; suppress growth of primary producers; and alter nutrient cycling. The magnitude of trawling disturbance is highly variable. The ecological effect of trawling depends upon site- specific characteristics of the local ecosystem such as bottom type, water depth, community type, gear type, as well as the intensity and duration of trawling and natural disturbances. It should also be noted that there is not a direct relationship between the overall amount of trawling effort and the extent of subsequent impacts or the amount of fauna removed because trawling is aggregated and most effort occurs over seabed that has been trawled previously (Pitcher et al. 2000). Yet, several studies indicate that trawls have the potential to seriously impact sensitive habitat areas such as SAV, hardbottom, and coral reefs. In regard to hard bottom and coral reefs, it should be recognized that trawlers do no t typically operate in these areas due to the potential dam age their gear may incur.

While trawl nets have been documented to impact coral reefs, typically resulting in lost gear (Bohnsack, personal observation), these incidents are usually accidental. Partially in response to accusations of trawl activity on hard bottom habitat, a recent research effort to investigate potential impacts on the Florida Middle Ground Habitat Area of Particular Concern concluded that there was no evidence of trawl impacts or other significant fishery related impacts to the bottom (Mallinson, unpublished report). However, low-profile, patchy hard bottom or sponge habitat areas are more likely impacted from trawls due to the gear‘s ability to work over these habitat types without damaging the gear. Regardless, while it may be concluded that trawls have a minor overall physical impact when employed on sandy and muddy substrates, the available information does not provide sufficient detail to determine the overall or long-term effect of trawling on regional ecosystems.

Recovery of substrate depends on sediment type, depth, and natural influences such as currents and bioturbation. Schoellhamer (1996) investigated sediment resuspension within Tampa Bay, a shallow estuary with fine non-cohesive material (muds absent), and found that sediment concentrations returned to pre-trawl conditions approximately 8 hours after disturbance. The cumulative effects of several trawlers operating were not investigated. DeAlteris et al. (1999) found that scars similar to those that occur from otter trawl boards disappear relatively quickly in a shallow sand environment, while those occurring in a deeper mud habitat took as long as two months to disappear. DeAlteris et al. (1999) also found that natural disturbances to mud substrate in 14 m of water are rarely capable of disturbing the seabed. Therefore, recovery of fishery-related impacts in deeper water may be protracted due to the lack of natural events that help deposit sediments and fill trawl scars. Ball et al. (2000) determined that intensive demersal trawling over muddy seabeds leads to apparent long –term alteration of the seabed. Trawl tracks in muddy sediments may last up to 18 months; however, in areas of strong tidal or wave action, they are likely to disappear rapidly. Also, in areas where levels of bioturbation are high, and regular turnover of sediment produces large numbers of mounds on the seabed, trawl tracks will be filled relatively quickly (Ball et al. 2000). Habitats in deeper water tend to recover at a slower rate. Berms and furrows generated by trawl doors generally disappeared within one year in sandy habitats in depths of approximately 120-146 m (Schwinghamer et al. 1998; Prena et al. 1999). More dramatic is the estimate of 50-75 years to fill a typical trawl mark (~15 cm scour depth) in deep water (greater than 175m) by Friedlander et al. (1999). The greater the water movement, the faster the scars will be filled in (Jones 1992). Churchill (1989) and Krost et al. (1990) reported an increase in the frequency of tracks attributed to trawl doors in deeper water, presumably where water movement and natural impacts are less pronounced.

In general, few studies document recovery rates of habitat. Those that do investigate recovery usually only do so after a single treatment which does not reflect the reality of fishing impacts which are ongoing and cumulative. For example, Van Dolah et al. (1983; 1987) noted that hard bottom habitat in his trawl study recovered within one year. However, the experiment did not investigate the cumulative and repetitive effects of trawling at commercial intensities. As noted by an ICES (1995) study, due to the cumulative effects of trawling, focus on the scale of individual trawl impacts may be inadequate for estimating the importance of impacts on benthic communities. ICES (1994) stated that deep water coral banks (e.g., *Oculina varicosa*), due to their fragility, long life spans, and infrequent recruitment, may be nearly exterminated by a single passage of a trawl and are unlikely to recover ―within a foreseeable future.‖ Likewise, SAV would also have a protracted recovery time in comparison to sediments. SAV recovery may vary by species and can be greater than two years if the rhizomes of the plant are removed (Homziak et al. 1982). Regardless, the majority of studies concur that shallow communities have proved to be resilient due to their adaptation to highly variable environmental conditions and thus, recovery is usually swift. Kaiser et al. (1996a) found epifaunal communities in 35m of water that were experimentally trawled were indistinguishable from control sites after six months.

In areas of low current or great tidal exchange (e.g., deep ocean), where the benthos is not adapted to high sediment loads, the adverse effects of sediment resuspension by gear could persist for decades (Jones 1992). Recovery of small epibenthic organisms may be relatively rapid, but recovery of larger epibenthic organisms would be expected to be much slower. Though they did not discuss depth as a controlling factor, Sainsbury et al. (1993; 1997) indicated that there would be a considerable time lag after trawling ceases before recovery of large epibenthic organisms is substantial. Boesch and Rosenberg (1981) predicted that recovery times for macrobenthos of temperate regions would be less than five years for shallow waters (including estuaries) and less than ten years for coastal areas of moderate depth.

The majority of management recommendations indicate that marine reserves or gear zoning may be the most effective at reducing habitat impacts. However, other specific recommendations can be extracted from several studies. Tabb (1958) recommended that otter trawls not be permitted to operate in the bait shrimp fishery due to potential impact to SAV communities. Van Dolah et al. (1987) suggested that trawls with doors attached directly to the nets would greatly reduce the bottom area damaged by trawling activities. The use of artificial reefs to protect the seabed, in particular along the perimeter of SAV habitat areas, from trawling has also been offered (Guillén et al. 1994; Ardizzone et al. 2000). The use of semi-pelagic trawls would avoid the majority of habitat impacts that demersal trawls are associated with. However, while the use of semi-pelagic nets does not significantly impact the benthos, catch efficiency may be greatly reduced. Furthermore, enforcement on the use of semi-pelagic nets remains difficult (Moran and Stephenson 2000). Carr and Milliken (1998) offered more straightforward recommendations: target certain species and modify gear appropriately; encourage the use of lighter sweeps; reduce the sea bottom available to trawlers that fish very irregular terrain; and opt for stationary gear over mobile gear. It is suggested that where fishing effort is constrained within particular fishing grounds, and where data on fishing effort are available, studies that compare similar sites along a gradient of effort have produced the types of information on effort impact that will be required for effective habitat management (Collie et al. 1997; Auster and Langton 1999). Additionally, the use of an indicator species (e.g., quahogs) that provides a historical record of fishing disturbance events could greatly enhance the interpretation of perceived changes ascertained from samples of present-day benthic communities (Macdonald et al. 1996; Kaiser 1998). Finally, the use of tracking devices (VMS) would provide a means for identifying the most heavily fished areas and those, if any, that are presently unfished (Macdonald et al. 1996; Kaiser 1998).

Comprehensive mapping of benthic habitats may provide the necessary information to determine what areas are at risk from fishery-related impacts. Utilized in conjunction with information that details fishing effort and area, gear zoning that limits the vulnerability of sensitive habitats while minimizing economic impacts to fishery participants should be considered.

Oyster Dredge

An oyster dredge consists of a metal rectangular frame to which a bag-shaped net of metal rings is attached. The frame's lower end is called the raking bar, and is often equipped with metal teeth used to dig up the bottom. The frame is connected to a towing cable and dragged along the seabed. Oyster dredges are widely utilized in state waters along the Gulf of Mexico, as well as the South Atlantic. Mechanical harvesting of oysters using dredges extracts both living oysters and the attached shell matrix and has been blamed for a significant proportion of the removal and degradation of oyster reef habitat (Rothschild et al. 1994; Dayton et al. 1995; Lenihan and Peterson 1998). Lenihan and Peterson (1998) observed that less than one season of oyster dredging reduced the height of restored oyster reefs by ~30%. Reduction in the height of natural oyster reefs is expected to be less than that of restored reefs because the shell matrix of natural reefs is more effectively cemented together by the progressive accumulation of settling benthic organisms, while restored reefs are initially loose piles of shell material. Regardless, it is likely that the height of natural reefs is also reduced by dredging because a large portion of extracted material from natural reefs by dredges is shell matrix. Lenihan and Peterson (1998) stated that it was probable that reduction in reef heights in a Neuse River, North

Carolina estuary was due to decades of fishery-related disturbances caused by oyster dredging. At an annual removal rate of 30%, restored reefs would be completely destroyed after less than 4 years of harvesting. Furthermore, they determined that the height reduction of oyster reefs through fishery disturbance impacted the quality of habitat due to the seasonal bottom-water hypoxia/anoxia which caused a pattern of oyster mortality and influenced the abundance and distribution of fish and invertebrate species that utilize this temperate reef habitat (Lenihan and Peterson 1998). Their results illustrated that tall experimental reefs – those mimicking natural, ungraded reefs – were more dependable habitat for oysters and other reef organisms than short reefs – those mimicking harvest-degraded reefs – because tall reefs provided refuge above hypoxic/anoxic bottom waters. Chestnut (1955) also documented that intensive dredging over a period of years resulted in the removal of the productive layer of shell and oyster, leaving widely scattered oysters and little substrate for future crop of oysters.

Glude and Landers (1953) noted that dredges mixed the sandy-mud layer and the underlying clay. Fished areas were found to be softer and have less odor of decomposition than the unfished control site. Glude and Landers (1953) also found a decrease in benthic fauna in the fished sites versus the unfished control sites. Conversely, a study conducted by Langan (1998) which looked at the impacts oyster dredging had on benthic habitat, as well as sediment resuspension resulting from dredging activity, concluded with different results. He noted that the size-frequency of oysters from the control site was biased towards older and larger specimen s with poor recruitment. Oysters from the dredged site illustrated good recent recruitment, while larger specimens were not as abundant as the control site. No significant differences between the two areas were found in number, species richness, or diversity of epifaunal and infaunal invertebrates, indicating that dredge harvesting had no detectable effect on the benthic community. Sediment suspension resulting from dredging activity appeared to be localized. It should be noted that the study failed to evaluate fishing activity (number of participants, effort) on the dredged site.

