OCS Study
BOEM 2020-002
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Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features

Volume 1: Fish Habitat Associations and the Potential Effects of Dredging on Fish of the Atlantic and Gulf of Mexico Outer Continental Shelf**:** Literature Synthesis and Gap Analysis

US Department of the Interior
Bureau of Ocean Energy Management
Headquarters (Sterling, VA)

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Volume 1: Fish Habitat Associations and the Potential Effects of Dredging on Fish of the Atlantic and Gulf of Mexico Outer Continental Shelf, Literature Synthesis and Gap Analysis

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List of Acronyms and Abbreviations

BMAPA	British Marine Aggregate Producer's Association
BOEM	Bureau of Ocean Energy Management
CEFAS	Centre for Environment Fisheries and Aquaculture Science
cm	centimeter
CPUE	catch per unit effort
DO	dissolved oxygen
DOI	Department of the Interior
EEZ	exclusive economic zone
EFH	Essential Fish Habitat
ESPIS	Environmental Studies Program Information System
FMC	fishery management council
FMP	fishery management plan
GIS	Geographic Information Systems
GLM	generalized linear model
GoM	Gulf of Mexico
Ha	hectare(s)
HAPC	Habitat Areas of Particular Concern
km	kilometer
km hr^{-1}	kilometers per hour
LDA	linear discriminant analysis
m	meter(s)
m^3	cubic meter(s)
mg L^{-1}	milligram(s) per liter
Magnuson-Stevens Act	Magnuson-Stevens Fishery Conservation and Management Act (US)
MARMAP	Marine Resources Monitoring, Assessment, and Prediction
mm	millimeter
MMP	Marine Minerals Program
MPA	Marine Protected Area
NCCOS	National Centers for Coastal Ocean Science
NEPA	National Environmental Policy Act
nGoM	northern Gulf of Mexico
nm	nautical mile(s)
NMFS	National Marine Fisheries Service
NNBF	Natural and nature-based features
NOAA	National Oceanic and Atmospheric Administration
OCS	Outer Continental Shelf
ODMDS	Offshore Dredged Material Disposal Sites
PIANC	World Association for Waterborne Transport Infrastructure
ppt	parts per thousand
psu	practical salinity unit(s)
SAV	Submerged aquatic vegetation
se	standard error
SST	sea surface temperature
TL	total length (specific fish measurement)
TSHD	trailing suction hopper dredge
TSS	Total suspended solids
USA	United States of America
USACE	United States Army Corps of Engineers
yd^3	cubic yard(s) (reported for sand dredging volumes to coincide with industry standards)

Abstract

Sand and sediment dredging in the Outer Continental Shelf of the Atlantic and US Gulf of Mexico has expanded in recent years as demand for sediments has increased. Marine sediment dredging occurs in shallow waters (≤ 50 m) and often utilizes sand shoals, where large volumes of sand can be efficiently extracted. Throughout this report, we use the term "sand" to broadly characterize sediment resources, and we recognize that sediment dredging may include a variety of grain sizes depending on the application. With the goal of understanding the effects of dredging on fish, we first synthesize the known effects of sand dredging. This includes the potential dredging effects of hydraulic entrainment, underwater sounds, suspended sediments, and substrate removal. Secondly, fish habitat use and distributions are major determinants of dredging effects. A synthesis of international literature on spatially explicit marine fish distribution models and habitat associations provides context for the latest technologies for mapping fish distributions. Documented regional habitat associations were synthesized specifically for federally managed species in the Gulf of Mexico, South Atlantic, and Greater Atlantic. Overall, the literature synthesis summarizes how dredging may affect fish species and how the distribution of fish is influenced by physical, biological, and chemical habitat factors.

1 Introduction and Background

Key Points

- Demand for sand resources is increasing rapidly in the United States (US) and worldwide. Sand is critical for beach renourishment and barrier island restoration, which help reduce damage to infrastructure from erosion and storms, and support coastal economies.
- The Bureau of Ocean Energy Management (BOEM) authorizes the use of Outer Continental Shelf (OCS) sand and gravel resources. BOEM consults with the National Oceanic and Atmospheric Administration's (NOAA's) National Marine Fisheries Service (NMFS) to assess dredging impacts to Essential Fish Habitat of federally managed species.
- A previous BOEM report by Rutecki et el. (2014) synthesized information on sand shoal geology, dredging impacts, shoal recovery time, benthic invertebrates, and provided basic information on fish related to sand shoals of the Atlantic and Gulf of Mexico (GoM).
- Volume 1 synthesizes two broad information needs: 1) What are the impacts of sand dredging on fish? 2) What are the habitat associations of federally managed fish species, particularly as they relate to geomorphology?

The demand for marine sand resources is increasing in the US (Drucker et al. 2004) and worldwide (Charlier and Charlier 1992; de Jong et al. 2014; Kim et al. 2008; La Porta et al. 2009). For example, the Netherlands alone uses an estimated 24 million m^3 (31 million yd^3) of dredged sand annually, and the amount is expected to grow with sea-level rise effects (de Jong et al. 2014). In the US, coastal and offshore sands are commonly used for beach renourishment, barrier island restoration, and wetland restoration. As human populations and associated infrastructure continue to expand in coastal zones, erosion will continue to be problematic. Recently, there has been an emphasis placed on the benefits of natural infrastructure to reduce erosion rather than sea walls or other hard structures (Ruckelshaus et al. 2016; Sutton-Grier et al. 2015). For example, the US Army Corps of Engineers (USACE) produced a report after Hurricane Sandy that demonstrated the many benefits of natural infrastructure such as living shorelines and beaches (Bridges et al. 2015). Similarly, the GoM coast benefits greatly from barrier islands that reduce storm surge (Grzegorzewski et al. 2011). In many cases, the restoration and maintenance of wide beaches, wetlands, living shorelines, and barrier islands require substantial sediment resources. Furthermore, the dredging of OCS sand shoals is likely to increase in the near future because nearshore sand resources are being depleted while renourishment cycles for beaches, emergency repairs of beaches after storms, and projected effects of sea-level rise continue to increase demand (Nairn et al. 2004).

BOEM, part of the US Department of the Interior, is responsible for the management and development of energy and mineral resources on the OCS, including renewable energy, oil and gas, and marine minerals. The Outer Continental Shelf Lands Act (1953) defines the OCS as submerged lands lying seaward of state coastal waters (3 nautical miles [nm] from most state shorelines; 9 nm from shorelines of Texas and the gulf coast of Florida) and within the US' marine jurisdiction defined by the Exclusive Economic Zone (EEZ). The EEZ is defined by a boundary 200 nm offshore of the US.

The primary purpose of our study is to inform BOEM's Marine Minerals Program (MMP), which authorizes access to OCS sand, gravel, and shell resources and negotiates these resources on a noncompetitive basis (Public Law 103-426 [43 U.S.C. 1337(k)(2)], enacted 1994). Another potential application of this review is BOEM's Renewable Energy Program, which manages proposals and leases for OCS wind energy development. The program may have considerable interest in the potential for offshore infrastructure (e.g., submarine cables) in the shallow OCS zones that are preferred for sand and gravel extraction. The USACE also uses Federal submerged lands to dispose of dredged materials from channels and inlets within approved offshore dredged material disposal sites. Much of the content developed here will be applicable to understanding how these dredge materials impact marine organisms in these areas. BOEM's authorization for use of OCS sand and gravel resources requires an analysis of impacts according to the National Environmental Policy Act (NEPA) using the best available science to understand environmental impacts. Additionally, measures may be applied to reduce potential impacts during sand dredging and conveyance to placement sites.

The Magnuson-Stevens Fishery Conservation and Management Act (Magnuson-Stevens Act), written in 1976 and amended in 1996 and 2007, has the objectives of preventing overfishing, rebuilding overfished stocks, increasing long-term economic and social benefits, and ensuring a sustainable supply of seafood. Under the Magnuson-Stevens Act, NOAA NMFS is responsible for identification and protection of Essential Fish Habitat (EFH) of federally managed marine and anadromous fishes during each of their life stages. These habitats include "those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity" (Magnuson-Stevens Act). Regional National Marine Fisheries Service programs, and their associated Fishery Management Councils (FMCs), work to define EFH. Over 1,000 federally managed species have defined descriptions and associated maps of EFH, including finfish, mollusks, crustaceans, and other marine animals and plants; birds and mammals are under the authority of the US Fish and Wildlife Service. These maps depict the EFH of species in wetlands, coral reefs, seagrasses, rivers, estuaries, and marine environments. Assemblages of species may be mapped together (e.g., reef fish, coastal migratory pelagics); highly migratory species are mapped for the entire extent of the Atlantic Ocean and GoM. The recognition of EFH provides for a process of consultation for projects permitted, or conducted by, the Federal Government to ensure that fish and their habitats are not adversely affected without full consideration of the environmental effects and possible mitigation measures. Therefore, BOEM consults with the NOAA NMFS on impacts to EFH through a written assessment. In response, NMFS recommends mitigation measures. Additionally, NMFS and FMCs may designate Habitat Areas of Particular Concern (HAPC) as types of EFH that are high priority for habitat conservation, management, and research. HAPC designations are based on the importance of the ecological function provided by the habitat (e.g., seagrasses), sensitivity or vulnerability to human-induced degradation, habitats that are slow to recover from disturbance (e.g., corals), and rarity of habitat types (Rosenberg et al. 2000). These areas do not have specific regulatory or management restrictions but are meant to bring special attention to specific areas of EFH.

As of November 2019, BOEM reports that > 125 million m^3 (> 164 million yd^3) of OCS sands have been authorized for use within 58 leases since 1995. These projects include sands used to restore 577.1 km of shoreline (**Table 1-1**) (BOEM 2019). Throughout this report, we use the term "sand" to broadly characterize sediment resources, and we recognize that sediment dredging may include a variety of grain sizes depending on the application. The quantity of sand leased has risen from < 1.53 million m^3 (2.0 million yd^3) per year to nearly 19.1 million m^3 (25.0 million yd^3) during this time (**Figure 1-1**). Many of these sites are likely to have repeated dredging events. Nairn et al. (2004) further emphasizes that nearshore sand resources are being depleted, so coastal managers may look farther offshore, increasing the demand for OCS sand deposits. All 17 Atlantic and GoM coastal states now have cooperative agreements with BOEM to identify available sand resources. As of 2019, there are no BOEM sand and gravel leases in New England, but storms and erosion have led to an anticipation of offshore sand dredging in the region. For example, Maine, New Hampshire, Massachusetts, Rhode Island, and New York all signed cooperative agreements to evaluate sand resources in 2014 following Hurricane Sandy. In addition to leasing greater volumes, BOEM is also providing access to more diverse material, such as mixed sediments, that are used in coastal restoration (J. Mallindine, BOEM, pers. comm.).

The overall strong upward trend of sand dredging necessitates a greater strategic vision for managing sand resources at a regional level rather than the site-by-site approach that has been undertaken to this point. As demand for OCS sands increases, BOEM faces complex multi-user interactions, including issues of resource allocation, cumulative impacts from repeated use, fisheries use and potential conflicts, protection of archaeological sites, oil and gas infrastructure, potential renewable energy infrastructure, and impacts on EFH (Michel et al. 2013). Marine sand dredging occurs in relatively shallow waters (≤ 50 m), often with ridge and swale complexes where large volumes of sand can be extracted over relatively small areas. The role of sand shoals as fish habitat, and as habitat that supports common fish prey species, is the subject of our literature review. Although EFH refers to multiple taxa of marine life, we refer to "fish" as being finfish in our literature review. We do cover a few specific invertebrate species, but only those known to be of importance as prey to finfish. By understanding the role of sand shoals in regard to fish habitat, BOEM will be able to make effective decisions to efficiently use these resources and to mitigate any ecological effects of sand dredging.

Table 1-1. Dredging projects in Federal waters by state 1995–2019

State	Number of Projects	Shoreline Restored (km)
Louisiana	11	68.6
Mississippi	1	5.6
Florida	22	277.8
South Carolina	7	73.4
North Carolina	3	59.2
Virginia	10	58.7
Maryland	3	15.3
New Jersey	1	18.5

Source: BOEM (2019).

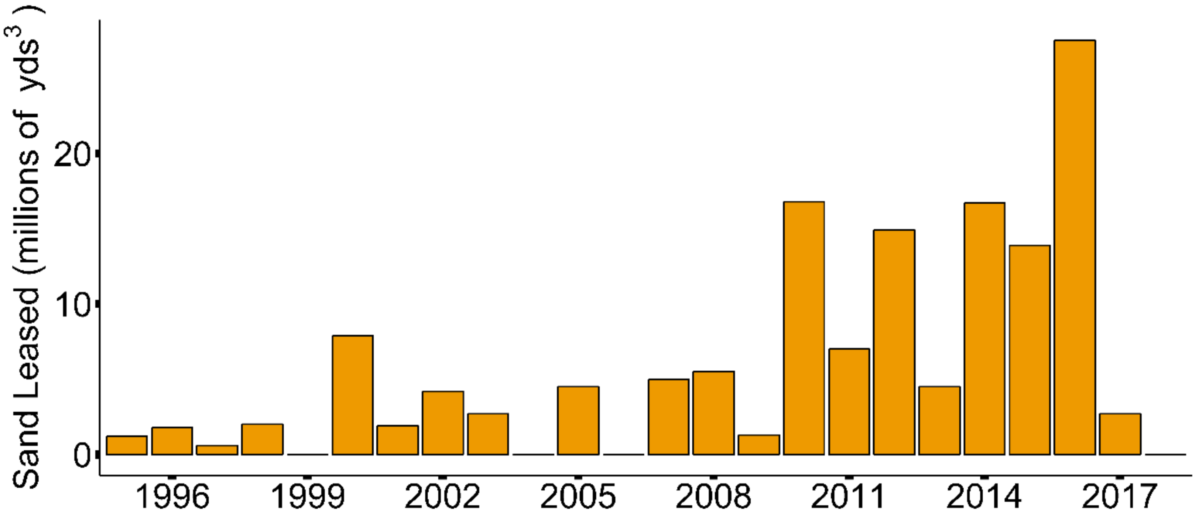


Figure 1-1. Trend in quantity of sand leased in Federal waters.
Source: BOEM Marine Minerals Information Systems sand lease data and https://www.boem.gov/MMP-in-Your-State/, accessed 2019 March.