Due to overfishing and disease, oysters may now be more economically valuable for the habitat they provide for other valued species than they are for the oyster fishery (Lenihan and Peterson 1998). Rothschild et al. (1994) suggested the establishment of broodstock sanctuaries that includes the designation of ―no-fishing‖ restrictions in specific areas. Lenihan and Micheli (2000) also recommended the closure of some oyster reefs to harvest. Maintaining high densities of oysters on some intertidal reefs may help to preserve future oyster harvests and broodstock. Furthermore, protecting some reefs will also preserve the ecological functions that oyster reef provide such as improving water quality and providing essential recruitment, refuge, and foraging habitat for numerous marine species.

Scallop Dredge (Inshore)
Scallop dredges are similar to crab scrapes, though scallop dredges utilized in the southeast generally do not have teeth located on the bottom bar. Scallop dredges are predominately used on SAV beds where bay scallops can be efficiently harvested, and thus, are primarily limited to state

waters. Popular bay scallop fisheries exist both off Florida and North Carolina. This gear, while similar, is not the same type of dredge utilized offshore to harvest calico scallops (*Argopecten gibbus*) or Atlantic sea scallops (*Placopecten magellanicus*). Though scallop dredges do not have teeth that would easily uproot SAV, studies have noted a reduction of algal and SAV biomass from their use (Fonseca et al. 1984; Bargmann et al. 1985). The reduction of SAV (*Zostera marina*) biomass was linearly related to the number of times a particular area was dredged, and the effects of dredging were proportionately greater on soft bottom than hard bottom (Fonseca et al. 1984). The Fonseca et al. (1984) study utilized an empty dredge that was 60% of the legal limit for a commercial dredge, and was not employed in conjunction with a boat as the commercial fishery does. Hand dredging was done to eliminate propeller scour which commonly occurs in shallow SAV beds. In commercial scalloping, the added dredge and scallop weight, as well as the propeller wash, could be expected to have a greater imp act (Fonseca et al. 1984). In general, more damage from scallop dredging occurred to SAV in soft substrates (i.e., mud) than hard substrates (i.e., sand). In softer sediments, plants were uprooted and damage to underground plant tissues, including meristems, occurred. In harder sediments, damage was found to be generally greater for above ground parts; underground meristems were left intact and able to begin to repair shoots or produce new ones after impacts had ceased (Fonseca et al. 1984).

Fonseca et al. (1984) determined that in a lightly harvested SAV area, with less than 25 % biomass removal, recovery occurred within a year. In areas where harvesting resulted in the removal of 65% of SAV biomass, recovery was delayed for two years. After four years, preharvesting biomass levels were still not obtained. These estimates were based on termination of fishery-related impacts. Continued fishing activity would likely lead to prolonged recovery and continued degradation. Homziak et al. (1982) estimated that SAV recovery can be greater than two years if the rhizomes of the plant are removed.

Due to the importance of SAV beds as a nursery area to other species, loss of eelgrass meadows should be avoided. Fonseca et al. (1984) suggested that harvest area rotation may minimize habitat impact.

Scallop Dredge (Offshore)

Scallop dredges (Figure 7) utilized to harvest calico or sea scallops consist of a metal frame that supports tickler chains and a metal ring bag that collects the shellfish. Though not widely utilized in the southeast, the gear has been included in this review due to their inclusion as an approved gear in the South Atlantic. The majority of studies on scallop dredge impacts originate from areas with extensive scallop fisheries such as the northwest and northeast Atlantic.

Due to the potential for the gear to have considerable weight and the fact that it is dragged along the bottom, habitat impacts are expected to occur. Drew and Larsen (1994) estimated that a scallop dredge maximum cutting depth would be 40 - 150mm. Kaiser et al. (1996a) found that scallop dredging greatly reduced the abundance of most species, causing significant changes in the community. It was noted that a large proportion of some animals (such as echinoderms) were not captured or passed through the mesh of the gear. The scallop dredge catches contained a low proportion of non-target species which indicates that the belly rings allow the bycatch to escape. However, the study did not investigate the extent of damage/injury to organisms that were not captured. Likewise, Collie et al. (1997) found areas on Georges Bank that were impacted by scallop dredges to have lower species diversity, lower biomass of fauna, and dominated by hard-shelled bivalves, echinoderms, and scavenging decapods. Areas less impacted by dredges had higher diversity indices. However, it should be noted that portions of Georges Bank consist of cobble habitat which is encrusted with a diverse array of epibenthic species.

Perhaps more applicable to the areas in the southeast where calico scallops are harvested off North Carolina and Florida, would be a study conducted by Butcher et al. (1981), who determined that scallop dredges had little or no environmental effect when they were used on large-grained, firm sand bottom that was shaped in roughly parallel ridges. The area in this study was also noted to be a fairly uniform, low species diversity community. Turbidity caused by the turbulence of the dredge quickly dissipated due to the nature of the substrate. Additionally, Jolley (1972) found no detrimental dredging effects on sand substrates. Yet, there is a potential for dredges to impact coral adjacent to scallop beds, especially the scallop grounds which occur in close proximity to the Oculina Bank off eastern Florida. Should a scallop dredge impact *Oculina* coral, there would be severe results, similar to the conclusion s reached by ICES (1994) for trawls. This study determined that deep water coral banks such as those composed of *Oculina varicosa*, due to their fragility, long-life spans, slow growth, and infrequent recruitment, may be nearly exterminated by a single passage of a trawl. Recovery of this habitat area, ―within a foreseeable future,‖ is unlikely (ICES 1994).

Skimmer Trawl
Skimmer trawls are positioned along the side of a boat and pushed through the water to harvest shrimp. Two nets are typically used, one on each side of the boat. Skimmer trawls are supported by a tubular metal frame that skims over the bottom on a weighted metal shoe or skid. Tickler chains are also utilized along the base of the net. Because of the construction attributes of this gear type, skimmer trawls are generally restricted to water 3.05m (10 ft) or less which would limit them to state waters.

Skimmer trawls work on mud bottoms in water generally 3.05m (10ft) or less. The weighted shoe and tickler chains impact the bottom, resulting in sediment resuspension. Skimmer trawls may cause bottom damage due to improperly tuned or poorly designed gear (skids and bullets) or prop damage in shallow areas (Steele 1994). Furthermore, because skimmer trawls are used in shallow water, they may have a detrimental impact on critical nursery areas such as the marsh/water interface, SAV, or other sensitive submerged habitats. However, skimmer trawls are expected to impact the bottom less than otter trawls due to the absence of doors (Nelson 1993; Steele 1993). Coale et al. (1994) believed that the skimmer trawl would not have any greater effects on SAV than the otter trawl. They found it doubtful that the inside weight and outer shoe of the skimmer trawl would cause greater detrimental effects to the benthos than the heavy doors of an otter trawl. Based on underwater observations, Coale et al. (1994) suggested that the weight and shoe combination may be less-damaging than otter trawls. However, habitat such as sponges and SAV are cut off by tickler chains and lead lines whereas otter trawl doors can dig in and tear up the bottom. Given the difference in the amount of area covered by each on normal tows, Kennedy, Jr. (1993) found it doubtful that there would be much difference in the amount of habitat loss between skimmer trawls and otter trawls.

Kennedy, Jr. (1993) recommended that the use of skimmer trawls in Florida should be restricted to those areas currently approved for otter trawls. Due to the associated impacts to SAV, a prudent recommendation would be to limit skimmer trawl use to non-vegetated substrates.

6.2.2.2 Static Gear

Channel Net

Channel nets are fixed to pilings, docks, or shore installation and utilize current flow to capture shrimp, therefore, channel nets are limited to use within state waters. Though impacts of channel nets were not discussed specifically, it may be inferred from Higman (1952) that channel nets have negligible impact on habitat due to catch composition and the lack of interaction with the benthos.

Gillnet and Trammel Net

Gillnets (Figure 9) consist of a wall of netting set in a straight line, equipped with weights at the bottom and floats at the top, and is usually anchored at each end. As fish swim through the virtually invisible monofilament netting, they become entangled when their gills are caught in the mesh, hence the name. Gillnets may be fixed to the bottom (sink net) or set midwater or near the surface to fish for pelagic species. A trammel net is made up of two or more panels suspended from a float line and attached to a single lead line. The outer panel(s) are of a larger mesh size than the inner panel. Fish swim through the outer panel and hit the inner panel which carries it through the other outer panel, creating a bag and trapping the fish. Smaller and larger fish become wedged, gilled, or tangled. Gillnet s are widely used in numerous fisheries, both in state waters and in Federal waters. Trammel nets are primarily used in state waters, though they are an authorized gear in the Caribbean for both the spiny lobster and shallow water reef fish fisheries.

The majority of the studies that have investigated impacts of fixed gillnets have determined that they have a minimal effect on the benthos (Carr 1988; ICES 1991; ICES 1995; Kaiser et al. 1996b). An ASMFC (2000) report determined that impacts to SAV from gillnets would be minimal. Likewise, West et al. (1994) stated that there was no evidence that sink net (gillnet) activities contributed importantly to bottom habitat disturbance. However, Carr (1988) noted that ghost gillnet s in the Gulf of Maine could become entangled in rough bottom. He observed one net that had its leadline and floatline twisted around each other and tightly stretched between boulders. Furthermore, Williamson (1998) noted that gillnets can snag and break benthic structures. Gomez et al. (1987) noted that gill nets set near reefs occasionally results in accidental snaring often resulting in damage to coral. Bottom set gillnets have led to habitat destruction in different regions (Jennings and Polunin 1996). Bottom gillnets set over coral may cause negative impacts as the weighted lines at the base of the net often become entangled with branching and foliaceous corals. As the nets are retrieved, the corals are broken (Öhman et al. 1993). This observation has also been noted in a study by Munro et al. (1987), which documented that reefs are frequently damaged by the hauling of set (gill) nets, and the problem has been exacerbated by the use of mechanical net haulers or power blocks. Aside from the potential impacts cited on coral reef communities, the available studies indicate that habitat degradation from gillnets is minor.

Several studies note that lost gillnets are quickly incorporated by marine species. Cooper et al. (1988) found ghost gillnets in the Gulf of Maine covered with a heavy filamentous growth, exceeding 75% coverage on some nets. Anemones, stalked ascidians and sponges were attached to and growing to the net float lines (Carr et al. 1985; Cooper et al. 1988). Erzini et al. (1997) found that lost trammel nets and gill nets in shallow water (15-18m) on rocky habitat (analogous to coral reefs and hard bottom habitat) were colonized by various species, primarily macrophytes, which after three months completely blocked the meshes of some parts of the nets. Some netting would contact reef habitat, becoming heavily overgrown and eventually blended into the background. After a year, most of the netting was destroyed; those remnants that remained were completely colonized by biota (Erzini et al. 1997). Erzini et al. (1997) also noted that the nets eventually became incorporated into the reefs, acting as a base for many colonizing plants and animals. The colonized nets then provided a complex habitat which was attractive to many organisms. For example, large schools of juvenile fish were often observed in the vicinity of these heavily colonized nets, which may provide a safe haven from predators. Johnson (1990) and Gerrodette et al. (1987) noted that as gillnets tend to collapse and ―roll up‖ relatively quickly, they may form a better substrate for marine growths and thereby attract fish and other predators which may get entangled, ultimately causing the net to sink. Therefore, one may assume that gillnets may be more of a ghostfishing problem and entanglement hazard to marine life than as an impact to habitat.

Catch by entanglement nets during 1988 was 1,398 lbs from North Carolina through Georgia (less than 1% of the combined state catch) and 253,739 lbs from the Florida East Coast (6% Florida East Coast catches). Much of the Florida landings are from a directed stab net fishery for gray snapper that operates in the EEZ. The Gulf Council and the State of Florida have prohibited entanglement nets. Florida regulations read as follows: ―No person shall harvest in or from state waters any snapper of the family of Lutjanidae or any member of the genera *Epinephelus* or *Mycteroperca* by or with the use of any gear other than those types of gears specified in Subsection 1, provided however that snapper and grouper harvested as an incidental bycatch of other species lawfully harvested with other types of gears shall not be deemed to be unlawfully harvested in violation of this section, if the quantity of snapper/grouper so harvested does not exceed the bag and possession limits as specified elsewhere.‖ The South Atlantic Council‘s actions track the Florida regulations in intent with respect to limiting possession to the bag limit and for species without a bag limit, no possession is allowed. Florida prohibited entanglement nets because it is an inappropriate gear to use on live bottom. Some of the reef fish are not necessarily found on the live bottom, however, many are and fishermen use stab nets to catch gray snapper on the live bottom areas.

The Council has concluded that entanglement nets are not an appropriate gear for the snapper grouper fishery and the prohibition will prevent use and/or expansion from North Carolina through Florida's East Coast. Entanglement nets targeting species other than those included in the management unit are limited to the bag limit if the species is under a bag limit, and if no bag limit is applicable, then no retention is allowed.

SAFMC Prohibition on the Use of Entanglement Nets

Snapper Grouper Amendment 4 prohibits the use of entanglement nets including, but not limited to, gill nets and trammel nets, for the harvest of species in the snapper grouper management unit

(SAFMC 1991a). The simultaneous possession of entanglement nets and species in the management unit is prohibited.

Hoop Net

A hoop net is a cone-shaped or flat net which may or may not have throats and flues stretched over a series of rings or hoops for support. The net is set by securing the cod or tapered end to a post or anchored to the bottom. The net is played out with the current until fully extended, and then is allowed to settle to the bottom. The net is marked with a buoy for easy retrieval and identification purposes. The duration of time that a hoop net is set depends on the same factors that influence the duration of the set of a gill net and should be determined in a similar fashion. To harvest, the hoop net is raised at the cod end and the fish are removed.

While there are no studies that document the effect of hoop nets on habitat, due to its use primarily on flat bottoms the gear probably has less of an impact than traps.

Longline

Longlines use baited hooks on offshoots (gangions or leaders) of a single main line to catch fish at various levels depending on the targeted species. The line can be anchored at the bottom (Figure 12) in areas too rough for trawling or to target reef associated species, or set adrift, suspended by floats (Figure 13) to target swordfish and sharks. Longlines are widely utilized in numerous fisheries throughout the southeast region.

When a vessel is retrieving a bottom longline it may be dragged across the bottom for some distance. The substrate penetration, if there is any, would not be expected to exceed the breadth of the fishhook, which is rarely more than 50mm (Drew and Larsen 1994). More importantly is the potential effect of the bottom longline itself, especially when the gear is employed in the vicinity of complex vertical habitat such as sponges, gorgonians, and corals.

Bottom longlines in the snapper grouper fishery

The Council prohibited the use of bottom longline gear for snapper grouper in the South Atlantic EEZ within 50 fathoms (SAFMC 1994). Catch by bottom longlines during 1988 was 470,306 lbs from North Carolina through Georgia (6% of the combined state catches) and 576,310 lbs from the Florida East Coast (13% Florida East Coast catch). The Council was concerned about the use of bottom longline gear targeting species in the snapper grouper management unit in live bottom areas. Habitat damage and intense competition among users are problems that arise when this gear is used within 50 fathoms where significant live bottom occurs and where competition with hook and line vessels occurs. The Council concluded that this gear is appropriate for use in the deep-water snowy grouper/tilefish fishery where much of the bottom is mud with sparse live bottom areas. Allowing use of this gear deeper than 50 fathoms would preserve the traditional fishery which takes place in deeper water out to 50 fathoms. Based on information from South Carolina, up until 1983 the snapper grouper fishery was limited to vertical hook and line or bandit reels. Bottom longlines were introduced in the Gulf of Mexico after hook and line gear became less effective due to decreases in resource abundance; use of the gear grew rapidly. Up until this point there has been no gear prohibition on bottom longlines. After the golden tilefish and snowy grouper fisheries were developed, bottom longlines became the predominant gear, again as resource abundance declined. For species like snowy grouper and tilefish, it was not very efficient to use vertical hook and lines as the resource abundance declined from unfished levels. As the tilefish and snowy grouper stocks off South Carolina declined, the number of people using longlines decreased. Off South Carolina virtually all of the golden tilefish occurred well outside the 50-fathom mark and there was more than enough gear to adequately harvest these resources in the mid-depth zone. Vertical lines are much more environmentally acceptable and less damaging than bottom longlines.

This regulation essentially segments the mid-shelf and the deep-water complex to the bottom longlines. This measure was supported during the public hearing process and the Council concluded that prohibiting use of longline gear within 50 fathoms will prevent the problems of habitat damage and intense competition while at the same time allow fishermen using this gear to continue fishing in deeper water. This action effectively limits longlines to targeting the deep water component of the snapper grouper fishery and keeps the use of longlines outside of the rough bottom habitat.

The Council very briefly considered moving the line in to the 40 fathom contour but was concerned that there are substantial Oculina coral banks along this depth zone. It was further noted that the 50 fathoms was a compromise from the 100 fathom contour (which was mentioned) and that the 50 fathom contour effectively separates the inshore and deep water snapper grouper complexes.

Impacts on habitat

Observations of halibut longline gear off Alaska included in a North Pacific Fishery Management Council Environmental Impact Statement (NPFMC 1992) provide some insight into the potential interactions longline gear may have with the benthos. During the retrieval process of longline gear, the line was noted to sweep the bottom for considerable distances before lifting off the bottom. It snagged on whatever objects were in its path, including rocks and corals. Smaller rocks were upended and hard corals were broken, though soft corals appeared unaffected by the passing line. Invertebrates and other light weight objects were dislodged and passed over or under the line. Fish were observed to move the groundline numerous feet along the bottom and up into the water column during escape runs, disturbing objects in their path. This line motion has been noted for distances of 15.2m (50ft) or more on either side of the hooked fish. Based on these observations, it is logical to assume that longline gear would have a minor impact to sandy or muddy habitat areas. However, due to the vertical relief that hardbottom and coral reef habitats provide, it would be expected that longline gear may become entangled, resulting in potential impacts to habitat. Due to a lack of interaction with the benthos, pelagic long lines would have a negligible habitat impact.

SAFMC Prohibition on the Use of Bottom Longlines

The Council prohibits bottom longlining in the wreckfish fishery in the entire South Atlantic EEZ (SAFMC 1991a). A bottom longline is a stationary, buoyed, and anchored groundline with hooks attached. Regulations prohibit simultaneous possession of wreckfish and all the necessary components for bottom longlining.

The Council was concerned about wastage of fish, gear loss, gear conflict, habitat damage, and negative economic effects (both short and long run) attributable to the use of bottom longline gear in the wreckfish fishery. The bottom habitat on the wreckfish fishing grounds, which comprise an area of the Blake Plateau of approximately 50-75 square nautical miles, is characterized by a rocky ridge system having a vertical relief greater than 50 meters and a slope greater than 15° (SAFMC 1993). The depth range in this area is 450-600 meters; the substrates in areas of the Blake Plateau exhibiting significant relief are generally characterized as composed of manganese phosphate pavements, phosphorite slabs and coral banks (Pratt and McFarlin 1966; Stetson et al. 1969). This high relief, in conjunction with the strong tidal effects, makes gear loss probable (as reported by fishermen who have already tried longlines in the wreckfish fishery) which results in the loss of all fish on the gear as well as those which get hooked subsequently. Testimony from fishermen indicated gear loss on wreckfish longline sets was as great as 100% of the gear taken out on a single trip. According to accounts from fishermen, extensive lengths of lost longline gear have been observed on their fathometers. Fishermen can apparently see fish hooked on parted longline gear but are unable to recover the parted gear and its catch. Wreckfish fishermen use circle hooks that virtually prevent fish from working the hook free. The Council recognized that there was also some ghost fishing potential from lost vertical gear but believes that the extent of potential loss with vertical gear is much smaller by virtue of the fewer number of hooks used and the greater control over the gear.

Although the area is 50-75 square nautical miles, virtually all wreckfish fishing takes place along limited, high relief ledge areas within this area because wreckfish are found along the ledges and are not evenly distributed over the wider area. The sub-areas that produce wreckfish are typically 300 yards wide and 1-4 nautical miles long. Thus far, fishermen fishing vertical drop gear have been able to work in relatively close proximity without any major conflicts. If bottom longlines had been allowed to be used in this area, vessels would have not only lost gear due to the rough bottom, but this lost gear would create a hazard for those using vertical lines which would result in loss of that gear. This problem would have become progressively worse over time as more gear was lost, the more hangs were created for both longline and vertical gear, creating even more gear loss. This condition could have continued until much of the ground is unfishable. The wire cable that is used will remain a hazard for many years as the rate of decay is slow. While extensive hangs may ultimately provide protection for the resource due to much of the fishing grounds being unfishable, it may well result in the loss of the fishery. The use of longlines will result in gear losses to vertical hook and line fishermen that far exceed their losses prior to the introduction of longlines. This will serve to reduce benefits to those fishing with the traditional vertical gear.