1.1 Preceding Syntheses of Sand Shoals

Our study follows an intensive literature review by Rutecki et al. (2014): *Understanding the Habitat Value and Function of Shoals and Shoal Complexes to Fish and Fisheries on the Atlantic and Gulf of Mexico Outer Continental Shelf. Literature Synthesis and Gap Analysis.* Rutecki et al. (2014) reviewed the geology, geography, and general biological values of sand shoals; sand shoals were explicitly defined as:

> "A shoal is a natural, underwater ridge, bank, or bar consisting of, or covered by, sand or other unconsolidated material, resulting in shallower water depths than surrounding areas. The term ***shoal complex*** refers to two or more shoals (and includes adjacent morphologies, such as troughs separating shoals) that are interconnected by past and or present sedimentary and hydrodynamic processes. These complexes are also known as shoal fields."

Furthermore, Rutecki et al. (2014) suggested a classification of sand shoal systems based on the geologic origins of shoals and previous research studies (**Table 1-2**). The process of sand shoal evolution was documented as well as sedimentary processes that result in recovery of sand shoal systems to pre-dredging conditions.

Table 1-2. Classification of sand shoals from Rutecki et al. 2014.

	Shoals associated with Relict Holocene or Pleistocene Deposits		Cape-Associated Shoals	Bedform Shoals	
	Isolated Shelf Shoals	Shoal Fields	Relict Shoals	Sorted	Ridges
Synonyms	Banks	Shelf retreat massifs	Shelf retreat massifs	Rippled Scour depressions	Ridge and trough, Ridge and swale
Examples	Sabine Bank, Heald Bank, St. Bernard Shoal, Ship Shoal	Platt Shoal, Oregon Shoal, Albermarle Shoal	Cape Lookout Shoals, Diamond Shoals, Frying Pan Shoals, Wimble Shoals-(abandoned Cape)	Shoals along Wrightsville Beach shore face and inner shelf	Shoals along the inner shelf north of Cape Lookout, along MD, DE, NJ, NY inner shelves

Rutecki et al. (2014) also reviewed habitat associations and distributions of benthic invertebrates utilizing sand shoals. In Mid-Atlantic studies (offshore of New Jersey, Delaware, and Maryland), shoal crests and troughs are known to differ in invertebrate species assemblages (Byrnes et al. 2000; Cutter Jr and Diaz 2000; Slacum Jr et al. 2010b); physical differences in shoals include depth, sediment composition, and hydrodynamic regime (i.e., waves, currents). Although benthic invertebrates are well known to be distributed in relation to sediment texture (Rutecki et al. 2014), few fish studies have investigated the effect of substrate (with the exception of coral substrates). Benthic invertebrate species reported as having important economic value include blue crab, shrimp, Atlantic surfclam, ocean quahog, American lobster, sea scallop, hard clam, Florida stone crab, Gulf stone crab, spiny lobster, and slipper lobster (Rutecki et al. 2014). These species are likely to be important for higher trophic levels, and particularly for fish species that directly depend on them as prey. Additionally, Slacum et al. (2010a) reported abundant squid in Mid-Atlantic shoal systems, and squid compose a substantial part of the diet of predatory fish (Bowman et al. 2000; Watanabe et al. 2004). In regard to fish, Rutecki et al. (2014) summarized major studies of sand shoals and listed species specifically documented in close proximity to shoals for the GoM, South Atlantic, North- and Mid-Atlantic. They also recognized that shoals serve as: 1) refuges for juvenile fish and schooling planktivores, 2) habitat for species adapted to dynamic substrate and as a trophic base for demersal fish, and 3) spawning sites for some demersal fish and schooling planktivores

(CSA International Inc et al. 2010; Gilmore 2008). Rutecki et al. (2014) also recognized substantial knowledge gaps and identified future research priorities regarding the distribution of fish in or near sand shoals, effects of keystone species utilizing shoals, temporal changes in fish communities, use by highly migratory fish species, habitat use of various life stages, and direct impacts of sand dredging.

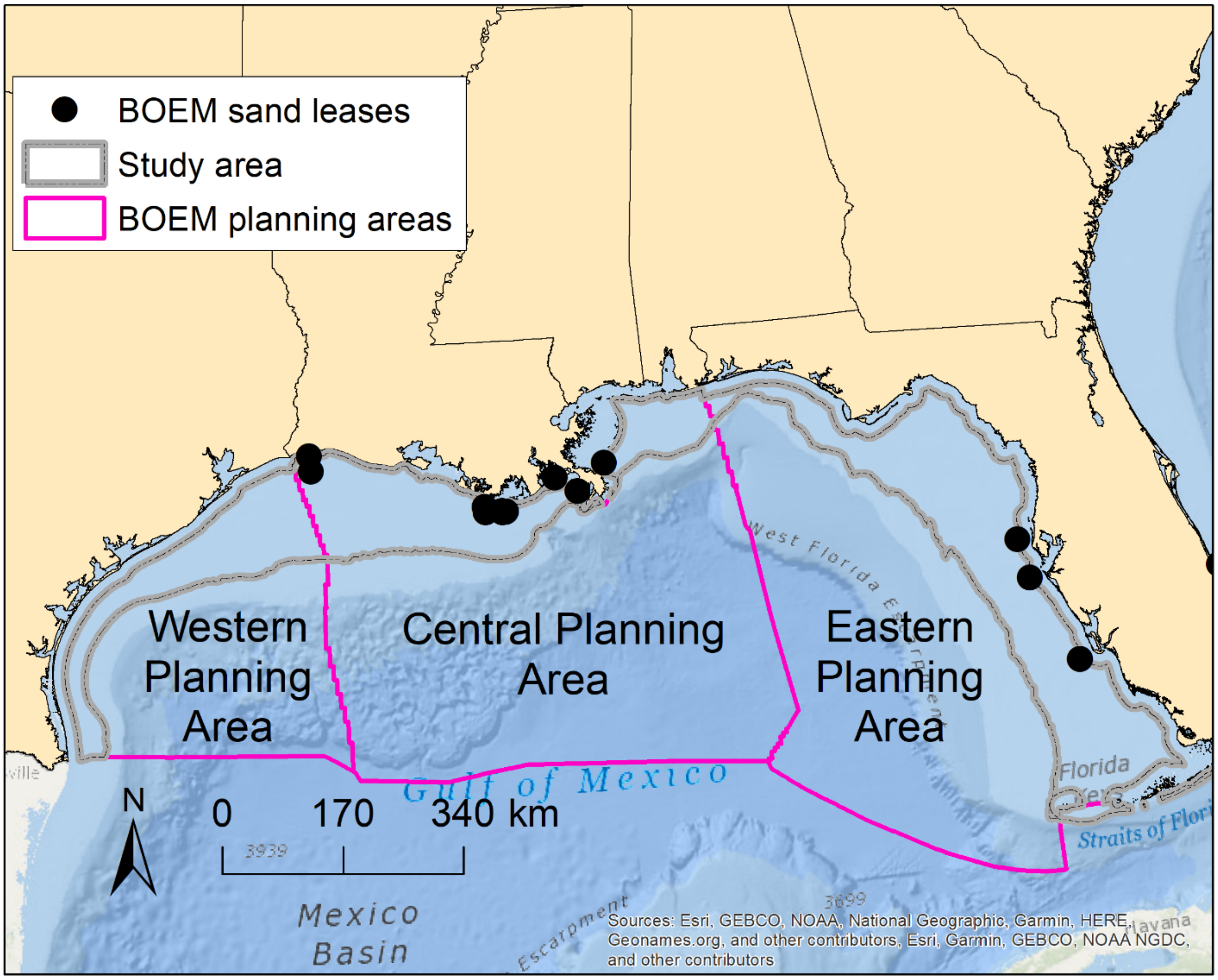
1.2 Scope of Literature Synthesis and Objectives

In the Magnuson-Stevens Act, ***fish*** are defined as *finfish, mollusks, crustaceans, and all other marine animal and plant life other than marine mammals and birds*. To date, the effects of sand dredging on marine resources has primarily focused on benthic infauna and epifauna invertebrate communities, which are directly impacted by substrate removal (Crowe et al. 2016; Newell et al. 1998; Palmer et al. 2008). In the literature synthesis presented here, we focus on finfish, particularly those that have EFH. However, we do address federally managed shrimp species in the GoM because of their importance as prey of fish and their value in commercial fisheries. Likewise, invertebrate ecology and succession is briefly reviewed in the context of substrate removal and recovery because of their importance in the food web of fish. The scope of our review does not cover sea turtles and marine mammals. These topics are extensive and beyond the scope of this review.

Unconsolidated sediments of the OCS, including sand shoals, are the vast majority of the seafloor, yet they are poorly studied. Kritzer et al. (2016) used expert opinion to rank the importance of marine benthic habitats along the Atlantic Coast in terms of fisheries. They found unconsolidated sediment was ranked as the most important benthic habitat in the North Atlantic, Mid-Atlantic, and South Atlantic (ranked second in south Florida). These shallow waters are critical to juvenile fish (Diaz et al. 2003; Steves et al. 2000; Walsh et al. 2006) and many commercially valuable fish species. Therefore, a further investigation into the effects of sand dredging on OCS fishes, and particularly their shoal habitats, may lead to a better understanding of individual and population-level effects to fish. The focus of our literature synthesis is on shallow (≤ 50 m depth) federally managed waters of the OCS (**Figure 1-2, Figure 1-3**, and **Figure 1-4**). These respective maps define our study area in each region. We do not attempt to review fish habitats in deeper waters, state-managed waters, or estuaries. The 50-m depth limit covers the maximum possible extent for sand dredging.

The objectives of our literature synthesis were to:

1) Synthesize the main effects of marine sand dredging on fish, including the severity, duration, and extent of the effects.
2) Summarize international literature on spatially derived fish habitat associations that are typically mapped and modeled for EFH and other applications.
3) Summarize fish habitat associations in the GoM, South Atlantic, and Greater Atlantic for federally managed species, particularly for common species on shoals and soft sediment substrates.



Sources: Esri, GEBCO, NOAA, National Geographic, Garmin, HERE, Geonames.org, and other contributors, Esri, Garmin, GEBCO, NOAA NGDC, and other contributors

Figure 1-2. The GoM study area defined by the landward boundary of Federal waters and the seaward boundary of the 50-m depth contour line.
Sand lease areas, as of the 2016 fiscal year, are shown as centroids for reference (not to scale).

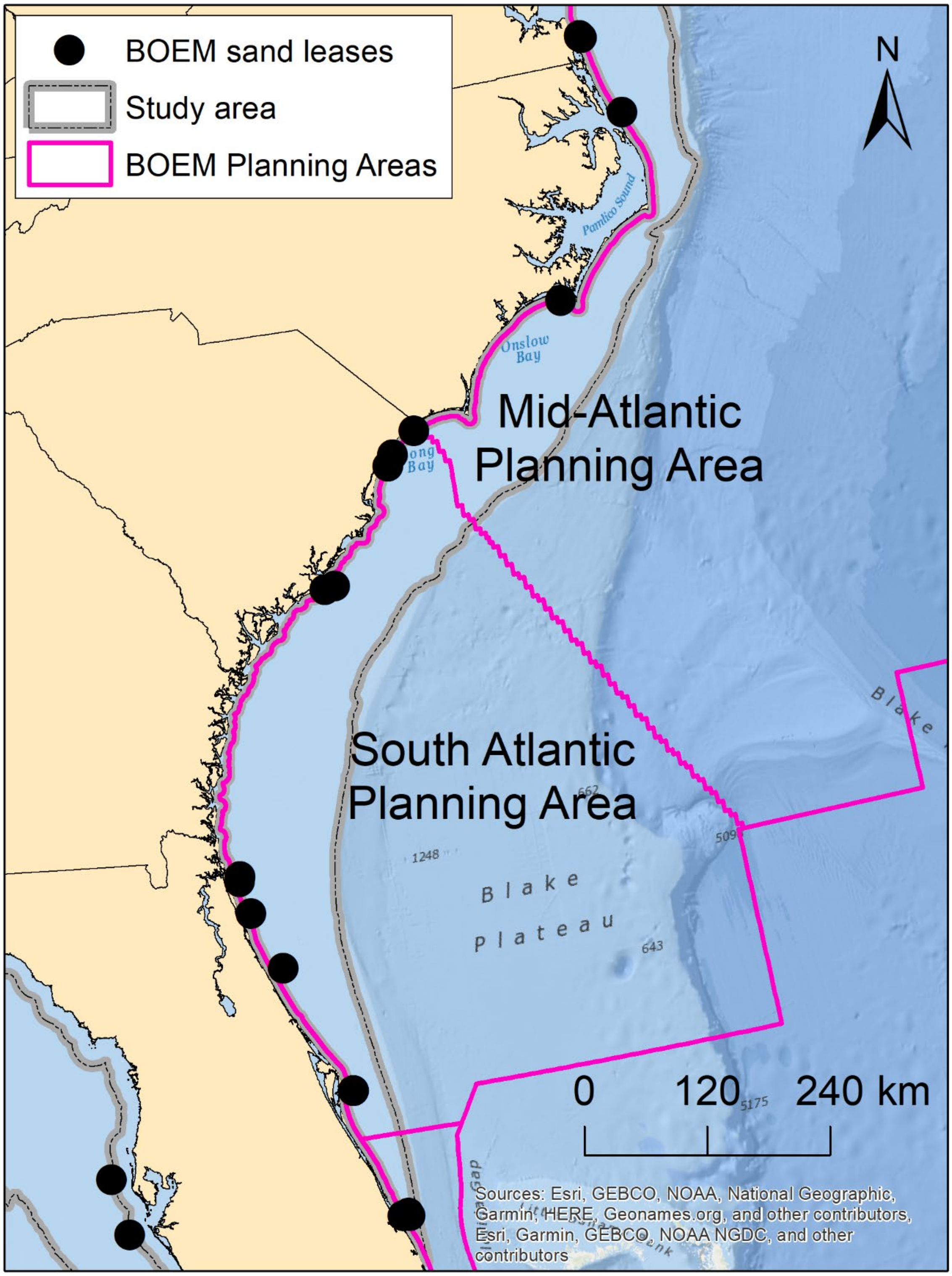
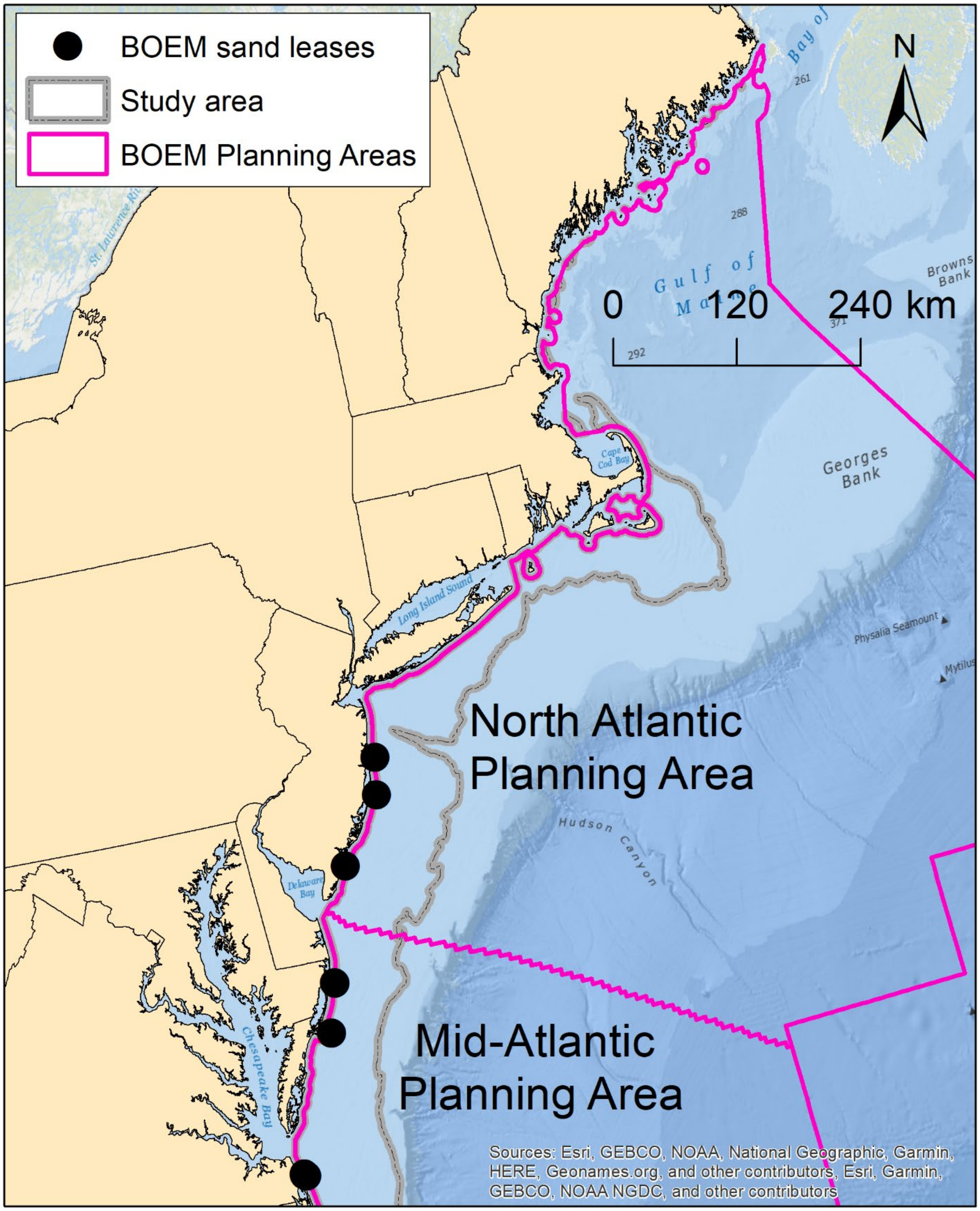