The potential for gear entanglement and gear conflict also raised the issue of vessel safety. It was the Council's opinion that this situation would have lead to conflicts that jeopardize the safety of the vessels and fishermen participating in the wreckfish fishery.

Longline cable on the bottom has the potential to break some of the ledges, overhangs and associated organisms, and otherwise damage the habitat on which the wreckfish depend. Habitat damage caused by the longlines would violate the SAFMC habitat policy and should be avoided.

The wreckfish fishery has employed efficient vertical gear since its inception, and the addition of longlines would have eroded benefits to the majority of fishermen and adversely impact the resource and habitat. If longlines had been allowed, then all or at least many wreckfish fishermen may have been forced to adopt the gear in order to compete resulting in more gear loss from parted longlines. The Council determined that bottom longlines were not in the best interest of the wreckfish resource, habitat, fishermen or society at large. Further, the problems outlined justified prohibiting this gear/fishing method in the wreckfish fishery.

Pound net
A pound net consists of a fence constructed of netting that runs perpendicular to shore which directs fish to swim voluntarily into successive enclosures known as the heart, pound, or pocket. Pound nets are exclusively utilized in state waters.

An ASMFC (2000) report determined that impacts to SAV from pound nets are expected to be minimal, unless the net is constructed directly on SAV. West et al. (1994) also stated that pound nets do not contribute to benthic disturbance. Due to the limited amount of space a pound net may impact, it is expected that pound nets have minimal impact on habitat.

Trap and Pot
Traps and pots are rigid devices, often designed specifically for one species, used to entrap finfish or invertebrates. Depending on the type of fishing a trap is used for, most traps are generally baited and equipped with one or more funnel openings, they are left unattended for some time before retrieval. Traps and pots are weighted to rest on the bottom, marked with buoys at the surface, and are sometimes attached to numerous other traps to one long line called a trot line. Traps and pots are widely used on a variety of habitats in both state and Federal waters to commercially harvest species such as lobster, blue crabs, golden crabs, stone crabs and black sea bass. Wire-mesh fish traps are one of the principal fishing gears used in coral reef areas in the Caribbean (Appledorn 2000).

SAFMC Prohibition on the use of fish traps
It should be noted that many of the studies used in forming this document refer to fish trap fisheries outside of the continental US. These fisheries are different from crustacean trap fisheries operating on the South Atlantic coast in that the traps are built to selectively capture crabs and lobster and avoid bycatch of untargeted finfish. There are few studies to date regarding the bycatch rate of finfish but anecdotal information from fishermen and fisheries managers point out that spiny lobster traps do not capture significant amounts of snapper, grouper and other ornamental reef fish.

The Council prohibited the use of fish traps in the South Atlantic EEZ; however, black sea bass traps may be used north of Cape Canaveral (Vehicle Assembly Building, 28° 35.1' N Latitude). Fish traps were banned in federal waters off the South Atlantic states in 1992 and banned in the Gulf of Mexico west of Cape San Blas (located at about the middle of the Florida Panhandle) in 1997.

In general, pots can cost anywhere between $30-$50 USD to construct. It does take some skill in determining an appropriate location for fishing trap gear, and efficiency is based on how many traps a fishermen can service in one working day. Traps will soak an average of 1-2 weeks before they are checked. While traps can catch a wide variety of marine organisms, fishermen place traps in specific areas to avoid bycatch of untargeted species. One downfall to trap fishing is that the gear can be lost in storm events, so to avoid ―ghost fishing‖ most traps have degradable panels and escape rings that rot off allowing fish to exit the trap. While trap fishing gear has created user-group conflicts in the past, managers are in the process of choosing particular fishing zones which will help the general public become more aware of trap fishing areas.

Due to their use to harvest species associated with coral and hard bottom habitat, traps and pots have been identified to impact and degrade habitat. Gomez et al. (1987) noted the incidental breakage of corals on which traps may fall or settle constitute the destructive effects of this gear. Within the Virgin Islands State Park, Garrison (1998) found 86% of the fish traps were set on organisms (live coral, soft coral, SAV) living on the sea floor. Damage to the live substrate has far-reaching negative effects on the marine ecosystem because the available amount of shelter and food often decreases as damage increases. Another study conducted by Garrison (1997) had similar results, as 82% of traps rested directly on live substrate, with 17% resting on stony corals. It is important to note that the aforementioned statistics (Garrison, 1987-1998) do not reflect the way trap based crustacean fisheries operate within the continental United States. Studies from the Florida Fish and Wildlife Conservation Commission (Tom Matthews, personal communication) confirm that only 2% of spiny lobster traps are fished on top of stony coral reef habitats. Hunt and Matthews (1999) found that lobster and stone crab traps reduce the abundance of gorgonian colonies from rope entanglement. Furthermore, seagrass smothering occurs from trap placement on SAV beds, resulting in SAV ―halos.‖ .‖ Studies also confirm that traps set for no longer than a two week period do not pose an adverse threat to seagrass ecology as the seagrass subsequently recovers. Van der Knapp (1993) noted that fish traps set on staghorn coral easily damaged the coral. It appeared that in all observed cases of injury due to traps, the staghorn coral regenerated completely, although the time for regeneration varied from branch to branch. The greatest impact noted from the setting of traps was observed when the point of the trap‗s frame ran into coral formations. Several different species of coral were observed to suffer damage from fish traps. Observations of at least one damaged coral specimen noted that algae growth prevented regeneration in the damaged portion of the coral. Additionally, complete deterioration of a vase sponge was observed after it had been severely damaged by a trap. Traps are not placed randomly, rather they are fished in specific areas multiple times before fishing activity moves to other grounds. Therefore, the damage caused by wire fish traps in this study has a concentrated cumulative effect in particular areas rather than being uniform over all coral reef habitat.

Appledorn et al. (2000) commented that fish traps may physically damage live organisms, such as corals, gorgonians, and sponges, which provide structure and in some cases, nutrition for reef fish and invertebrates. Damage may include flattening of habitats, particularly by breaking branching corals and gorgonians; injury may lead to reduced growth rates or death, either directly or through subsequent algal overgrowth or disease infection. During initial hauling, a trap may be dragged over more substrate until it lifts off the bottom. Traps set in trotlines can cause further damage from the trotline being dragged across the bottom, potentially shearing off at their base those organisms most important in providing topographic complexity.

Traps that are lost or set unbuoyed are often recovered by dragging a grappling hook across the bottom. This practice can result in dragging induced damage from all components (grappling hook, trap, trotline). The area swept by trotlines upon recovery is orders of magnitude greater than the cumulative area of the traps themselves. Appledorn et al. (2000) documented that single-buoyed fish traps off La Parguera, Puerto Rico, have an impact footprint of approximately 1 m2 on hard bottom or reef. Of the traps investigated in the study, 44% were set on hard bottom or reef, resulting in 23% damage to coral colonies (70 cm2 average), 34% damage to gorgonian colonies (56 cm2 average), and 30% damage to sponges, though sponges were less frequently impacted due to their patchy distribution. Hauling these wire fish traps resulted in 30% of the traps inflicting additional damage to the substrate. In a similar study focusing on fish trap impacts conducted off St. Thomas, U.S.V.I., by Quandt (1999), 40% of all traps investigated were found to be resting on reef substrate. On average, 4.98% of all hard corals and 47.17% of all gorgonians were damaged; tissue damage averaged 20.03% to each gorgonian. Secondary impacts, such as trap hauling and movement due to natural disturbances were not investigated. However, the effects of pulling a string of two or more wire fish traps would most likely be much greater than one trap alone.

Eno et al. (1996) found pots that landed on, or were hauled through beds of bryozoans caused physical damage to the brittle colonies. It was noted that several species of sea pens bent in response to the pressure wave created by a descending pot and were lying flat on the seabed. When the pot was removed, the sea pens were able to reestablish themselves in the sediment. A species o f sea fan also was found to be flexible and specimens were not severely damaged when pots were hauled over them. This suggests that in some instances the direct contact of certain gears may not be the primary cause of mortality, rather the frequency and intensity may be more important. Additionally, Sutherland et al. (1983) cited little apparent damage to reef habitats inflicted from fish traps off Florida. The study found four derelict traps sitting atop high profile reefs with four other traps observed within a live-bottom area. There was no visual evidence that traps on the high profile reef killed or injured corals or sponges. One uprooted gorgonian was observed atop a ghost trap in a live bottom area. However, these observations were made on randomly located derelict traps. Thus, the primary impacts that may occur during deployment and recovery could not be evaluated.

Trap loss

Gear failure, theft, and improper placement are several of the many reasons why traps are lost both inshore and offshore. Gear failure can occur because of pot warp (line) parting, buoys separating from the pot warp, or buoys breaking up. Normal wear and tear, powerboat propellers, and sea turtles or sea gulls biting the buoys or pot warp can cause gear failure. Theft is also a major cause of lost traps in many areas. Losses also occur because of setting the traps too deep or on too steep a slope. Storm surge and wave action can cause loss of traps, particularly in shallow inshore waters during hurricane and foul weather events. Traps without buoys are less susceptible to storm damage, but may be moved from a site by currents or wave action and become irretrievable. In coralline areas, the buoy lines may become entangled on coral, chafe, and break. Offshore, losses are primarily caused by large vessels cutting or dragging gear, gear failure, and storms. Strong currents submerging buoys or sweeping traps away from the locations where they were set and traps becoming entangled with other fishing gear and anchors have also been cited as causes of trap loss.

The percentage of traps lost varies considerably among studies by both area and depth fished. Wolf and Chislett (1974) reported fish trap losses of 10-20% per trip in exploratory efforts in deep water shelf edges in the Virgin Islands. They attributed these losses to pots tumbling down steep slopes. Craig (1976) reported a fish trap loss rate of about 20% for a period of six months with some loss due to theft while trap fishing off Boca Raton, Florida. In Broward County, Florida, the same study reported that fish trap fishermen reported an average of 20.3% annual loss due mainly to strong currents, entanglement and theft. Dade County, Florida trap fishermen reported losing 1-5 traps per trip, with an annual loss of 100%. Losses were due to theft or loss of buoys. Trap theft was such a problem that traps were brought back to port at the end of each fishing day in Dade (Sutherland and Harper 1983). Sutherland and Harper reported that Monroe County, Florida trap fishermen had estimated average annual trap losses of 63%. The losses were mainly from currents and severance of buoys by large ships in deep water and from vandalism inshore. Trap loss in the spiny lobster fishery was not a problem in Collier County, Florida with an annual loss of only 5% due to the fact that fishermen brought back traps to the dock after each trip (Taylor and McMichael 1983). About 85% of traps used off Key Biscayne, Florida were lost with most losses attributed to theft (Sutherland et al. 1987). Fish trap loss from theft and severed fouling lines was reported as a major problem in the Virgin Islands (Swingle et al. 1970; Olsen et al. 1974; Sylvester 1972).

In Jamaica, Munro and Thompson (1973) had such a theft problem in their study that the use of buoyed traps had to be abandoned. Losses due to theft, storms, and vessels cannot easily be controlled, but fish trap fishermen can inspect gear frequently for wear and tear and use more durable materials.

Fish traps that fishermen cannot locate and retrieve or that are abandoned but are still capable of catching fish, are often referred to as ghost traps. Ghost traps have long been a subject of concern, but opinions have changed considerably over time. Since Olsen et al. (1978) made their observations that if traps were lost, juvenile and forage species mortality could decimate a fishing ground, they suggested that considerable mortality could take place over the 1-2 years before the wire mesh corroded away, and indicated corrosion time would be longer and mortality would be greater for small sizes of mesh. A study by Harper and McClelland (1983) estimated the average fishing life of eight traps observed off Key Biscayne to be from 5.5 to 157 days before becoming unable to capture fish. They also found that 19.2% of the fish that entered the trap died (Harper and McClelland 1983). While the decay and catch rates of ghost traps are not well documented, at least some evidence indicates that lost traps quickly become damaged and ineffective (Sutherland et al. 1978). Most reports of injury and mortality from lost ghost traps are anecdotal but underwater video presented to the South Atlantic Fishery Management Council on June 11, 1990 documented dead and injured fish in ghost traps in the Florida Keys. The video was presented by Capt. Fernand Braun (a charter fishing guide) in an effort to persuade the Council to ban fish traps.

Derelict traps are lost or abandoned traps that are incapable of catching fish due to structural damage or deterioration. Derelict traps may have small holes or breaks and gaps between ceiling and floor panels and walls, or entire panels degraded or missing (Smolowitz 1978). Traps become derelict in a number of ways. Predator damage, corrosion, escape windows opening, and materials fastened to escape devices decomposing have all been documented.

Munro et al. (1971) speculated that lost fish traps that have accumulated large numbers of fish may be attacked and rendered ineffective by large predators such as nurse sharks (*Ginglymostoma cirratum*). Harper and McClelland (1983) found funnel openings enlarged with the prongs bent back and speculated that the damage was by large predators such as cubera snapper (*Lutjanus cyanopterus*), great barracuda (*Sphyraena barracuda*), yellow jacks (*Caranx bartholomae*), and lemon sharks (*Negaprion brevirostris*) attempting to escape and that mortality of these fish was high. Craig (1976) found that escapement through trap holes caused by predators became a problem if traps were not hauled after five or six days. Fish are rarely caught in traps with holes or breaks in the mesh and even small holes or breaks in the wire mesh apparently render them ineffective (Craig 1976; Sutherland and Harper 1983; Ward 1983).

Sutherland et al. (1983) found juvenile fish numerous in and around derelict fish traps. The derelict traps and other manmade objects appeared to serve as artificial reefs on barren sand sea floor areas (Sutherland et al. 1983; Harper and McClelland 1983). Sutherland et al. (1983) observed that fish were absent or rare near traps on or adjacent to reefs.

Impacts on habitat

The Council concluded that the issue of wire fish traps was a critical issue to the State of Florida and in the long term to the entire South Atlantic as well. Florida deliberated the issue of fish traps for many years and the Florida State Legislature prohibited the use of wire fish traps in 1980. The snapper grouper stocks are more overfished off Florida than they are anywhere else in the South Atlantic.

The Council concluded that fish traps are non-selective by size and by species (e.g., red grouper recruit to the hook and line fishery at around 19" and to the trap fishery at around 11"). Bohnsack et al. (1989) notes that modifications to mesh size will alter the size of fish caught. This study concluded that total value, species caught, number of individuals and mean total weight per haul declined with meshes larger or smaller than 1.5" hexagonal mesh. The mesh sizes required to correlate with the 20" minimum sizes would be so large as to result in de facto prohibition on use of fish traps.

Based on studies regarding fish traps between 1980-1990, the SAFM Council has concluded that wire fish traps capture a significant amount of bycatch. Information contained in Bohnsack et al. (1989) documents the bycatch of these species. Unfortunately, a variety of ornamental reef fish were not recorded separately in the commercial landings data until recently, thus the commercial landings data are not available to quantify the extent to which catches of these species have increased. We also expect that there has been a decline in the amount of ornamental reef fish caught as bycatch since wire fish traps were banned from federal waters (and state waters in Florida).

Since March 1, 1991 the State of Florida has prohibited the harvest of tropical fish: ―The purpose and intent of this Chapter is to protect and conserve Florida‗s tropical marine life resources and to ensure the continued health and abundance of these species. . The affect of selective removal of herbivores on the health of coral reefs was discussed by LaPointe (1989). These species were harvested by fish traps more frequently than by hook and line gear. Again, due to the fact that commercial statistics did not record these fish by species, data was unavailable to document the level of harvest by fish traps or by hook and line.

The further intent of this Chapter is to ensure that the harvesters in this fishery use non-lethal methods of harvest and that the fish, invertebrates and plants so harvested be maintained alive for the maximum possible conservation and economic benefits.‖ Allowing fish traps in federal waters would make Florida‘s regulations difficult, if not impossible, to enforce and would not address Problem #5 which is, that ―the existence of inconsistent state and federal regulation makes it difficult to coordinate, implement and enforce management measures and may lead to overfishing. Inconsistent management measures create public confusion and hinder voluntary compliance.

The way in which fish traps were used made enforcement extremely difficult. All other kinds of fishing gear are eventually brought back to the dock where they can be examined by state marine patrol officers or other law enforcement personnel. Once traps are placed in the water, they were seldom are brought back to the dock. Testimony documents the various kinds of violations recorded in the Key West area (e.g., biodegradable panel requirement violations). The loss of traps was high ranging from 20% to 63% and in certain sectors trap loss may be as high as 100%.

The SAFMC Law Enforcement Committee and Advisory Panel were established to advise the Council on enforceability of various management approaches. They noted that the existing system is difficult to enforce and is incompatible with Florida state law, that the 100-foot contour limitation is difficult to enforce and that poaching is a big law enforcement problem in the fish trap fishery. These two bodies recommended to the Council that a total prohibition on use of fish traps in the South Atlantic EEZ was the most enforceable of all alternatives considered.

The enforcement issue was summarized by Kelley (1990): ―Enforcement is the largest problem of all. There are widespread abuses of the regulations governing the use of fish traps. There seems to be no effective way to enforce regulations in a fishery, such as fish trapping fishing, where gear can't be observed readily by enforcement officials. The largest present day problems in the Florida Keys and southeast Florida are the extensive trap poaching and the use of illegally constructed or deployed traps.‖

The Council recognized that gear that is not brought back to shore at the end of a fishing trip makes enforcement more difficult. The Council considered other, less drastic measures that would allow traps to be used but concluded that the at-sea enforcement required to effectively monitor and ensure compliance with existing regulations does not and will not exist. Therefore, the Council was persuaded that nothing short of a total ban on fish traps would be enforceable.

There is evidence that fish trapping causes habitat damage where fish traps are set in trawls on live bottom and where grappling hooks are dragged across live bottom to retrieve them. Testimony and video records of damaged *Oculina* reefs off Palm Beach County, provided to the Council at the February 1991 meeting, depicted significant and measurable damage to coral reef and live bottom communities. These activities leave an imprint of the trap upon the bottom communities and trenches caused by grappling hooks dragged over the bottom for the purpose of locating and recovering traps. Lost traps not only continue to fish, as it has been pointed out in the ghost trap discussion, but may contribute considerable secondary habitat damage by becoming mobilized at times of storm activity and impacting delicate bottom communities.

The affect of selective removal of herbivores on the health of coral reefs was discussed by LaPointe (1989). These species were harvested by fish traps more frequently than by hook and line gear. Again, due to the fact that commercial statistics did not record these fish by species, data was unavailable to document the level of harvest by fish traps or by hook and line.

Prohibiting fish traps was determined to be consistent with Florida's Coastal Zone Management Plan. Also, internationally, a number of countries (e.g., Bermuda) have tried to manage fish trap gear only to end up prohibiting their use. Bermuda has managed their snapper grouper fishery for a number of years and imposed a limited entry system with trap limitation. In addition, modifications to mesh size were also attempted. The Bermudian Government concluded that regulation the fish trap fishery was not effective and recently imposed a total ban on use of fish traps. The Council concluded that a total prohibition on the use of fish traps was the most effective alternative to address the stated problems and to achieve the plan's stated objectives.

6.2.2.3 Other Gear

Allowable Chemical

Collectors of live tropical reef fish commonly employ anesthetics such as quinaldine. Quinaldine (2-methy lquinoline, C10H9N) is the cheapest and most available of several substituted quinolines (Goldstein 1973). As a result of using this compound near corals where tropical species shelter, there may be residual effects which was discussed in a study by Japp and Wheaton (1975). Short-term impacts of quinaldine include increased flocculent mucus production, retraction of polyps and failure to re-expand with a five minute observation period, and tissue discoloration in certain species. At both study sites, octocorals were found to suffer no long-term impacts. However, a minority of Scleractinians displayed minor damage, including mild discoloration and small patches of dead tissue, three months after quinaldine treatment. Two of these specimens degraded to poor condition or displayed areas of dead tissue more than six months after initial treatment. Overall, Japp and Wheaton (1975) determined that quinaldine exposure resulted in minimal damage to corals.

Barrier Net

Barrier nets are used in conjunction with small tropical nets or slurp guns to collect tropical aquarium species. The net is deployed to surround a coral head or outcropping and may or may not have a pocket or bag that fish are ―herded‖ into for capture. Barrier nets may be utilized by tropical fish collectors in both state and Federal waters. The American Marine Life Dealers Association conducted a survey (Tullock and Resor 1996) that focused on tropical collection practices. The survey defined a sustainable fishing practice as one that a) does not cause physical damage to the reef environment; b) does not impair the captured specimen's longevity in a properly maintained aquarium environment; and c) does not damage non-target species such as coral polyps, other invertebrates, or non-aquarium fish. The survey concluded that barrier nets were a sustainable fishing practice. However, a study conducted by Öhman et al. (1993) summarized that moxy nets, a type of barrier net that is used in other regions to collect ornamental fish species, may break corals during their use. However, it is likely that damage inflicted by barrier nets would be infrequent and incidental in nature, and therefore, the gear would have a negligible effect on habitat.