Figure 1-3. The South Atlantic study area defined by the landward boundary of Federal waters and the seaward boundary of the 50-m depth contour line.
Sand lease areas, as of the 2016 fiscal year, are shown as centroids for reference (not to scale).



Sources: Esri, GEBCO, NOAA, National Geographic, Garmin, HERE, Geonames.org, and other contributors, Esri, Garmin, GEBCO, NOAA NGDC, and other contributors

Figure 1-4. The northeast Atlantic study area defined by the landward boundary of Federal waters and the seaward boundary of the 50-m depth contour line.
Sand lease areas, as of the 2016 fiscal year, are shown as centroids for reference (not to scale).

1.3 Literature Search Methods

We used databases and various search engines to identify appropriate white papers, grey literature, government reports, and peer-reviewed publications. Whenever possible, peer-reviewed publications were cited. Preference was given to citing the most current research studies, summaries provided by review papers, and research that has been highly cited on the topic of relevance. Of particular relevance to the documentation of dredging impacts, the most current evidence presented by research studies helps to support, or refute, speculation or informal personal observations that were previously the only information available to interpret. Review papers contain dozens to hundreds of citations for papers of relevance, and we have attempted to derive summaries of the findings, concepts, and applications as concisely as possible. When necessary to provide case studies or derive specific metrics, we obtained original works cited by the most recent papers.

Search engines used included Google Scholar, Web of Science, and government agency databases (USACE Dredging Operations and Environmental Research, USACE Research and Development Center, BOEM's Environmental Studies Program Information System [ESPIS]). These searches found research from the British Marine Aggregate Producer's Association (BMAPA), Marine Aggregate Levy Sustainability Research, and the Centre for Environment Fisheries and Aquaculture Science (CEFAS). All fields, including the title and abstract, were searched for each set of terms. For important or highly cited articles, we checked more recent papers that cited the original article to ensure that the synthesis included the most recent articles. References from Rutecki et al. (2014) were checked for more in-depth information on fish not already covered by their literature synthesis. In addition, details of fish habitat associations documented in FMC and EFH documents are presented in *Volume 4: ShoalMATE (Shoal Map Assessment Tool for EFH) Manual and Data Tables.*

Search terms and phrases that were queried in the databases included a combination of taxonomic groups or species, environmental characterizations, and geography. Geography-specific searches included *Gulf of Mexico, South Atlantic, Mid-Atlantic, Atlantic, or Atlantic Ocean.* Species-specific searches were also conducted. A separate search of international literature is further described below.

Examples of keywords for literature search:

Dredging effects:
'dredging' AND 'entrainment'
'dredging' AND 'fish' AND 'marine'
'dredging' AND 'marine'
'dredging' AND 'sound'
'dredging' AND 'sound' AND 'fish'
'fish' AND 'sound' OR 'noise'
'marine' AND 'fish' AND 'sound' OR 'noise'
'dredging' AND 'suspended sediments'
'dredging' AND 'suspended sediments' AND 'marine'
'fish' AND 'marine' AND 'suspended sediments'
'fish' AND 'suspended sediments'
'corals' AND 'suspended sediments'
'hard bottom' AND 'suspended sediments'
'marine dredging' AND 'recovery'

Fish species:
<species name> AND 'Gulf of Mexico' AND 'habitat'
<species name> AND 'Gulf of Mexico'
<species name> AND 'Atlantic' AND 'habitat'
<species name> AND 'Atlantic'
'shark' AND 'Gulf of Mexico' AND 'habitat'
'shark' AND 'Atlantic' AND 'habitat'

2 Effects of Sand Dredging on Fish

Key Points and Knowledge Gaps (*gaps are in italics*)

- Fish are most vulnerable to dredging effects during egg or larvae stages, spawning periods, or during migration, when compared to other life stages. *Demersal species have been suggested to be more vulnerable than pelagic, though evidence is lacking.*
- Entrainment of benthic fish and invertebrates occurs locally during dredging. *A few studies have examined entrainments rates of fish in estuaries, but rates in marine ecosystems are lacking.*
- Turbidity occurs during and shortly after dredging activity, but resuspension of sediments at the borrow area has reoccurred 1.5 years post-dredging. Studies have regularly found turbidity to influence a 3-km radius around dredging, though concentrations are not high enough to cause direct fish mortality.
- Sedimentation may threaten hard bottom and coral reef fish habitats because of burial and coral mortality.
- Underwater sounds during dredging are not severe enough to cause fish mortality, but sounds may persist above ambient conditions for 400 m to 2.7 km.
- *Avoidance responses (including response distance) of fish to underwater sounds and turbidity are unknown.* Fish behavioral responses will determine habitat loss, disruptions to migration, and other impacts.
- Substrate removal by dredging may result in bathymetric depressions or more homogeneous, flattened topography within the footprint of dredging.
- Recolonization by early successional benthic invertebrates and restoration of the density of individuals have been documented after one year post-dredging, while recovery of the full species assemblage ranges from 2.5 to > 7 years. Full recovery of invertebrate species tends to correspond with a return to the pre-dredging sediment grain size.
- More frequently, or intensively, dredged substrates may take double the time (~15 years) to recover compared to less intensively dredged sites. Frequent dredging tends to change sediment grain size more dramatically than less intensively dredged sites.

Two types of dredges are commonly used for sand dredging in marine environments, a trailing suction hopper dredge (TSHD) and a cutterhead dredge. The TSHD are more mobile and commonly used to dredge offshore sands, especially in the Atlantic; cutterhead dredges are better suited for calm seas and work closer to shore with some offshore work in the GoM. The TSHD vessel works by moving at 3–5 km hr^{-1} (1.5–3 knots) as an onboard dredge pump creates a suction that is transmitted through 1–3 pipes leading to each pipe's draghead, which are 1.5–4 m in width and lie on the seafloor (Michel et al. 2013) (**Figure 2-1**). Sand is suctioned through the trailer arm pipe and into the hopper located in the hull of the ship. The dredge then moves to a stationary in-water pump-out station to pump sand to shore via pipelines. A hydraulic cutterhead dredge agitates the sediments as the cutterhead rotates (**Figure 2-2**). To allow the cutterhead to swing back and forth, anchors, studs, or a stud pole, are used to moor the vessel. A cutterhead may pump sand directly to shore or use pipelines. For both dredges, additional boats are used to support operations, conduct monitoring, and to move anchors for cutterhead dredging. Further details of offshore dredging vessels and their operation are reviewed elsewhere (CSA International Inc et al. 2010; Michel et al. 2013).

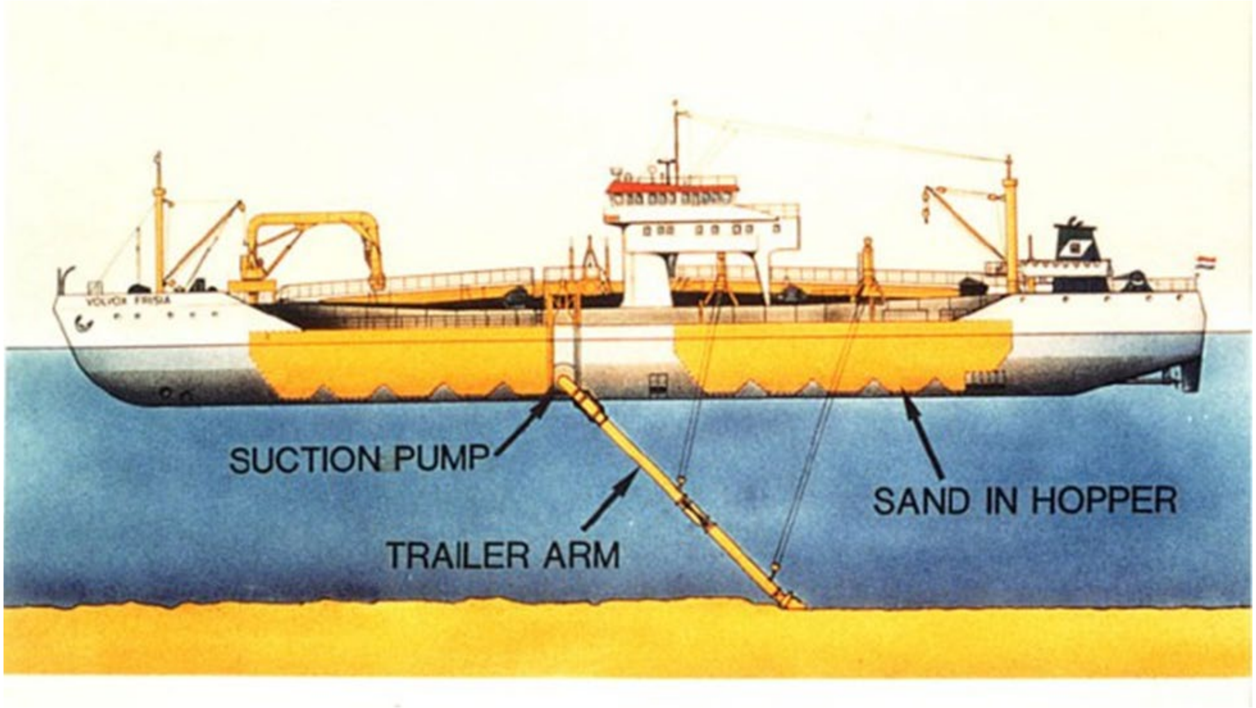


Figure 2-1. Main components of a TSHD. Additionally, the draghead is attached to the bottom of the trailer arm and is where the hydraulic suction is generated.
Source: Adapted from https://oceanandairtechnology.wordpress.com/2013/06/11/trailing-suction-hopper-dredger/

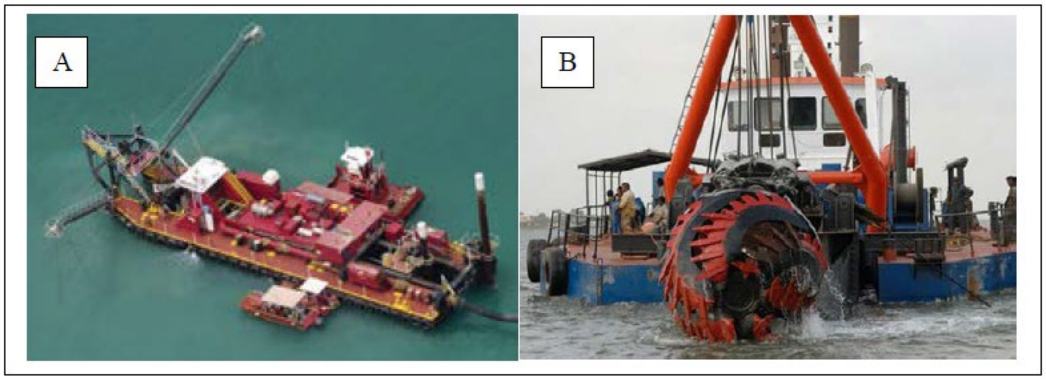


Figure 2-2. A) Cutterhead suction dredge and B) cutterhead.
Source: Adapted from Michel (2013) and www.dredgepoint.org.