Castnet

Used to capture baitfish and shrimp, castnets (Figure 18) are circular nets with a weighted skirt that is thrown over a schooling target. Castnets are primarily used in shallow areas such as estuaries, though they may be used to catch baitfish offshore in Federal waters. Castnets have the potential to dislodge organisms or become entangled if utilized over heavily encrusted substrates. Observations by the author have noted numerous castnets entangled amongst sponges and other growth around rough bottom. However, a study conducted by DeSylva (1954) determined that castnets have no detrimental effect on habitat.

Clam Kicking

Clam kicking is a mechanical form of clam harvest primarily practiced in the state waters of North Carolina. The practice involves the modification of boat engines in such a way as to direct the propeller wash downwards instead of backwards. The propeller wash is sufficiently powerful in shallow water to suspend bottom sediments and clams into a plume in the water column, which allows clams to be collected in a trawl net towed behind the boat (Peterson et al. 1987a).

Several studies have noted that the practice of clam kicking reduces algal and SAV biomass (Fonseca et al. 1984; Bargmann et al. 1985; Peterson et al. 1987a). Reduction of SAV biomass was noted to increase with harvest intensity. Intense clam kicking treatments reduced SAV biomass by approximately 65% (Peterson et al. 1987a). Because of the importance of SAV to coastal fisheries and estuarine productivity, Peterson et al. (1987a) noted that intense clam kicking could have long-lasting and serious impacts on many commercially important fisheries. However, clam harvesting had no detectable effect on the abundance of small benthic invertebrates and outside of SAV habitat, clam kicking does not appear to have any serious negative impacts on parameters of ecological value (Peterson et al. 1987a).

SAV recovery can be greater than two years if the rhizomes of the plant are removed (Homziak et al. 1982; Peterson et al. 1987a). Peterson et al. (1987a) observed that SAV had yet to recover after four years of an intense clam kicking treatment. Although Peterson et al. (1987a) designated their heavier clam kicking treatment as ―intense,‖ they conceded that it probably falls well short of the effort that commercial clammers would apply to a productive SAV bed. Limiting the intensity of clam fishing in SAV habitat would probably be beneficial. Peterson et al. (1987a) offered that a restriction of mechanical clam harvesters to unvegetated bottoms may be a suitable mechanism to minimize habitat damage.

Clam Rake, Scallop Rake, Sponge Rake and Oyster Tong

Rakes are used to harvest shellfish and sponges from shallow areas such as bays and estuaries. Oyster tongs, similar to two rakes fastened together and facing each other like scissors, are used by fishermen from the deck of a boat. As these gears are limited by water depth, they are exclusively utilized in state waters.

Lenihan and Micheli (2000) reported that the harvest of shellfish utilizing clam rakes and oyster tongs significantly reduce oyster populations on intertidal oyster reefs. Both types of shellfish harvesting, applied separately or together, reduced the densities of live oysters by 50-80% compared with the densities of unharvested oyster reefs. While oysters are removed, Rothschild et al. (1994) concluded that hand tongs probably have a minor effect on the actual oyster bar structure. Peterson et al. (1987b) compared the impacts of two types of clam rakes on SAV biomass. The bull rake removed over 89% of shoots and 83% of roots and rhizomes in a completely raked area while the pea digger removed 55% of shoots and 37% of roots and rhizomes. Loss or impact on SAV by bull rake was estimated to be double the impact of the smaller pea digger rake. Peterson et al. (1987a) found raking with a pea digger rake reduced SAV biomass by approximately 25%. An earlier study conducted by Glude and Landers (1953) noted that bull rakes and clam tongs mixed the sandy-mud layer and the underlying clay. Fished areas were also softer and had less odor of decomposition than the unfished control site. A decrease in benthic fauna was noted in the fished sites versus the unfished control sites.

Sponges are an important fishery in the Florida Keys and along the west coast of Florida (NOAA 1996). Sponges are dominant organisms in deepwater passes and along hard bottom habitat communities. Sponges create vertical habitat which provides shelter and forage opportunities for other invertebrates and tropical fish species. The fishery in the Keys typically employs a four-pronged iron rake attached to the end of a 5–7 m pole, which hooks the sponges from the bottom. While no studies document the extent of habitat damage from this gear type, it may be concluded that the harvest of sponges directly reduces the amount of available habitat, and thus may present a negative localized impact.

Peterson et al. (1987a) found that SAV biomass recovered to equal and even exceeded expected values within one year. Lenihan and Micheli (2000) recommended the closure of some oyster reefs to shellfish harvest. Maintaining high densities of oysters on some intertidal reefs may help to preserve future oyster harvests and broodstock. Furthermore, protecting some reefs will also preserve the ecological functions that oyster reef provide such as improving water quality and providing essential recruitment, refuge, and foraging habitat for numerous marine species. Due to the extensive habitat that sponges provide, further ecological study on the directed harvest of these organisms should be conducted.

Dipnet and Bully Net

Widely utilized to catch baitfish, crabs, or lobster, varieties of dipnets (Figure 22) consist of a long pole with a bag of netting of varying mesh size that are lowered into the water. Dipnets may also be employed to capture tropical reef fish (Figure 23), though these utilize a short handle and very fine mesh. Additionally, landing nets or hand bully nets (Figure 24) used to capture lobster can be considered a form of dipnet. Varieties of dipnets may be used both in state and Federal waters.

DeSylva (1954) determined that dipnets have no detrimental effect on habitat. However, the use of small dipnets (i.e., tropical fish nets and lobster hand bully nets) may result in minor isolated impacts to coral species as individuals attempt to capture specimens (Barnette, personal observation).

Hand Harvest

Hand harvest describes activities that capture numerous species such as lobster, scallops, stone crabs, conch, and other invertebrates by hand. As many small biogenic structures occur on the sediment surface, even gentle handling by divers can destroy them easily. Movements by divers were observed to cause demersal zooplankters to exhibit escape responses (Auster and Langton 1999). A study that assessed recreational SCUBA activity in the US Caribbean (Garcia-Moliner et al. 2000) concluded that approximately 2% of the total recreational divers in the USVI and 1.9% of the total recreational divers in Puerto Rico were lobstering. Potential impact of approximately 13,532 units occurred in the USVI and 14,946 units occurred in Puerto Rico. In this study, impact units consisted of two hands and two feet (4 units per diver) and impact was broadly defined as ranging from touching coral with hands to the resuspension of sediment by fins. No assessment of habitat degradation or long-term impacts was discussed. Divers pursuing lobster along coral or hard bottom communities have been observed to impact gorgonians and other encrusting organisms (Barnette, unpublished observations).

Harpoon

Harpoons, thrown from the decks of a vessel, are utilized to target swordfish and tuna. As this gear is employed to harvest pelagic species, there is no contact with the benthos and, thus, no impact to habitat.

Haul Seine and Beach Seine

A haul seine is an active fishing system that traps fish by encircling them with a long fence-like wall of webbing. It is made of strong netting hung from a float line on the surface and held near the bottom by a lead line. They are fished either along the shoreline (beach seine) where they are deployed in a semi-circle to trap fish between shore and net or, more typically, fish are encircled away from shore, worked into an even smaller pocket of net and lifted onto a boat for culling (Sadzinski et al. 1996). The use of this gear is limited to state waters. Sadzinski et al. (1996) found no detectable effects from haul seining on SAV. However, possible damage from haul seining to sexual reproduction, such as flower shearing, was not examined. There are possible long-term or cumulative impacts at established haul-out sites, resulting in loss of SAV biomass (Orth personal communication). As the seine is generally used in flat benthic areas to prevent the net becoming damaged, in most cases the impact from seines would be expected to be minor and temporary.

Hook and Line, Handline, Bandit Gear, Buoy Gear and Rod and Reel

These gear types are widely utilized by commercial and recreational fishermen over a variety of estuarine, nearshore, and marine habitats. Hook and line may be employed over reef habitat or trolled in pursuit of pelagic species in both state and Federal waters.

Few studies have focused on physical habitat impacts from these gear types. Impacts may include entanglement and minor degradation of benthic species from line abrasion and the use of weights (sinkers). Schleyer and Tomalin (2000) noted that discarded or lost fishing line appeared to entangle readily on branching and digitate corals and was accompanied by progressive algal growth. This subsequent fouling eventually overgrows and kills the coral, becoming an amorphous lump once accreted by coralline algae (Schleyer and Tomalin 2000). Lines entangled amongst fragile coral may break delicate gorgonians and similar species. Due to the widespread use of weights over coral reef or hardbottom habitat and the concentration of effort over these habitat areas from recreational and commercial fishermen, the cumulative effect may lead to significant impacts resulting from the use of these gear types.

Patent Tong
Similar to hand tongs, hydraulic patent tong s (Figure 26) are much larger and are assisted with hydraulic lift, allowing them to purchase more benthic area in pursuit of oysters. Patent tongs are utilized in the oyster fisheries that occur in state waters. Rothschild et al. (1994) found that hydraulic-powered patent tongs are the most destructive gear to oyster reef structure because of their capability to penetrate and disassociate the oyster reef. The capability arises from the gear weight and hydraulic power. Patent tongs operate much like an industrial crane with each bite having the ability to remove a section of the oyster bar amounting to 0.25m3.

Due to overfishing and disease, oysters may now be more economically valuable for the habitat they provide for other valued species than they are for the oyster fishery (Lenihan and Peterson 1998). Rothschild et al. (1994) suggested the establishment of broodstock sanctuaries that includes the designation of ―no-fishing‖ restrictions in specific areas. Lenihan and Micheli (2000) also recommended the closure of some oyster reefs to harvest. Maintaining high densities of oysters on some intertidal reefs may help to preserve future oyster harvests and broodstock. Furthermore, protecting some reefs will also preserve the ecological functions that oyster reef provide such as improving water quality and providing essential recruitment, refuge, and foraging habitat for numerous marine species.