The short-term effects of dredging include entrainment, human-made sounds, loss of prey/food web effects, suspended and resuspended sediment plumes, sedimentation of the seafloor, and release of contaminants (Kim et al. 2008; Suedel et al. 2008; Wenger et al. 2017). Wenger et al. (2017) provides a comprehensive review of dredging effects on fish in freshwater, estuarine, and marine environments. Some of these effects of dredging have been more commonly studied in environments like estuaries (e.g., entrainment of fish, release of contaminants), and information may be limited in offshore marine environments. Therefore, we note estuarine studies when necessary; no specifics of freshwater or restricted channel studies have been included because types of dredging, entrainment rates, and other factors differ dramatically in those systems.

Given the uncertainty surrounding the effects of dredging, seasonal restrictions on dredging operations are sometimes implemented in the US. Of the dredging operations implemented by the USACE from 1987 to 1996, time window restrictions for dredging to address biological concerns were implemented in 85% of Atlantic operations, and for 18% of cases in the GoM (Dickerson et al. 1998). Seasonal restrictions on dredging are most likely to be implemented for protected species; however, a variety of species have been a basis for seasonal restrictions. Fish species used as a basis for seasonal dredging restrictions include American shad, Atlantic tomcod, blue crab, Gulf sturgeon, shortnose sturgeon, striped bass, winter flounder, brown shrimp, pink shrimp, and white shrimp (Dickerson et al. 1998) as well as Pacific herring (Suedel et al. 2008). Unfortunately, the lack of available data may continue to result in inefficient restrictions on dredging activity, which can drive up costs, increase transportation distances, and delay projects (Dickerson et al. 1998).

2.1 Effects of Hydraulic Entrainment on Fish

2.1.1 Entrainment Rates of Fish and Invertebrates

Hopper and cutterhead dredges use hydraulic suction fields to obtain and transport unconsolidated sediments from the seafloor of aqueous ecosystems. These actions, occurring at the seafloor, may result in the entrainment of benthic fish and invertebrates, as defined as the direct uptake of organisms due to the hydraulic suction field generated by a draghead or cutterhead dredge (Reine and Clarke 1998). The effect of entrainment of fish results in mortality of most individuals. However, entrainment is limited to the duration of the dredging activity, and the effects are localized to the direct area affected by the draghead or cutterhead of the dredge vessel.

Entrainment rates are determined by species' population density at the time of dredging, footprint of area affected by the dredge, and vulnerability of the fish species present. The quantification and publication of entrainment rates of fish and invertebrates in dredging gear within marine ecosystems are limited. Most entrainment studies have been conducted at power plants or water-diversion structures in freshwater environments where anadromous species are particularly at-risk over continuous time periods (e.g., Kelso and Milburn 1979; Mussen et al. 2014). The relatively short duration of dredging activities and the relatively open nature of marine settings (rather than restricted waterways like channels or rivers) makes entrainment comparisons unrealistic in many situations. Therefore, we focus on information derived from entrainment studies of estuaries, which are more open and include similar species (or the same species) to the marine environment. We note that marine studies of entrainment rates are still needed, as estuaries or estuarine navigation channels are still very different than offshore marine borrow areas. Studies are also needed in the Atlantic and GoM to better understand entrainment rates and species affected in these regions. Additionally, marine borrow areas may provide opportunities for mitigation that may not be possible for navigation channels.

For the closest approximation of entrainment rates with applications to marine environments, Reine and Clarke (1998) provides an exhaustive review of documented entrainment rates, including several estuarine studies. The Dungeness crab has received considerable attention because they are commercially valuable, congregate in navigation channels, and migrate in and out of estuaries. As noted, navigation channels are far different than offshore marine environments. Dredging must be thorough in navigation channels and mitigation measures may be more limited. However, these are the most comparable entrainment rates currently available. The entrainment rate for combined studies of Dungeness crabs ranges 0.03–0.45 crabs/m^3 (0.04–0.59 crabs/yd^3) of dredged material for adult crabs; comparatively, juvenile crabs were entrained at a higher rate (range 0.24–8.24 crabs/m^3 (0.31–10.8 crabs/yd^3), mean = 3.17 crabs/m^3) (4.15 crabs/yd^3) compared to the adults (Reine et al. 1998). The authors had limited data that suggested entrainment mortality rate differed by crab size, as those >75 mm had observed mortality of 86% while crabs of 7–10 mm had mortality estimated at 5%. Entrainment rates were twice as much for male crabs compared to females, probably because of the timing of female emigration from the estuary. Plus, seasonal changes of crab densities are substantial (Wainwright et al. 1992). During entrainment studies focused on Dungeness crabs, benthic sand shrimp were the most commonly entrained species, and the rate of entrainment was reported as 2.58 shrimp/m^3 (3.37 shrimp/yd^3) of dredged material for a TSHD (Armstrong et al. 1981). There are no entrainment data documented for the commercially valuable shrimp species of the Atlantic Ocean or GoM (Reine and Clarke 1998). Overall, the impact to shrimp are a concern because they are the prey base for many fish species and support a large fishery. In regard to fish captured by the Dungeness crab studies in Pacific Coast estuaries, entrainment rates reported by Reine and Clarke (1998) are presented in **Table 2-1** (Larson and Moehl 1990; McGraw and Armstrong 1990).

A specific concern for the Atlantic is the potential entrainment of economically and ecologically important horseshoe crabs, although the primary concern has been with dredging navigation channels or inlets where the species may congregate (Ray and Clarke 2010). Adults span much of the Mid- and South Atlantic, and the eggs and larvae provide a critical prey base for long-distance migratory shorebirds (Botton et al. 1994), particularly the federally threatened red knot (Karpanty et al. 2006). Adult horseshoe crabs are found in estuaries and shelf habitats at depths of < 30 m, although they have been found at a 290-m depth offshore of Cape Hatteras, North Carolina (Botton and Ropes 1987). They have been documented as users of sandy shoals (Rutecki et al. 2014), although it remains unclear if they spawn at offshore shoals. In estuaries, observers documented entrainment of horseshoe crabs as ranging from 0.000003–0.004 horseshoe crabs/m^3 (0.000003–0.005 horseshoe crabs/yd^3) of dredge material (10–5,521 horseshoe crab individuals entrained per project) (Ray and Clarke 2010). Study sites in their study included harbors, channels near harbors, an Atlantic Ocean channel, and a beach renourishment project near Virginia. In their study, differences in entrainment rates were ascribed to location, time of year, type of equipment, and specifics of the operation (Ray and Clarke 2010). Localized mortalities from entrainment may exacerbate stresses that are already incurred by horseshoe crab populations, such as harvesting for bait and mortality related to bleeding individuals for biomedical purposes.

Table 2-1. Entrainment rates of Pacific Coast estuary studies.

Common name	Scientific name	Fish entrained per m^3 of dredged material
Anchovy	*Engraulididae* spp.	0.0008–0.0061
Northern anchovy	*Engraulis mordax*	0.0138
Herring	*Clupeiformes*	0.0061
Flounder, sole, sanddab, flatfish	*Atheresthes stomias*, *Platichthys stellatus Pleuronectes vetulus*, *Psettichthys melanostictus*, *Lyopsetta exilis,* *Citharichthys* spp., *Pleuronectiformes* spp.	0.0008–0.0581
Pacific sand lance	*Ammodytes hexapterus*	0.0275–0.454
Pacific sandfish	*Trichodon trichodon*	< 0.0008–0.0015
Surfperch	*Embiotocidae*	≤0.0008
Pipefish	*Syngnathidae*	0.0061
Big skate	*Raja binoculata*	< 0.0008
Longnose skate	*Raja rhina*	0.0023
Spiny dogfish	*Squalus acanthias*	< 0.0008

Source: Modified from Reine and Clarke (1998).

2.1.2 Factors Affecting Vulnerability of Fish to Entrainment

Broad concepts and characteristics that render a species vulnerable to entrainment in estuaries are likely to be comparable in marine ecosystems, although the rates may differ. In fact, Wenger et al. (2017) suggests that general management guidelines could be combined for freshwater, estuarine, and marine ecosystems. Kim (2008) suggests species vulnerable to entrainment are benthic organisms, including shellfish (e.g., blue crab, shrimp) and demersal fish (e.g., flounder, flatfish) because of their position in the water column. However, documented evidence to support these assumptions is limited. Similarly, Reine and Clarke (1998) notes concerns over potential entrainment of anadromous fish, shrimp, crabs, shellfish, and threatened/endangered species; species of concern included Gulf and shortnose sturgeon, salmonids, American shad, blue crab, oyster larvae, and winter flounder. Again, supporting data are lacking. Drabble (2012) developed a sensitivity index and projected entrainment rates for marine fish based on qualitative rankings of potential for entrainment. Factors in the index were a) previous evidence of entrainment, b) sensitivity to sound (i.e., ability to avoid impact area), c) ability to move quickly, d) burial behavior, and e) fecundity. For example, sand lance bury themselves in response to disturbance, do not have a swim bladder, and may hibernate; thus, they are extremely vulnerable to entrainment. Entrainment rates were projected with the sensitivity index combined with the dredge production rate, footprint, and speed as well as distribution data for fish species (Drabble 2012).

In a meta-analysis incorporating dredging studies in all aquatic environments, eggs and larvae have shown more mortality effects compared to other life history stages (Wenger et al. 2017). This may be because of their size and inability to move quickly away from dredging disturbances. Although a variety of external factors have been proposed to affect entrainment rates (bottom depth, dredge speed, flow-field velocities, volume of dredged materials, and direction of dredging with reference to tidal flow), these factors have been shown to have little influence (Reine and Clarke 1998). Reine and Clarke (1998) reviewed the potential for biological impacts from hydraulic dredging and emphasized the following:

- Bottlenecks or congregations of fish in migration corridors (e.g., anadromous fish) are particularly vulnerable
- Because of the timing of migration, particularly for anadromous fish and crabs, the male to female ratio of entrained individuals may differ
- Life stages involving dormancy or limited movement (e.g., egg or larvae stages, blue crab dormancy) are particularly vulnerable
- Juveniles may be more vulnerable to entrainment compared to adults
- Demersal fish are likely more vulnerable, though pelagics have been regularly entrained in the past (e.g., anchovy, herring)
- Small and large fish may be equally vulnerable (Armstrong et al. 1981)

2.2 Effects of Human-made Sound on Fish

2.2.1 Measures of Underwater Sound

Before detailing underwater sounds produced by dredging and dredge vessels, we first describe characteristics of sound and how underwater sounds are typically measured. A fundamental measure of sound is its loudness as quantified by the amplitude of sound waves at a range of frequencies and measured in units of decibels (dB) or micropascals (µPa). Each 10 dB increase represents a 10-fold increase in sound pressure. As an example, humans can hear a minimum of approximately 0 dB (or 20 µPa), and sounds greater than 130 dB are generally considered painful to humans. Studies of underwater sounds often use decibel measurements as a ratio of root mean squared (rms) (average sound over an acoustic event) of pressure to background reference pressure. In water, the reference pressure is 1 µPa, and therefore, measurement units are dB re 1 µPa for studies conducted underwater. Sound frequency, or pitch, is the number of cycles per second with hertz (Hz) as the unit of measurement. For reference, humans hear in the range of 20 Hz to 20 kilohertz (kHz). Taken together, the frequency and amplitude measures describe what an organism can hear and how loud the sound is projecting.

Studies of underwater sounds typically use hydrophones placed at a few depths and over a range of distances from the source to measure the received level, and a mathematical interpolation of decibels by distance can show how sound varies by distance from the source. Conversely, when the source level of sound cannot be measured directly, this model can also be used to estimate the source level (at 1 m from a sound-producing object). A common measurement standard of comparison for underwater sounds is that of ambient sound levels, which represents background sound. Natural ambient sounds include waves, wind, or living organisms and are dominant where human activities are scarce, such as the open ocean. In contrast, human-made sounds such as vessel propeller cavitation and generators are likely to be a dominant source of ambient sounds in harbors or in close proximity to human activities. Sound transmission may be highly variable in shallow waters due to environmental conditions such as seabed type, bathymetry, salinity, and stratification (Reine et al. 2014a). In a recent study, Halvorsen and Heaney (2018) found sound traveled further over sand substrates compared to mud at 10–30 m depths; however, differences were more pronounced among water depths of 10, 30, and 100 m compared to substrate type.

2.2.2 Underwater Sounds Produced by Dredging-related Activity

Dredging operations emit underwater sounds as vessels are in transit as well as during the dredging activity itself. Studies of underwater sounds produced by dredging operations have recently received increased attention, though only a few studies have specifically examined sand dredging. More commonly, studies have examined underwater sounds produced by the extraction of aggregates (i.e., sand/gravel mixtures) or dredging that involves the breaking of rocks; these operations are conducted worldwide to obtain materials for construction. In regard to hydraulic dredges, sounds are produced from the following sources (Reine et al. 2014b; Reine and Dickerson 2014; Robinson et al. 2012) (see **Figure 2-1** for reference):

- Removal of material from the substrate when the draghead, or cutterhead, contacts the seafloor
- Pumps and impellers driving suction of materials into the pipes from the seafloor
- Movement of materials through the pipes
- Loading of materials into the hopper and overflow of undesirable materials
- Dredge machinery such as winches, generators, thrusters; for cutterhead dredges, propeller-induced cavitation movements of spuds and anchors
- Offloading of materials to placement site
- Water turbulence around ship's hull
- Echosounding instruments
- Supporting vessels

Of research quantifying the underwater sounds emitted by dredging operations, Reine et al. (2014b) and Robinson et al. (2012) provide the primary studies that examine sand dredging in marine environments (**Table 2-2**). In a comparison of underwater sounds produced by sand and gravel mining, Robinson et al. (2012) showed TSHD conducting sand dredging in the United Kingdom generally had a greater decibel level than gravel mining, although gravel mining did result in greater decibels at the highest frequencies.

Table 2-2. Underwater sounds emitted by trailing suction hopper dredges. Distances of received levels differed by study as indicated.