Purse Seine and Lampara Net
Purse seines are walls of netting used to encircle entire schools of fish at or near the surface. Spotter planes are often used to locate the schools, which are subsequently surrounded by the netting and trapped by the use of a pursing or drawstring cable threaded through the bottom of the net. When the cable has pulled the netting tight, enclosing the fish in the net, the net is retrieved to congregate the fish. The catch is then either pumped onboard or hauled onboard with a crane-operated dip net in a process called brailing. Purse seines are utilized to harvest menhaden in the Gulf and South Atlantic. Similarly, the lampara net has a large central bunt, or bagging portion, and short wings. The buoyed float line is longer than the weighted lead line so that as the lines are hauled the wings of the net come together at the bottom first, trapping the fish. As the net is brought in, the school of fish is worked into the bunt and captured. In the Florida Keys a modified lampara net is used to harvest baitfish near the top of the water column. The wing is used to skim the water surface as the net is drawn in and fish are herded into the pursing section to be harvested with a dip net. Purse seines in the Gulf menhaden fishery frequently interact with the bottom, resulting in sediment resuspension. Schoellhammer (1996) estimated that sediments resuspended by purse seining activities would last only a period of hours.

Pushnet
Employed to harvest shrimp in shallow water, pushnets (Figure 30) consist of netting supported by a frame that is mounted on to a pole, which is then pushed across the bottom. Pushnets are generally utilized on SAV beds where shrimp can be harvested in abundant numbers. DeSylva (1954) determined that push nets have no detrimental effect on habitat.

Slurp Gun

A slurp gun is a self-contained, handheld device that captures tropical fish by rapidly drawing seawater containing such fish into a closed chamber. Slurp guns are typically employed on hardbottom and coral reef habitat in both state and Federal waters. It is possible that tropical collectors may impact coral or other benthic invertebrates in pursuit of tropical species that are harvested on hardbottom or coral habitat areas. However, due to the limited force applied by a diver in an errant fin kick or hand placement, the likely effects to habitat would be minor.

Snare

Recreational divers pursuing spiny lobster often use a long, thin pole that has a loop of coated wire on the end called a snare. The loop is placed around a lobster that may be residing in a tight overhang or other inaccessible location, and then tightened by a pull toggle at the base of the pole in order to capture and extract the lobster.

While there are no studies that evaluate this gear type, it is probable that use of this gear may minimize impacts to habitat in comparison to divers that use no additional gear (hand harvest). Due to the more surgical precision with the snare, divers likely impact the surrounding habitat to a lesser extent than if capturing by hand only due to the required leverage needed by the divers to capture a lobster by hand.

Spear and Powerhead

Divers use pneumatic or rubber band guns or slings to hurl a spear shaft to harvest a wide array of fish species. Reef species such as grouper and snapper, as well as pelagic species such as dolphin and mackerel, are targeted by divers. Commercial divers sometimes employ a shotgun shell known as a powerhead at the shaft tip, which efficiently delivers a lethal charge to their quarry. This method is commonly used to harvest large species such as amberjack.

Gomez et al. (1987) concluded that spearfishing on reef habitat may result in some coral breakage, but damage is probably negligible. A study that assessed recreational SCUBA activity in the US Caribbean (Garcia-Moliner et al. 2000) concluded that approximately 0.7% of the total recreational divers in the USVI and 28% of the total recreational divers in Puerto Rico are spearfishing. Potential impact would be approximately 4,736 units in the USVI and 220,264 units in Puerto Rico. In this study, impact units consisted of two hands and two feet (4 units per diver) and impact was broadly defined as ranging from touching coral with hands to the resuspension of sediment by fins. No assessment of habitat degradation or long-term impacts was discussed. It may be assumed that divers pursuing pelagic species have no effect on habitat due to the absence of any interaction with the benthos.

6.3 Cumulative impacts of fishing and non-fishing activities

This section analyzes cumulative impacts, which are defined by the Council on Environmental Quality (CEQ) as ―impacts on the environment that result from the incremental impact of an action when added to other past, present, and reasonably foreseeable future actions, regardless of who undertakes such actions.‖ Increasing evidence suggests that the most severe environmental effects may not result from the direct impacts of a particular action, but rather from cumulative environmental effects. The incremental loss of important habitat can irreversibly alter the structure and function of the nearshore marine ecosystem and ultimately affect human activities

(Jackson 1997). Further, regional problems are highly vulnerable to small decision effects – the tyranny of small decisions, as evidenced in the Florida Everglades (Odum 1982).

The overall cumulative impact of human-induced activities and natural events remains poorly documented, understood, and in dire need of more study. Nationally, one report noted that ―federal agencies have struggled with preparing cumulative effect analyses since the CEQ issued its National Environmental Policy Act (NEPA) regulations in 1978.‖ (CEQ 1997).

It is evident that the effect of human activity on aquatic systems has been substantial in locations where access and economically profitable modification could be readily accommodated. Dahl (1990) reports that in the 1780's there were about 20.3 million acres of wetlands in Florida, about 6.8 million acres in Georgia, about 6.4 million acres in South Carolina, and about 11.1 million acres in North Carolina. By the 1980's Florida‘s wetlands had been reduced to 11.0 million acres, Georgia‘s to 5.3 million acres, South Carolina‘s to 4.7 million acres, and North Carolina‘s to 5.7 million acres. Overall about 36.3% of all wetlands in states under SAFMC purview have been eliminated. On a state-by-state basis this includes 46% of Florida‘s wetlands, 23% of Georgia‘s wetlands, 27% of South Carolina‘s wetlands, and 49% of North Carolina‘s wetlands. A 2001 National Research Council report found that, as a result, by the 1980s the area of wetlands in the contiguous United States had decreased to approximately 53% of its extent one hundred years earlier (NRC 2001).

According to the FWS Status and Trends of Wetlands in the Conterminous United States 1998 to 2004 there was an estimated net gain in wetlands of 191,750 acres, however the report did not draw conclusion regarding the quality of the nation‘s wetlands and counted over 700,000 acres of open water ponds as wetlands. Intertidal wetlands declined by an estimated 28,416 acres, with the greatest percent change attributed to marine intertidal wetlands. The overriding factor in the decline of estuarine and marine wetlands was the loss of emergent saltmarsh to open saltwater systems due to manmade activities such as dredging, water control, and commercial and recreational boat traffic. There was an estimated 800 acre gain of estuarine shrub wetlands, however most of this gain came from areas formerly classified as estuarine emergent wetland. Estuarine vegetated wetlands have continued to decline over time as losses to the estuarine emergent category have overshadowed the small gains to estuarine shrub wetlands (Dahl 2006).

As an indication of the scope of developmental pressure, hence one aspect cumulative effect on EFH (coastal and tributary wetlands), NOAA Fisheries Service data show receipt of more than 20,778 individual development proposals (COE permit applications, federal projects, etc.) in North Carolina, South Carolina, Georgia, and Florida between 1981 and 1996 (See Tables 26, 27, 28, & 29). A subsample of 4,000 of these development proposals involved over 13,856 acres of various wetland habitats. Between 1996 and 2006, NOAA Fisheries Service reviewed an additional 20,896 applications to impact areas known to support EFH.

In addition to the substantial loss of wetlands in the southeastern United States, Nocholls et al. (1999) determined that by the 2080s, sea-level rise could cause the loss of up to 22% of the world‘s coastal wetlands. When combined with other losses due to direct human action, up to 70% of the world's coastal wetlands could be lost by the 2080s, although there is considerable uncertainty. Therefore, sea-level rise would reinforce other adverse trends of wetland loss.

While it is believed that most regulated activities are implemented as planned, Mager and Thayer (1986) report that limited monitoring indicate that about 20% of the projects they examined did not comply with provisions of the associated permits. Notably, most of the differences observed related more to design of structures and not the area of habitat affected. As shown in the following tables, individually and cumulatively significant impacts to EFH can be moderated through the COE regulatory program; however, significant wetland perturbations persist. This situation is largely perpetuated by (1) regulatory provisions that exempt regulation of certain wetland types and activities and (2) by severe staffing limitations within regulatory and environmental review agencies. In the absence of substantial correction in these two areas, significant wetland areas will continue to be adversely altered or eliminated, and regulatory and review agency effectiveness will be limited.

In addition to the direct cumulative effect incurred by developmental type activities, EFH is also jeopardized by persistent increases in certain chemical discharges. In that case incremental change in habitats, hydrology, and chemical inputs produced, over time, an enormous and extremely harmful result whose negative economic and social implications may far exceed any benefits related to the causative factors. Unfortunately, the effect of adding ever greater volumes and varieties of chemicals to surface waters is often insidious and resulting declines in the abundance and quality of affected and harvested resources may be slow and difficult to identify. As illustrated by Scott et al (1997), the effects may be realized at rudimentary trophic and ecological association levels in key portions (including EFH) of estuarine environments.

The rate and magnitude of anthropomorphic change on EFH, whether cumulative, synergistic, or individually large, is influenced by natural parameters such as temperature, wind, currents, rainfall, salinity, etc. Consequently, the level of threat posed by a particular activity or group of activities may vary considerably from location to location. This situation may be most acute in locations that are subject to extreme weather and oceanic conditions such as hurricanes and large waves, or where the effects of periodic or global change are most prevalent.

Nutrient over-enrichment has become a large cumulative problem for southeastern EFH. Excessive nutrients may be directly toxic. Even relatively low nitrate-nitrogen levels (as low as 3.5 uM NO3-N) have been found to cause impacts on both growth and survival in eelgrass (*Z. marina*) during spring and fall growing seasons (Burkholder et al. 1992). In contrast, Cuban shoal grass (*Halodule wrightii*) and widgeon grass (*Ruppia maritima*) were stimulated by nutrient enrichment (Burkholder et al. 1994). Eelgrass provides important brackish water habitat element for finfish, crustaceans and molluscs in North Carolina (Thayer et al. 1984). Nitrate toxicity to eelgrass in the field has yet to be documented, although nitrate concentrations in the range found to have an impact in mesocosm experiments certainly occurs in many estuarine settings.

The effects of nutrient enrichment and stimulation of toxic dinoflagellates and other algae, especially *Pfiesteria piscidida*, have been widely reported by the news media. The high abundance of small heterotrophic algae in southeastern estuaries was well known among plankton researchers during the 1980s and earlier; however, the toxic nature of *Pfiesteria* was not reported until the late 1980s (Burkholder et al. 1992, 1993, 1995; Noga et al. 1993). Analyses suggest that a large suite of *Pfiesteria*-like small heterotrophic dinoflagellates exist in most southeastern estuaries (P. Tester, personal communication). These organisms include toxic forms, like *Pfiesteria*, and may be responsible for a significant number of fish kills associated with eutrophic estuaries (Burkholder et al. 1992). Fish kills in North Carolina and Maryland have been attributed, at least in part, to these organisms (Burkholder et al. 1995), and analyses suggest that toxic dinoflagellates (and related organisms) are on the rise at a global scale (Paerl 1988; Smayda 1989; Paerl et al. 1995a).

The stimulation of toxic organism population growth by nutrient enrichment may be related to factors outside the South Atlantic region. The most notable recent case was the transport of the toxic dinoflagellate *Ptychodiscus brevis* in 1989 by the Gulf Stream and associated eddies into Onslow Bay, North Carolina. Among other impacts offshore and inshore, this seriously impacted scallop production in Bogue Sound, North Carolina (Tester et al. 1989).

Enrichment of estuarine algal and bacterioplanktonic communities by excessive nutrients is probably the most often cited example of estuarine degradation globally (Nixon 1995; NRC 1994; Ryther and Dunstan 1971). In general, the ecological pathway involves enhanced algal or bacterial production and metabolism followed by excessive oxygen uptake and subsequent deoxygenation. Anoxia and hypoxia have been identified as the fundamental problems facing Chesapeake Bay, the Gulf of Mexico, the Tar-Pamlico and Neuse River Estuaries, and other locations throughout the world (Paerl 1988).

Associated processes may be complex. For example, nutrient uptake and excessive autotroph production may result in deposition of organic material into benthic sediments, where increased sediment oxygen demand may occur at some later time. In stratified estuaries, the process may even be exacerbated by the re-release of nutrients as sediment oxygen demand is exerted in bottom, anoxic waters. The ecological effects of modification of production patterns also includes hypercapnia (elevated levels of carbon dioxide), which exerts powerful effects on some organisms (Burnett 1997).

Algal blooms in southeastern waters represent a major threat to EFH. Important algal blooms have been documented in Albemarle Sound, the Chowan River, the Tar-Pamlico River, the Neuse River Estuary, the New River Estuary, Bogue Sound, the St. Johns River, and Indian River (NOAA 1996). Algal levels can be extremely high in grossly enriched waters. A one-day survey of the Pamlico Estuary in 1988 found chlorophyll a (an algal pigment) in excess of 200 ug/l, compared to a North Carolina Water Quality Standard of 40 ug/l (15A NCAC 2B.0200). Another type of algal community stimulation occurs when airborne nitrogen from all sources, including agriculture, is deposited through wet and dry deposition into distant oceanic waters. This phenomenon was largely unrecognized until recently (Paerl 1985, 1993). Consequences of this type of deposition, where the majority of ―new‖ primary production comes from this source, can be quite significant, both on patterns in primary and secondary production and in the taxonomic makeup of that production, including the toxic forms cited above.

Among the most serious problems caused by algal blooms and other effects of over enrichment is the removal of oxygen from the water. The extent of deoxygenation in southeastern estuaries has been well documented (Rader et al. 1987; Stanley 1985). A more recent survey of the South

Atlantic region found periodic hypoxic conditions in 13 of the 21 estuaries surveyed, with bottom-water anoxia in 11 locations. Only one instance of anoxia was found along the Sea Island Coast of South Carolina and Georgia, and this was linked to stratified conditions in the Savannah River. Major anoxic events were documented in the Neuse River, the Tar-Pamlico River Estuary, the Indian River and St. Helena Sound (NOAA 1996). Although seasonal low-oxygen events may be natural in southeastern stratified estuaries, expansion in the size or persistence of deoxygenated areas has been identified for some of the above listed waters (Breitburg 1990; Rabalais et al. 1996).

Effects of deoxygenation on resident and post-larval fish, crustacean, and mollusc communities can be significant. The enormous fish kills that have plagued the Tar-Pamlico and Neuse River Estuaries have received abundant popular press since the late 1980's, and have recently been systematically analyzed (Pietrafesa and Miller 1997). This study identified 246 kills in the Pamlico during the period 1985-1995, and 73 in the Neuse, including many over 1,000,000 fish. Fish kills have also been documented in the St. John River, Florida and Charleston Harbor, South Carolina (Burkholder et al. 1995).

Another possible manifestation of nutrient over enrichment is the occurrence of chitonoclastic shell disease in blue crabs. This is believed by some to be related to water pollution (either stress incurred after exposure to anoxic conditions or cadmium). Little is known absolutely (Noga et al. 1990). In addition, fish diseases have been implicated throughout polluted estuaries, but the link to pollution remains uncertain (Noga et al. 1989).

The impact of fish kills from nutrient over enrichment is difficult to assess in terms of their effect on stocks of commercially important fish. Many of the fish killed are juveniles and Atlantic menhaden appear especially vulnerable. If these stocks are density independent, then kills translate directly into reduced adult population sizes. Vaughan (1986) found that in Atlantic menhaden, catastrophic kills, where 10% mortality events occur periodically, coupled to the accumulating 1% annual losses from permanent habitat loss, could cause a loss of 60% of the fishery within 30 years.

Impacts of atmospheric deposition of nutrients on inshore EFH is well documented, as cited above (and in Fisher and Oppenheimer 1991). Some studies suggest that nutrient enrichment from atmospheric and more traditional surface water sources can also modify planktonic and epibenthic algal communities to the detriment of fish. Changes in the phytoplankton community lead to changes in the grazer community, including the reduction or elimination of preferred prey items for planktivorous fish and fish larvae. One example is the plankton community of Western Albemarle Sound, North Carolina, where nanoplankton (the small-celled algae that are the principal food source for crustacean zooplankters) are replaced in part in some years by blue-green algae of low food value, with a concomitant elimination of the zooplankters preferred by some anadromous fish larvae and juveniles (Rulifson et al. 1986).

Besides fish, plankton, and algae, vascular marine plants also are adversely affected by excessive nutrients and their consequences. Eutrophication may cause the reduction in coverage of SAV due to shading associated with water column turbidity and the growth of epiphytic filamentous algae. Although significant die-offs of SAV have occurred in some locations in the southeast, including the Pamlico River Estuary, the direct causes of algal growth stimulation has not been established (Davis et al. 1985). NOAA's 1996 survey of impacts on SAV found declines in 5 of 21 estuaries of the southeast, including Albemarle/Pamlico Sounds, but increases in Biscayne Bay and Charleston Harbor (NOAA 1996).

A major problem with regard to assessing cumulative effects is that the majority of the methods developed to evaluate cumulative effects were developed in a terrestrial context and the applicability to marine resources and EFH is not clear. However, new analytical approaches may advance management evaluations of cumulative environmental effects. Ecological risk assessment procedures provide a useful frame for comprehensively structured analyses of anthropogenic effects (EPA 1992). These procedures involve the systematic evaluation of stressors and effects using flexible methods that foster detailed evaluations of effects (Harwell et al. 1995). The application of risk assessment principles to environmental assessments could result in more comprehensive scientific products that also carry more administrative weight. In addition, systematic applications of decision support systems can offer logically consistent methods to evaluate multiple policy alternatives. Decision support systems aid the objective identification of appropriate decision combinations according to multiple priorities and they support group-based policy evaluations (Saaty 1990; Keyes and Palmer 1993; Schmoldt et al. 1994). Combined utilization of these approaches may identify previously underemphasized factors and objective policy alternatives (Lindeman 1997b). Ultimately, they may foster more logical and explicit decision-making regarding cumulative effects issues.

A cumulative assessment of population-scale fishing effects in the Florida Keys documents that 13 of 16 grouper species, 7 of 13 snappers, and 2 of 5 grunts are recruitment overfished (Ault et al. 1998). The cumulative result of technologically enhanced fishing effort has been the accelerated removal of those top predators with most economic value. Therefore, intensive effort is now being expended to obtain species that are lower on the food chain (Pauley et al. 1998). This has serious implications; as the lower levels of the food chain decline, the chances of revival at the top of the food chain are diminished even further (Williams 1998). Top-down ecosystem degradation can result in a variety of unfavorable species abundance shifts (Goeden 1982) and, potentially, outright ecosystem collapse (Pauley et al. 1998). Further cumulative assessments of managed species in the South Atlantic may reveal long-term declines similar to those now identified in the Keys. Under such circumstances, traditional management measures (e.g., size and harvest limits), may not be adequate to rebuild sustainable fisheries for the most desirable species.

NMFS staff comments: Eliminate unneeded sections and add to sections that are too short

- *List 10 threats that represent the larger threats to fishery habitat in the SA*
- *Identify any missing major threats missing or ones to be excluded.*

Target Audience: The tone and scientific content of the document should be useful to the following target audience:

- *SERO HCD as supporting document for EFH consultations*
- *Coastal managers of other federal and state agencies (e.g., USACE, state water quality agencies)*
- *Consultants preparing EFH Assessments*

The overall goals the "Threats" Section should accomplish are:

- *Prioritize threats based on frequency, regulatory review authority, and degree of impact*
- *Provide comprehensive review of priority threats:*
    - *Describe components of action that is threatening*
    - *Describe habitats threatened*
- *Provide case studies for major threats and findings from each study's monitoring efforts (e.g., beach nourishment sand source surveys, seagrass impacts from shading associated with transportation projects)*
- *Provide recommendations to promote avoidance and minimization for each threat*

Plan needs updating for the following:

- *Number of projects NMFS consults on under Magnuson-Stevens*
- *Update population growth numbers and forecast in next 10, 20, 50 years.*

The current FEP is lacking elements that make the document useful for the target audience. These elements include:

- *Scientific-based analysis of each threat including tables, charts, graphs, etc.*
- *Specific analysis/examples of each threat for all states- currently heavy on Florida*

NMFS has developed the following list of threats and affected habitats that should be included in the revised FEP:
--Transportation: SAV, mangrove, shallow bottom, wetlands
--Navigation dredging (maintenance and new): SAV, coral, inlets, marsh, shallow bottom
--Beach nourishment (including mining offshore areas): Coral, inlets, hardbottom, shoals
--Aquaculture: SAV, shallow bottom
--Coastal development: wetlands (including St Johns River), oysters, tidal creeks, SAV
--Non-point source pollution: wetlands, oysters, tidal creeks, SAV
--Impoundments/dams (including hydropower): rivers, tidal creeks, marsh
--Alternative energy technologies: hardbottom
--Oil and gas exploration: hardbottom
--Sea level rise/Climate change: all EFH

**Agriculture not sufficient enough a threat to warrant own section (non-point section 3 pages wrong/ 1 paragraph of agriculture)*