Study	Dredge activity	Estimated sound at source (dB re 1 µPa at 1 m)	Observed sound (dB re 1 µPa at distance measured from source)
Reine et al. (2014b)	In transit to borrow site	168–174	133–137 at 50 m
Reine et al. (2014b)	During dredging	171–174.5	141.2–146.8 at 50m; < 130 at 1,150 m; Approached ambient at 850–2,700 m
Reine et al. (2014b)	Return transit with full hopper	NA	130-140 at 100 m
Robinson et al. (2012)	During dredging	183	139 at 100 m

The Reine et al. (2014b) study was conducted in the Mid-Atlantic, USA, and the Robinson et al. (2012) study was conducted in the United Kingdom.

Although dredging studies have included rivers, navigation channels, sounds, estuaries, and marine environments (Clarke et al. 2003; Reine et al. 2014a; Reine and Dickerson 2014), the results of sound production are relatively consistent (**Table 2-2**). Maximum sounds are commonly 140–150 dB re 1 µPa measured at 40–50 m from the source vessel, sound level at the source range from 161–183 dB re 1 µPa, and ambient sounds range from 112–119.5 dB re 1 µPa (e.g., **Figure 2-3**). Generally, strong declines in decibels are expected within the first 100–300 m of the source (Reine et al. 2014a; Reine et al. 2014b), and underwater sound levels remain elevated for a distance of 400–2,700 m from the dredging vessel **(Figure 2-3, Table 2-3**).

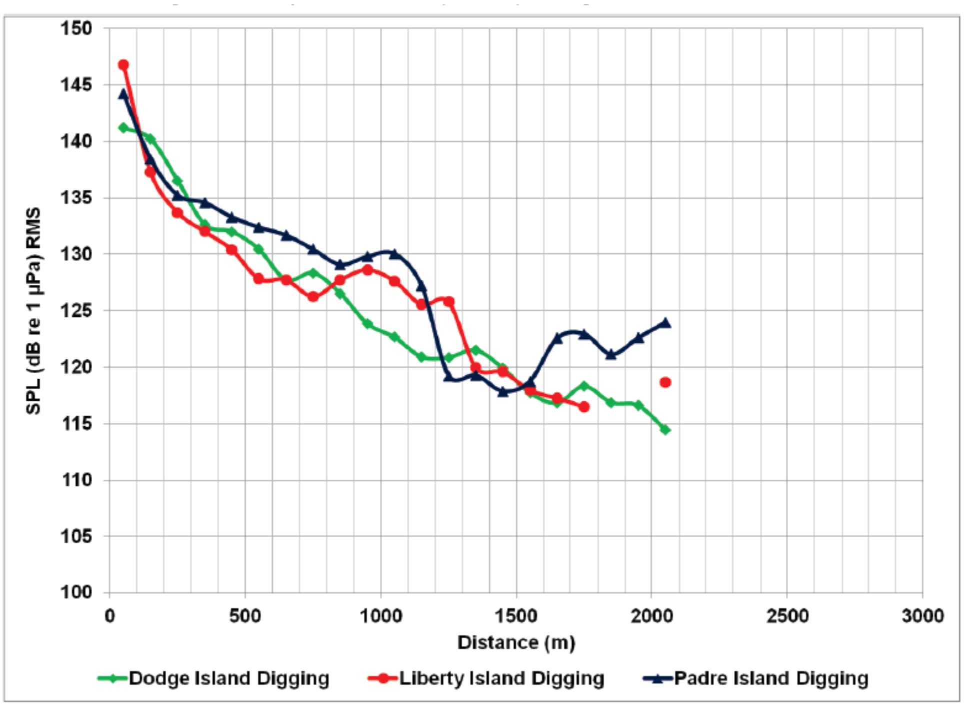


Figure 2-3. An example of underwater sound emitted, as measured in sound pressure levels (SPL) (y-axis), at incremental distances (x-axis) from three trailing hopper suction dredges operating in offshore marine waters of the Atlantic Ocean near Wallops Island, Virginia.
Source: Image from Reine et al. (2014b).

Table 2-3. Summary of studies reporting underwater sounds produced by hydraulic dredges.

Study	Type of dredge (hopper capacity if applicable)	Location, ecosystem	Estimated sound at source (dB re 1 µPa at 1 m)	Maximum sound observed (dB re 1 µPa)	Mean ambient conditions (dB re 1 µPa)	Distance for dredge noise to reach background ambient conditions
Reine et al. (2014b)	Hopper (2,754–5,003 m^3 or 3,603–6,544 yd^3)	Virginia, marine	161–178	141.2–146.8 (at 50 m)	112	850–2,700 m
Robinson et al. (2012)	Hopper (1,418–4,832 m^3 or 1,855–6,320 yd^3)	United Kingdom, marine	156–183	139 (at 100 m)	112	NA
Clarke et al. (2003)	Hopper (8,517 m^3 or 11,140 yd^3)	Mississippi, sound	NA	142 (at 40 m)	74	NA
Unpublished data from Reine et al. (Reine and Dickerson 2014)	Hopper (994 m^3 or 1,300 yd^3)	Maine, Kennebec River (removing sand shoals)	172–180	NA	NA	700 m
Reine and Dickerson (2014)	Pipeline cutterhead	California, shipping channel (river, estuary)	157.4	148.3 (at 87 m)	119.5	400–480 m, ~122 dB
Reine et al. (2014a)	Cutterhead (fracturing rock; no material movement)	New York / New Jersey Harbor, estuary	181	151 (at 100 m)	117.1	~2 km; Remained ≥132 dB re 1 µPa at 740 m from vessel
Clarke et al. (2003)	Cutterhead	Mississippi, sound	NA	112	74	Inaudible at ~ 500 m
Reine et al. (Reine et al. 2014a)	Commercial vessels	New York / New Jersey Harbor, estuary	NA	Ferry approach= 136 (at 750 m) Container ship = 141.7 (at 321 m)	117.1	NA

Notes: Sounds of other vessels are reported as reference points. Studies include both sand and aggregate dredging.

2.2.3 The Effects of Underwater Sounds on Fish with Implications for Dredging Impacts

Although fish respond most directly to the particle motion generated by sounds, most studies focus on the effect of sound waves (i.e., pressure) in underwater environments (Nedelec et al. 2016). Thus, our review focuses on the effects of sound waves. For fish, the otolith organs are the auditory portion of the ear. Sound pressure is also detected by the swim bladder, or other gas-filled structures, that re-radiate the energy to the otolith organs (Popper et al. 2014). Importantly, fish use hearing to establish an "auditory scene," and this may include cues to navigation, detection of predators, detection of prey, conspecific attraction, and sensing of environmental characteristics (Bregman 1994). Most fish can detect sounds ranging from a frequency of approximately 50 to 1,500 Hz (Popper and Hastings 2009a), though some species with specialized hearing can detect frequencies beyond this range. For context, continuous sounds emitted from ships or smaller vessels are generally < 1,000 Hz (Popper et al. 2014), which overlaps well with fish hearing capability. There is broad recognition that fish with swim bladders (or a similar gas-filled cavity) are more vulnerable to the effects of human-made sounds and are more likely to detect underwater sounds over a broad range of frequencies (Popper et al. 2014). In contrast, fish without gas-filled cavities such as flat fishes, skates, rays, gobies, sharks, and other deep-sea species are likely to be comparatively less vulnerable to human-made sounds (Popper et al. 2014).

The effects of underwater sounds on fish has become an increasing concern as fish are exposed to human-made sounds caused by shipping vessels, offshore wind farms, dredging operations, marine construction, and other human activities across all oceans (Hawkins et al. 2015; Popper and Hastings 2009a; Slabbekoorn et al. 2010). As outlined by Normandeau Associates (2012) and Popper and Hastings (2009a), the fundamental effects of man-made sound on fish include the following:

- Mortality or injury
- Tissue damage
- Temporary or permanent hearing loss
- Masking of sounds that fish react toward or depend on
- Behavioral changes such as avoidance

For fish, major concerns are that dredging sounds may block or delay anadromous fish migration, impair communication, or affect foraging (Reine et al. 2014b). The masking of communication among conspecifics could be detrimental to spawning and courtship of fish (Slabbekoorn et al. 2010). Much of the research conducted on human-made sounds have examined fish mortality rates and physiological changes, whereas the most likely effects are with fish behavior, including startle responses, temporary movements, avoidance, and changes to migratory behavior (Popper and Hastings 2009b). Furthermore, the direct effect of dredging sounds on fish have not been studied directly (McQueen et al. 2019); therefore, we review studies that have examined the effects of human-made sounds on fish and then provide context by comparing these effects to the summary of sounds emitted from hydraulic dredges. Because of the tremendous number of fish species (~32,000 species), an enormous anatomical diversity exists and each anatomy may respond differently to sounds that vary in frequency, duration, and intensity (Popper et al. 2014). Therefore, previous results should be taken in the context of generalizations with knowledge that each individual species will vary in their response to sounds.

Research regarding the effects of underwater sounds on fish has primarily been conducted on pile driving for marine construction, military sonar, and high intensity seismic surveys (Popper and Hastings 2009b; Reine et al. 2014a). Only impulsive sounds such as pile driving have been documented to result in fish mortality (≥ 193 dB re 1 µPa), although results are inconsistent across studies with similar sound emissions (Popper and Hastings 2009b). Popper and Hastings (2009a) showed temporary hearing loss and tissue damage were more common than mortality, and damage decreased as distance from the source increased. Temporary or permanent hearing loss varied by species, duration, and intensity of exposure; damage to fish tissues were commonly noted when sounds were > 180 dB re 1 µPa. Because study methods have relied on the caging of fish, the effects of sound on behavior remain unknown (Popper and Hastings 2009a). A few studies have documented a decline in catch rate, movement away from vessels, and movements to greater depths, including studies of the federally managed Atlantic herring and Atlantic cod (Popper and Hastings 2009a; Popper et al. 2014; Slabbekoorn et al. 2010). In an experimental study, settlement-stage coral reef fish larvae moved toward a reef sound playback, but the response was reduced, or avoidance behavior was observed, when boat sounds were added as a playback (Holles et al. 2013). Based on reviews and multiple studies, experts have detailed generated sound exposure guidelines (Popper et al. 2014).

To conclude, we have summarized the available evidence on the impact of underwater dredging sounds on fish. Assuming the maximum underwater sound directly at the source of dredging vessels of 183 dB re 1 µPa (range = 157.4–183 dB re 1 µPa), fish mortality due to sound is extremely unlikely even 1 m from the vessel itself (minimum sound level at which mortality has been observed is ≥ 193 dB re 1 µPa). Damage to tissues, temporary, or permanent hearing loss is possible in a localized area near the dredge vessel, as such damage has been observed with sounds of > 180 dB re 1 µPa. Given the attenuation of sound over distance (**Table 2-2** and **Table 2-3**, **Figure 2-3**), such damage would likely be limited to < 50 m from the vessel if fish do not avoid this particular area.

Behavioral responses and masking of natural sounds are expected because of dredging sounds, but there has been little research to confirm these expectations (Hawkins et al. 2015). Expected behavioral changes where sound is above ambient conditions may include avoidance, masking of conspecific communication, masking of predator or prey detection, or other behavioral changes (Hawkins et al. 2015; Slabbekoorn et al. 2010). Avoidance could have severe consequences if the particular area is critical for spawning, habitat is limited in the near vicinity, migratory corridors are blocked, or the area is important for other life history requirements. The distance for sounds associated with dredging to decrease to ambient levels ranges 400–2,700 m, although the maximum sound levels emitted by dredge activities are only present for approximately 0–300 m from the source of the vessel. Underwater sounds emitted from dredging operations could affect the behavior of fish at a considerable distance from the dredge operation. For example, Reine et al. (2014b) showed sounds near 130 dB re 1 µPa at approximately 1,200 m from a dredge operation (**Figure 2-3**) (Reine et al. 2014b). A 1,200 m radius around a dredge operation corresponds to a 452 ha area (1,117 acres) with a potential effect. But the range of response by fishes remains poorly understood for many species. In addition to these potential behavioral effects, chronic or cumulative impacts of non-lethal sound levels to fish are also poorly known.

2.3 Effects of Suspended Sediments on Fish and Their Habitats

2.3.1 Suspended Sediment Measures

Suspended sediment concentrations can be measured directly as total suspended solids (TSS) in units of mg L^{-1} or measured indirectly as nephelometric turbidity units (NTUs). NTUs measure scattered light from a water sample at a 90-degree angle from an incident light. Turbidity includes TSS but is also influenced by other factors such as plankton. We report studies that include TSS whenever possible. Ambient conditions of TSS in an offshore marine environment have been documented in the range of 5–10 mg L^{-1} (Duclos et al. 2013; Hitchcock and Bell 2004). Sediment plumes in marine environments can also be detected and tracked via satellite imagery, though the imagery can represent relatively low TSS (Fisher et al. 2015).

2.3.2 Extent, Duration, and Concentration of Suspended Sediments Related to Dredging

For cutterhead dredges, the rotation of the cutterhead itself produces substantial sediment resuspension in the lower part of the water column; plume concentrations at the water surface may be half of the concentration at the bottom (Havis 1988). In addition to increased TSS at the seafloor, TSHDs are often allowed to overflow until the slurry is of an appropriate density, and the overflow can be extremely turbid in close proximity to the dredge, as fine-grained TSS may reach > 750 mg L^{-1} (Havis 1988). Additionally, undesirable fine sediments may be discarded in the sorting and screening process (Michel et al. 2013; Sutton et al. 2009). For example, a recent study conducted near Biloxi, Mississippi, USA found the screening process at the borrow site removed 61% of the fine sediments before placement on the beach (Smith SJ et al. 2019). Havis (1988) compared TSHD and cutterhead dredges, and showed TSS concentrations were much greater for TSHD (with overspill allowed), particularly at greater depths.

Suspended sediments resulting from dredging can result in light reduction, suspended and resuspended sediments in the water column, and deposition of sediments on the seafloor. In this context, resuspension of sediments is defined as sediments that enter the water column after initial settlement on the seafloor. Michel et al. (2013) reviewed offshore sand dredging and noted that the extent and duration of suspended sediment plumes are highly variable based on site-specific criteria. A comprehensive list of relevant environmental and technical variables that impact TSS levels were identified by Anchor Environmental (2003), with the following most relevant factors for the marine offshore environment:

- Water depth
- Grain size
- Density and specific gravity of sediments
- Organic/detritus content
- Debris content
- Dredge type, size, and production rate
- Dredge methods (cut depth, swing of cutterhead, overspill of hopper, design of hopper dredge overflow)
- Currents
- Tides
- Waves
- Ambient salinity, temperature (thermoclines), and water chemistry

Within the last decade, a plethora of research has been published on the effects of dredging regarding the extent, duration, and concentration of suspended sediment plumes in marine environments. Characteristics of TSS may include a near-field settlement of more coarse sediments and a far-field settlement of finer-grained sediments (Van Lancker and Baeye 2015). To a lesser extent, sedimentation has also been documented. Below, we highlight several case studies:

- Spearman (2015) reviewed TSHD studies of aggregate extraction (sand and gravel) in United Kingdom waters. He found sediment plumes often travel < 500 m, but may reach up to 3 km. Sediment plume concentrations exceeded 70 mg L^{-1} within approximately 100 m of the dredge. However, projects were often monitored for only a few days, and the authors discarded two studies because they did not think the measurements were valid. In one of those studies, a near-bed extended down current 4.5 km from the dredge (Hitchcock and Bell 2004).

- Dredging offshore of Spain resulted in an increased TSS within 600 m of the dredging operation, and a second plume developed 3 hours post-dredging and moved 7.8 km (Van Lancker and Baeye 2015). The sediments included medium and coarse sands with < 1% silt-clay.

- Duclos et al. (2013) reported a subsurface plume of silty sands that declined dramatically within 10 minutes but remained at an elevated TSS concentration for two hours. During slack conditions, the plume expanded 600 m from the dredge; the range expanded up to 8.5 km (and a 100 m width) with tidal currents **(Figure 2-4** and **Figure 2-5).** Sediment deposition was estimated up to 800 m from the dredging. For this aggregate dredging, the overflow sediments were described as 55.3% sand, 30% silt, and 14.7% clay.

- Fisher et al. (2015) documented TSS effects within 3 km of dredging, but one unusual instance resulted in a plume moving ~20 km because of a local oceanographic feature and the resulting flow direction. TSS fluxes continued over 1½ years post-dredging, while no such turbidity fluxes were observed > 2 km from the dredge site. Dredge sites ranged from finer sediments (sand, silt, and clay at approximately 30% each) to a coarser, more offshore site (sand=70%, silt=10%, clay=10%). However, oceanographic factors and storms had a greater role in TSS movement compared to sediment grain sizes.

Overall, the pattern has emerged that extremely high TSS concentrations occur for a relatively short duration during and immediately following dredging. The area affected by high TSS, and potential sedimentation, is generally within 300–600 m of a dredging site, but moderate effects are expected to 3 km. Under strong currents, sediments plumes may extend up to 20 km from the dredge site. Although the vast majority of studies measure TSS for only hours to a few days post-dredging, resuspension of sediments has now been documented for up to 1½ years following dredging (compared to control sites). These resuspension events may span from concentrations similar to the dredge event to plumes with a very low TSS concentration. Given the limited number of studies that have monitored TSS beyond a few hours or days post-dredging, the frequency and mechanism of sediment resuspension events over a longer timeframe remains poorly understood and requires further study.

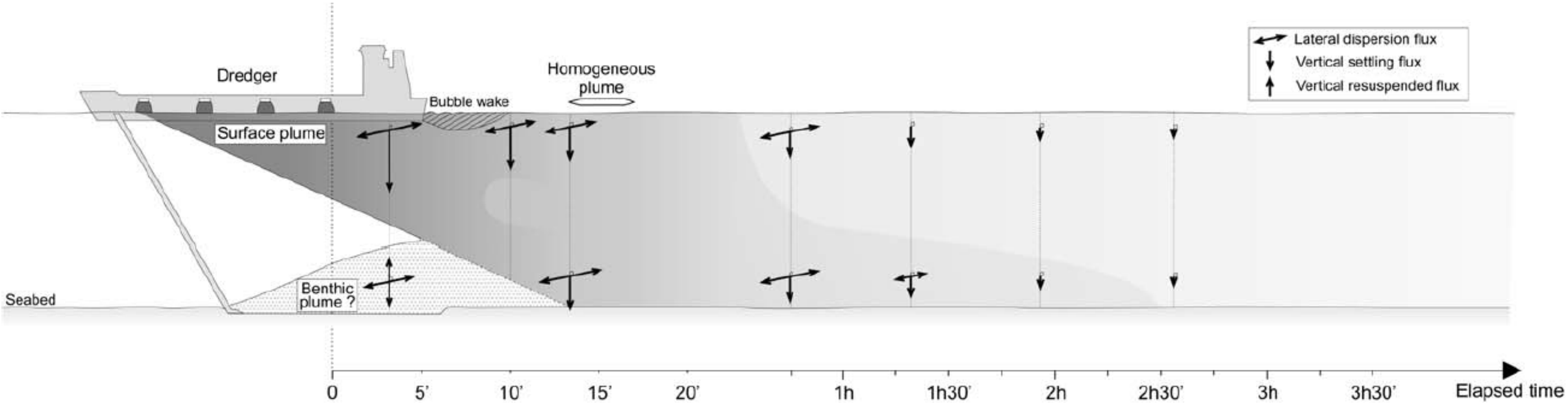


Figure 2-4. A conceptual model of suspended sediment movement from a dredging operation over time.
Source: Adapted from Duclos et al. (2013). Reproduced with permission from the Coastal Education and Research Foundation, Inc.

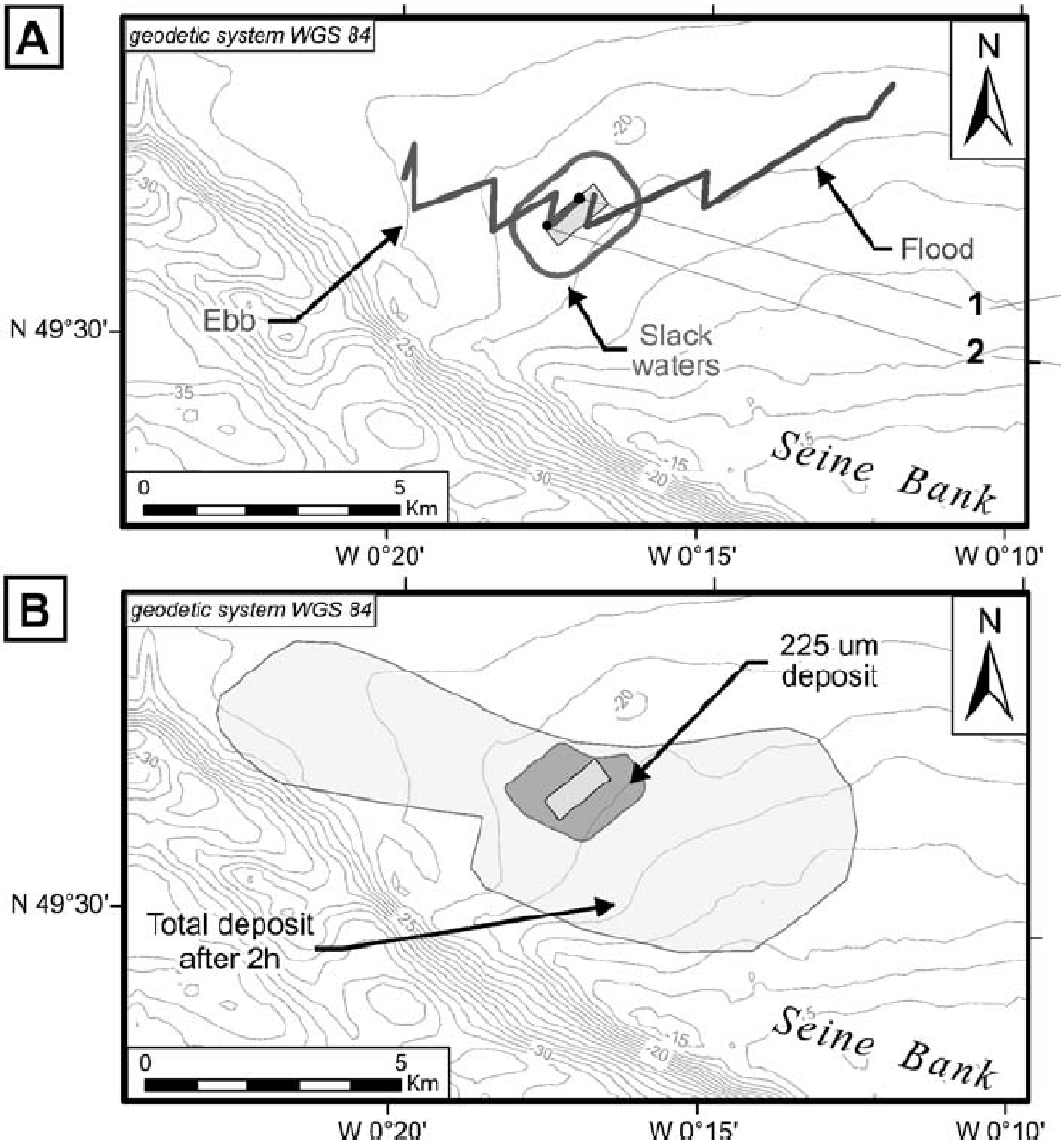


Figure 2-5. The extent of a subsurface sediment plume and sedimentation resulting from aggregate dredging in the English Channel, UK. A) sediment plume extent and direction during ebb, flood, and slack tidal phases of a mean tide; B) sediment deposition two hours after dredging.

Source: Adapted from Duclos et al. (2013). Reproduced with permission from the Coastal Education and Research Foundation, Inc.

2.3.3 Consequences of Suspended Sediments and Sedimentation on Fish

The vast majority of studies on the effects of suspended sediments and sedimentation on fish species have been conducted in freshwater and estuarine environments because of heightened TSS concentrations associated with erosion, flood pulses, runoff pollution, dredging, and aquaculture (Wilber and Clarke 2001). Relatively few studies have examined the effects of suspended sediments on marine fish species (Au et al. 2004; Partridge and Michael 2010). Generally, the combination of duration and concentration of TSS determine the consequences to fish. Environmental changes resulting from suspended sediments are 1) reduced light transmission, 2) reduced visibility, 3) decreased dissolved oxygen, 4) greater water temperature, and 5) potential release of contaminants (Kjelland et al. 2015). In offshore marine environments, contaminants are not as likely to be an important effect compared to rivers, estuaries, or harbors where pollution is commonly problematic (see Wenger et al. 2017 for a review). Potential responses of fish to suspended sediments are avoidance; changes in foraging and predation rates; physiological stress; reduced growth; physical damage; and mortality of adults, juveniles, larvae, or eggs (Kjelland et al. 2015; Wilber and Clarke 2001). Additionally, sedimentation may change the grain size of the substrate and bury invertebrates that are prey for fish. Here, we focus on marine species whenever possible, but salmon, trout, and other anadromous species have been most intensively studied and provide further context to assess the effects of suspended sediments and sedimentation.

2.3.3.1 Fish Eggs and Larvae

Fish eggs and larvae are particularly susceptible to sedimentation and suspended sediments because of their lack of mobility, relatively high oxygen demand, and anatomy (Appleby and Scarratt 1989; Wilber and Clarke 2001). Each species' natural history determines its exposure to suspended sediments. The timing and duration of the egg stage as well as its depth in the water column are particularly important. For example, herring eggs are most susceptible to mortality if exposed to ≥ 250 mg L^{-1} of TSS during the first 2 hours following egg release (Griffin et al. 2009). Eggs of bottom-spawning species and larvae near the substrate are particularly vulnerable to sedimentation (Wilber et al. 2005). In a review, Wilber and Clarke (2001) synthesized data on estuarine fish eggs and larvae. They found high variability among species in their tolerance to suspended sediments:

- Blueback herring and alewife eggs were not affected at 1,000 mg L^{-1}
- American shad larvae mortality increased at 100 mg L^{-1}
- Atlantic herring eggs were unaffected at 500 mg L^{-1}

Striped bass larvae showed decreased feeding at 200 mg L^{-1} and increased mortality at 485 mg L^{-1}. Partridge and Michael (2010) examined pink snapper larvae with a specific limestone-based suspended sediment. When larvae began the open-mouth stage of their development, 12 hours of exposure resulted in the first observable effects at 4–14 mg L^{-1}, and LC_{50} (lethal concentration when mortality $\geq 50\%$) was reported as 142–157 mg L^{-1}. By day 15, food intake of snapper larvae was reduced when TSS was as low as 15 mg L^{-1} (Partridge and Michael 2010). For a species that inhabit regularly turbid estuarine environments, Pacific herring larvae exposed to TSS of 200-400 mg L^{-1} for 16 hours did not differ from controls in regard to mortality, growth, heart rate, prey capture, or swimming velocity (Griffin et al. 2012). In contrast, coral reef fish have been found to have delayed larvae development with TSS of 15–45 mg L^{-1} (Wenger et al. 2014), and suspended sediments have been shown to interfere with larvae settlement into coral habitat at 45–180 mg L^{-1} (Wenger et al. 2011).

2.3.3.2 Juvenile and Adult Fish

For adult fish, the effects of TSS concentrations are thought to be primarily behavioral rather than physical because they are able to move away from such disturbances (Kjelland et al. 2015; Wilber et al. 2005). In support of this notion, examples of TSS lethal concentrations and durations of exposure determined in lab settings are extremely high (**Table 2-4**). Additionally, bottom-feeding fish may be more tolerant of such events compared to species such as marine pelagics (Humborstad et al. 2006). Yet, evidence supporting this idea is sparse. Overall, sublethal physical effects of suspended sediments to fish include clogging or coating of the gills, lesions, swelling, mucus and tissue production, less oxygen uptake, respiratory problems, and general changes to the structure of the gill (Wenger et al. 2017; Wilber and Clarke 2001). Although the effects are rarely studied outside of lab settings, potential consequences are increased energy expenditure, reduced foraging, reduced growth, and high susceptibility to predation (Wilber and Clarke 2001). Below, we highlight several lab studies on juvenile and adult fish exposed to suspended sediments.

- Adult Atlantic cod exposed to TSS of 550 mg L^{-1} of mud spanning 24 hours to 10 days had no mortality observed (Humborstad et al. 2006). However, after 24 hours, all cod were observed to have acute lesions to gill tissues at low to moderate TSS concentrations. Ten days following exposure, cod produced 1–2 layers of epithelial cells on their gill tissues, which may act as a protective measure.
- Juvenile orange-spotted grouper exposed to TSS of 128 mg L^{-1} for 10 days resulted in clogging of gills, increased mucous cells, and other signs of physiological stress (Wong et al. 2013). Exposure of the species to 6 weeks of TSS of 50–200 mg L^{-1} had mortality of rates 20–30%.
- For three coral reef fish species, TSS of ≥ 45 mg L^{-1} affected gill structure, reduced oxygen diffusion distances, and increased energy expenditures (Hess et al. 2017). They also showed oxygen uptake decreased in one species after exposure.

Table 2-4. The effect of TSS concentrations on select marine and estuarine fish species and northern quahog.

Species	TSS (mg L^{-1})	Duration of exposure (days)	Percent mortality
Bluefish (juvenile)	800	1 day	100%
Atlantic menhaden (juvenile)	800	1 day	100%
Bay anchovy	2,310	1 day	10%
Striped bass	1,500	14 days	0%; Only physiological stress
Atlantic silverside	580	1 day	10%
Spot	13,090	1 day	10%
Northern quahog	1,000	10 days	10%

Source: Data summarized from Kjelland et al. (2015) and Ray et al. (2005).

Fish use both visual and chemical cues to detect prey, and turbidity may affect fish behavior, foraging success, and conversely, the survivorship of prey species. Collin and Hart (2015) and Utne-Palm (2002) provide reviews of how fish vision and turbidity interact in the context of foraging. The vast majority of studies reviewed were on freshwater species (trout, bluegill, walleye, largemouth bass) or salmon. Only a few estuarine fish species have been studied in this context. Nonetheless, some useful conclusions can be drawn. The reaction distance of adult fish in response to planktonic prey are directly and negatively related to turbidity (Utne-Palm 2002; Wilber and Clarke 2001). Furthermore, increased turbidity is expected to decrease the ability of visual specialists to obtain food (Collin and Hart 2015). In this respect, turbidity is most likely to interfere with foraging of large piscivores that detect prey from a distance compared to other species such as planktivores (Utne-Palm 2002). Common prey species may also benefit from less predation. For example, shrimp are less predated in turbid environments (Macia et al. 2003). The turbidity relationship with fish foraging and predation may even explain differences in fish community composition, including the evolutionary benefit for juvenile fish life stages to use more turbid estuaries compared to clear marine waters used by adults (Utne-Palm 2002).

The effects of suspended sediments on the distribution, foraging, avoidance, and other behaviors of fish are often difficult to discern. For salmon, pulses of suspended sediments cause alarm reactions, disruption of schooling behavior, cessation of fish feeding, increased swimming, and relocation to undisturbed areas (Wilber and Clarke 2001). Such examples are rare, or nonexistent, for estuarine or marine fish species. A summary of relevant studies are provided below:

- Lowe et al. (2015) investigated juvenile pink snapper in New Zealand and showed estuaries with relatively high TSS (range 4–37 mg L^{-1}) had fewer fish captures and more gill structural changes. Snapper in low TSS environments fed primarily on pelagic prey, whereas snappers in estuaries with higher TSS fed on large benthic invertebrates (Lowe et al. 2015). A complementary lab study showed suspended sediments reduced foraging success, and physiological stress indicators were present after 1 month of TSS of 20–160 mg L^{-1} (Lowe et al. 2015).
- Leahy (2011) showed a coral reef fish foraged 40% less in turbid environments, and they had the strongest antipredator responses at the highest TSS concentration of 41 mg L^{-1}.
- A coral reef planktivore exposed to a TSS concentration of 45 mg L^{-1} had a slower reaction to food availability. Fish in this treatment consumed all of their food in only 52% of tests compared to 87% of control tests (although not statistically significant) (Wenger et al. 2012).
- For a coral reef fish, Wenger et al. (2012) found a TSS of 90 mg L^{-1} effected reaction time and food consumption; fish growth over the first two weeks of the experiment was less in the TSS treatments of 90 and 180 mg L^{-1} compared to the control (Wenger et al. 2012). Correspondingly, the fish showed less movement and settled in fewer live coral habitats when TSS was at 30 mg L^{-1}, presumably because of the disruption of visual cues (Wenger and McCormick 2013).
- Given the paucity of data on the behavior of marine fish species exposed to relatively high TSS concentrations, further investigations are warranted for species that are likely to be affected by dredging.

2.3.4 Consequences of Suspended Sediments and Sedimentation on Nearby Corals, Coral Reefs, and Hard Bottom Habitats

Although sand dredging occurs in soft bottom substrates, impacts to nearby corals, coral reefs, and hard bottom habitats have occurred. Corals and coral reef habitats determine the distribution of a diverse array of fish species. The impacts of dredging on corals via increased TSS and sedimentation has a history of concern that has led to consistent research and reviews of such research (Dodge and Vaisnys 1977; Rogers 1990; Jones et al. 2016). In a comprehensive review of the impacts of dredging on corals, and coral reefs, Erftemeijer et al. (2012) documented 35 cases where dredging effects were studied. These studies ranged worldwide and included seven cases documented in Florida, USA. The dredging effects included no impacts, complete physical removal, burial, and tissue damage, as well as lethal and sublethal stress caused by relatively high TSS and/or sedimentation. Stress can result in reduced growth, reduced calcification rates, and bleaching (Erftemeijer et al. 2012; PIANC 2010). In some cases, high percentages of coral reefs were immediately destroyed, whereas other studies have shown that corals continue to die over the course of a few months as the cumulative stress added to the mortality rate (Erftemeijer et al. 2012). Examples of measured effects include no detectable impacts 200 m from a dredging operation (Doorn-Groen and Foster 2007) to a loss of 80% of corals at a distance of 1 km from dredging activity (Stoddart and Stoddart 2005). Most recently, dredging at the Port of Miami, Florida, USA navigation channel resulted in a heightened rate of coral mortality up to 700 m from the dredged channel despite environmental monitoring initially being limited to within 50 m of the channel (Miller et al. 2016b). In this case, disease and complete colony loss was over twice as common for colonies near the dredging site compared to reference colonies; mortality was caused by burial of corals and stress (Miller et al. 2016b). Fisher et al. (2018) quantified sediment deposition, turbidity, and benthic light reduction at 26 sites ranging up to 33 km from a dredge operation, including control sites where coral mortality was monitored before, during, and after dredging. Water quality effects were detected as far as 19.6 km in the direction of a strong current and 2.1 km in the opposite direction (Fisher et al. 2015). In regard to corals, mortality was substantial and was disproportionately more common within 2 km of the dredging site compared to farther away sites (Fisher et al. 2018). Recommendations for best practices for dredging near corals and coral reefs are further provided by PIANC (2010).

Several mechanisms are responsible for stress and mortality of corals because of suspended sediments and sedimentation. Reef-building corals depend on photosynthetic activity of symbiotic, unicellular algae called zooxanthellae. For these corals, a heightened level of TSS causes stress or mortality as ambient light reaching zooxanthellae are reduced; non-photosynthetic corals may be vulnerable to clogging or smothering from sediments deposited on the coral's surface (Erftemeijer et al. 2012). Both fine and coarse grain sediments can cause stress to corals. Fine sediments strongly reduce light levels, whereas coarse particles cause scouring and abrasion of coral tissues (PIANC 2010). Sedimentation on the surface of coral tissues may reduce feeding rates, reduce larvae survival and settlement, and increase energy expenditures to expel sediments (Erftemeijer et al. 2012). Corals can expel sediments through mucus secretion or new tissue production; however, the redirection of energy can lead to stress, suppressed growth and reproduction, and/or mortality (Erftemeijer et al. 2012). Morphology of corals generally corresponds with sensitivity of the species to sedimentation; branching corals tend to be more vulnerable to heightened TSS and less sensitive to sedimentation, and plate corals are less sensitive to suspended sediments, but are vulnerable to sedimentation (Erftemeijer et al. 2012; PIANC 2010).

The tolerance of coral species to suspended sediments and sedimentation varies widely by species and are dependent upon the duration and intensity (i.e., TSS) of the event, as well as the natural conditions in which the coral grows (e.g., inshore vs. offshore). Thus, a single threshold where TSS concentrations will cause mortality is difficult to predict (Erftemeijer et al. 2012). Stress effects have been commonly observed at concentrations spanning 50–200 mg L^{-1}, but some species do not show negative effects even at concentrations as high as 1,000 mg L^{-1} (Erftemeijer et al. 2012). Such differing responses of corals to TSS concentrations may lead to changes in coral reef community composition and a reduction of coral biodiversity, as only particularly tolerant species may remain. Fisher et al. (2018) studied a range of variables and found coral mortality was best predicted by mean daily sediment deposition over 60 days, a 14-day running mean of turbidity, and a 14-day running mean of light reduction. An accumulation of multiple stressors may also interact to affect corals. For example, an experimental study showed bleached corals (generally associated with warming temperatures) could not clear sediments from their tissues at the same rate as unstressed corals (Bessell-Browne et al. 2017).

For a hard bottom ecosystem in the South Atlantic, Lindeman and Snyder (1999) conducted pre- and post-dredging fish surveys 0.8 km offshore of Florida, USA; the dredging buried 4.9–5.7 hectares (ha) of hard bottom with sediments. Both numbers of fish species (54 pre-dredging vs. 8 post-dredging) and their abundances declined for at least 15 months following dredging. For this study, the mechanism of fish loss could be either suspended sediment effects on fish behavior/physiology or the degradation of benthic habitats (Hess et al. 2017).

2.4 Effects of Physical Removal of Sediments with Implications to Fish

2.4.1 Oceanographic and Physical Changes Associated with Shoals and Dredging

Sand dredging may affect both the physical substrate and oceanographic conditions. Nairn et al. (2004) categorizes the physical changes resulting from marine sand dredging as follows:

- Morphodynamics (e.g., topography, sediment transport and mobility, elevation/depth change)
- Oceanographic conditions (e.g., dissolved oxygen, water temperature, waves, currents)
- Seabed composition (e.g., sediment grain size, stratigraphy, compaction, mineralogy, dissolved oxygen, organic content)

We do not address mineralogy or stratigraphy in this review because they are not likely related to fish distribution. We also do not explicitly review dredging impacts on compaction and dissolved oxygen of sediments. These properties are likely associated with invertebrates, which are much better studied. Changes in sediment grain sizes are closely related to benthic invertebrates, therefore, we focus on these topics together. The magnitude of the effects of physical removal will depend on the volume of sediment removed, surface area dredged, location of dredging in relation to shoals, accretion rates, the proportion of the shoal area that is dredged, and the availability of similar habitats nearby. Mitigation measures will also determine the effect of removal, though mitigation has likely been included in the reported studies.

Morphodynamic changes in elevation/depth and topography are straightforward (**Figure 2-6**). Shallow areas may be partially or completely removed, and the terrain may be flattened by sand dredging. Therefore, fish species that prefer shallow waters, heterogeneity in water depths, or structural complexity (i.e., slope, hills, and valleys) may lose suitable habitat until the substrate is able to recover. Additionally, dredging may create depressions, furrows, or pits. As a result, changes in wave energy may affect sediment transport processes; particularly of concern is sediment transport to shorelines because of its impact on beach accretion or erosion (Nairn et al. 2004).

The removal of shoals has the potential to affect dissolved oxygen, water temperature, currents, and wave energy. In turn, these conditions may affect fish distribution. Dubois et al. (2009) presents evidence that Ship Shoal in the GoM may be a hypoxia refuge (i.e., when dissolved oxygen (DO) is > 2.0 mg L^{-1}) for benthic invertebrates where individuals may temporarily disperse to, and then later recolonize the surrounding waters when DO returns to higher levels. Likewise, Craig et al. (2012) found a high abundance of demersal species (Atlantic bumper, Atlantic croaker, brown shrimp, spot, and sand seatrout) near large shoals as well as the edges of hypoxic zones of the GoM. Few species were abundant in low DO waters. They also found strong gradients of DO over 1 to 5 km distances, particularly over “shallow, wind-swept shoals” that were surrounded by low DO (Craig 2012).

Reeves et al. (2017a) also documented higher DO levels during summer on an artificial reef on Ship Shoal compared to a seaward artificial reef. Their study showed that hypoxia events resulted in reef fish avoiding waters > 12 m in depth because of low DO at the bottom.

Explicit documentation of shoals, or shoal removal, associated with water temperature changes are sparse. At Cape Canaveral, shoals were observed to be associated with a change in ocean currents and may result in water temperatures 2–3°C higher on the landward side of shoals (Reyier et al. 2014). In their study, Reyier et al. (2014) suggested this was enough to make Cape Canaveral a warm water refuge for sharks. To our knowledge, no simulations have been conducted to determine the threshold at which shoal dredging might disrupt oceanographic processes to the point where water temperature is changed.

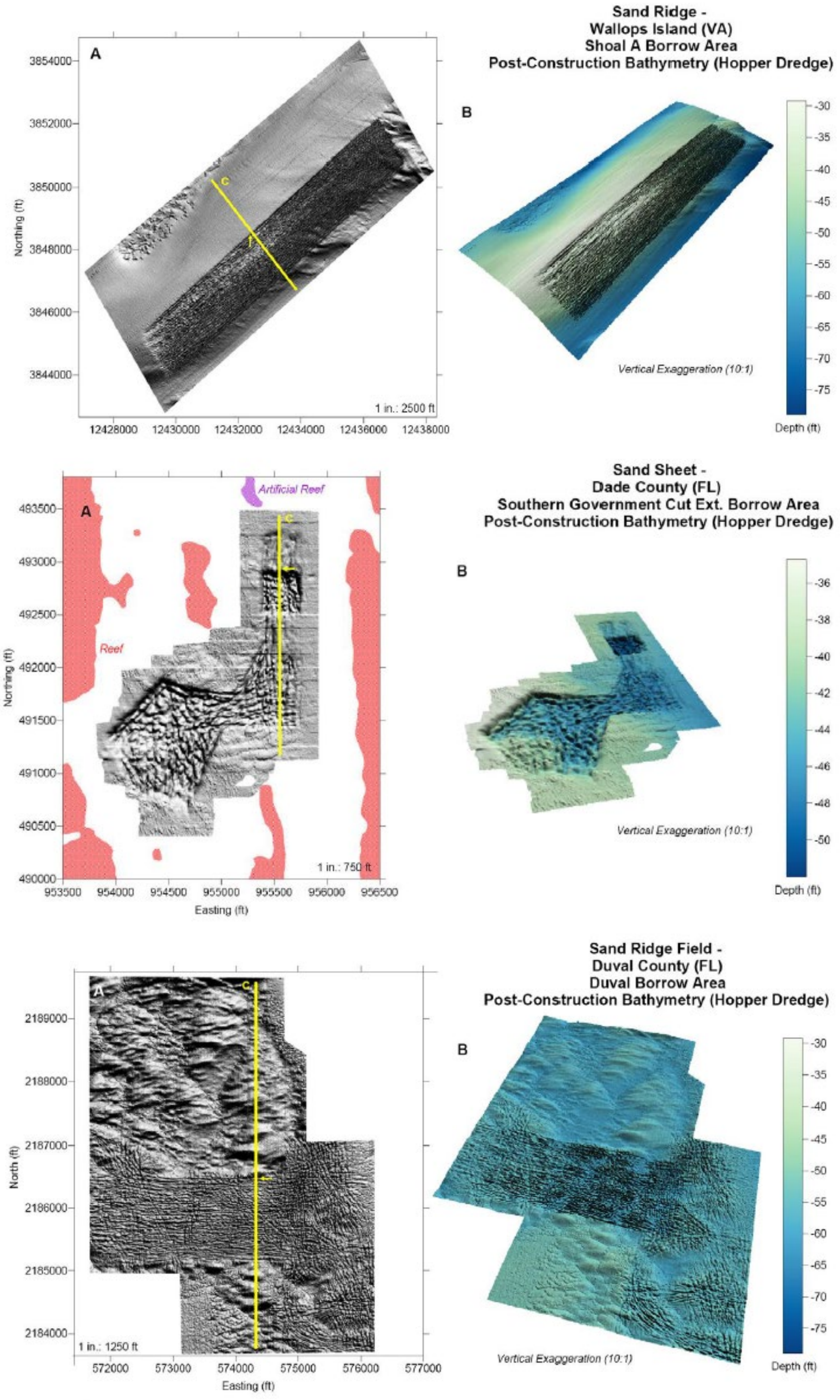


Figure 2-6. Digital elevation models of post-dredging bathymetry for representative OCS borrow areas.

Source: Figure modified from Michel et al. (2013). The pink color represents hard bottom habitat for Dade County, Florida, USA.

Numerous studies have simulated or modeled how hydrodynamics and wave energy are expected to change as a result of dredging (Byrnes et al. 2004b; Kelley et al. 2001; Maa et al. 2004; Stone et al. 2009), although wave energy is not a common variable used to model the distribution of fish species. The emphasis on wave energy is because of the potential for the reduction of shoals to influence sediment transport to the shoreline and subsequent changes in erosion or accretion rates (Hayes and Nairn 2004; Maa et al. 2004). Currents are more often associated with fish, but modeling the effects of dredging on currents is rare.

Offshore of Alabama, USA Byrnes et al. (2004b) used wave transformation modeling to examine scenarios of sand extraction. Simulations of dredging showed increased wave heights of 0.20–0.50 m in the lee of the shoal and decreased wave heights (maximum -0.4 m) adjacent to the shoal. Changes became minimal when waves approached the shoreline, although it is noted that simulating the impact of storms is more difficult than projecting normal wave conditions (Byrnes et al. 2004b). Importantly, Byrnes et al. (2004b) also suggest that borrow site size, orientation, and proximity to shoreline played a role in the variability. Similarly, offshore of New Jersey, USA Byrnes et al. (2004a) showed that shoals affected the divergence and convergence of waves; sand extraction scenarios showed maximum wave height changes spanned 0.06–0.6 m with a variable amount of wave height dissipation when approaching the shoreline. Offshore of Delaware and Maryland, USA Maa et al. (2004) investigated the response of waves and bottom currents after the simulated removal of 24 million m^3 (31.4 million yd^3) of sand shoals, as estimated for an accumulated impact over 10–20 years. In this study, modeled wave height increased by a factor of two after simulated removal, and these changes have the potential to cause increased shoreline erosion (Maa et al. 2004). Based on small- and large-scale sand removal simulations, Stone et al. (2009) found that neither simulation would change abrupt changes in current direction. However, large-scale sand removal can change the velocity of currents and may affect sediment transport (Stone et al. 2009). A remaining concern with sand dredging is whether there is a threshold beyond which sand removal from a shoal might result in deflation or the ultimate disappearance of the feature because the wave pattern is reduced in magnitude or if depth is increased to a point where sand deposition is minimal (Hayes and Nairn 2004).

2.4.2 The Effect of Dredging on Benthic Invertebrates, Sediment Grain Size, and Potential Food Web Changes

The effect of substrate removal as the result of dredging has the potential to disrupt food webs and negatively affect fish species that feed on benthic invertebrates (Kim et al. 2008; Nairn et al. 2004; Newell et al. 1998). The link between demersal fish and benthic invertebrates is important for fisheries (Brooks et al. 2006; Newell et al. 1998; Smith et al. 2013); however, it has proven difficult to study. Pinnegear et al. (2000) reviewed documented marine trophic cascades, defined as the linkage of three or more trophic levels. They concluded that trophic cascades likely exist in soft-substrate ecosystems, but research simply has not been conducted in these less conspicuous ecosystems where polychaetes and crustaceans dominate (as opposed to the well-studied sea urchin populations or intertidal communities). A caging experiment of soft bottom substrates in the Chesapeake Bay has shown that the exclusion of blue crab and spot led to large increases in benthic infauna density and diversity (Virnstein 1977), which suggests a strong link exists between benthic infauna and these common prey species.

Generally, benthic invertebrates are classified as epifauna or infauna. Epifauna live on the surface of the substrate, whereas infauna live within unconsolidated sediments. Both types of species are vulnerable to removal from dredging. Benthic invertebrates are related to sediment grain size, topography, and oceanographic variables (Rutecki et al. 2014). For a further review on this topic, Rutecki et al. (2014) details factors that affect the distribution of benthic invertebrates and documents common species found on soft bottom substrates of the US Atlantic and GoM.

Marine benthic communities, their successional states, and the effects of dredging are reviewed by Newell et al. (1998) (**Figure 2-7**). From a summary of worldwide studies, they showed that dredging consistently results in reduced benthic invertebrate species richness, abundance, and biomass as well as a change in species composition. Each of these changes have variable recovery times. In addition, measures of species evenness may increase because of the increased abundance of a few pioneer species (Crowe et al. 2016). In their review, Newell et al. (1998) state that benthic recovery to a pre-dredging state may come in 2–3 years except for complex biological associations that depend on relatively slow growing fauna; in these cases, 5–10 years may be needed to fully recover. Since the 1998 review, further studies have primarily supported these conclusions (Crowe et al. 2016; Newell et al. 2004; Simonini et al. 2007; Waye-Barker et al. 2015).

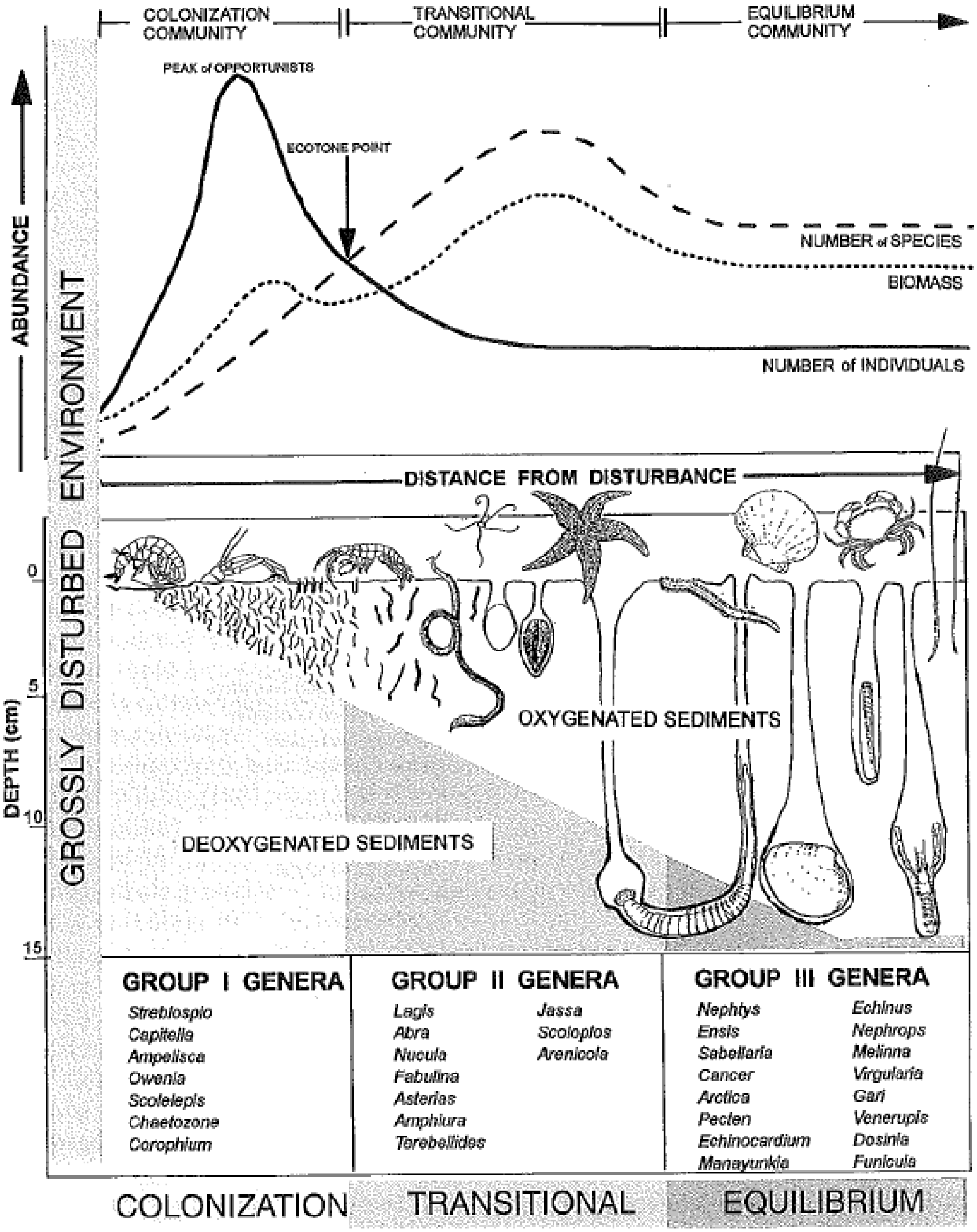


Figure 2-7. Conceptual diagram of ecological succession of benthic communities.
Source: Adapted from Newell et al. (1998) and based on Pearson & Rosenberg (1978) and Rhoads et al. (1978). Reproduced with permission from the Taylor and Francis.

We have provided further details on the effect of dredging on benthic invertebrates. Below, we highlight major conclusions reached from international studies post-1998. We do note that recovery rates may differ based on project-specific attributes such as sediment grain size, depth of dredging, proximity to sediment sources and the sizes of those sediments, and mitigation measures that may assist with recovery. Nonetheless, these conclusions are robust in that they represent similar findings from multiple studies.

- **Conclusion #1: Recolonization of a few early successional benthic invertebrates occurs quickly, while full recovery of the benthic community may take from 2.5 to > 7 years to recover. Specific evidence includes the following:**
    - Species composition shifts quickly (i.e., < 100 days) toward early successional species, sometimes with only a few species that are short-lived and colonize rapidly (Boyd and Rees 2003; Newell et al. 2004).
    - Crowe et al. (2016) found more abundant amphipods and polychaetes following dredging, but fewer mollusks and other species. They found density, species richness, and species composition still differed with reference sites 6–8 years post-dredging.
    - In the GoM, a dredge pit was dominated by one species 3 years post-dredging, leading to lower benthic community metrics compared to reference sites (Palmer et al. 2008). In this study, three species were found inside the pit and 9 to 27 species were found at reference sites.
    - One year after dredging, density of individuals were similar to reference sites because of a single pioneer species (Cooper et al. 2007). In their study, Cooper et al. (2007) found full recovery of benthos occurred after 7 years post-dredging.
    - Advanced stages of recovery may be observed as soon as 12 months following extraction, and the original community can be restored in 2.5 years (Simonini et al. 2007; Simonini et al. 2005).

- **Conclusion #2: For aggregate dredging, more frequently, or intensively, dredged substrates take longer for benthic communities to recover compared to less frequently dredged substrates. We note that such studies are lacking where sediments are primarily sand. Specific evidence includes the following:**
    - At a site with high frequency dredging, recovery of the benthic community was similar to reference sites after 15 years, nearly double the time found for nearby low frequency dredging sites (Waye-Barker et al. 2015). The reference sites in this study had approximately 50% gravel, 30% coarse to fine sands, and 20% silt/clay.
    - Compared to a site with a low frequency of dredging, a high frequency dredging site (deeper pits, anchor dredging technique) showed a greater post-dredging reduction in benthic species richness, density, and biomass up to ~1.5 years post-dredging; one pioneer species was particularly dominant at this site (Newell et al. 2004). Sediments in this study primarily included silty sands, silty gravel, and sandy gravel.
    - Community composition of benthos at low and high frequency dredging sites differ and differences persisted for ≥ 6 years (Boyd et al. 2005; Boyd and Rees 2003). In these two studies, sediments at reference sites ranged from 41–66% gravel and 24–40% sand.
    - The recolonization process for areas that sustained multiple dredging events may create new habitat for pioneer species not observed at lower frequency dredging sites (La Porta et al. 2009).
    - A low frequency dredge site had benthic fauna recovery 7 years post-dredging, but a high frequency site (i.e., with 25 years of dredging) had not recovered 11 years after dredging ceased (Hussin et al. 2012). The reference sites in their study were approximately 45% gravel, 35% coarse to fine sands, and 20% silty/clay.

- **Conclusion #3: Change in sediment grain size corresponds to changes in benthic communities. Specific evidence includes the following:**
    - Simonini et al. (2007) found little change in sand sediment grain size post-dredging and found recovery of microbenthic communities to be complete by 2.5 years.
    - Offshore of South Carolina, Crowe et al. (2016) showed surficial sand sediment became more fine-grained after dredging and infauna species were still different than reference substrates 8 years later.
    - Sediments inside a dredge pit offshore of Louisiana had more silt and clay compared to reference substrates and recovery had not occurred after 38 months (Palmer et al. 2008).
    - Desprez (2000) showed benthic communities changed corresponding to the sediments, which changed from coarse to fine sands. Species richness was restored by 16 months, but density and biomass were still reduced by 40% and 25%, respectively, 28 months post-dredging.
    - Recovery of benthic fauna of a high impact dredging site occurred after 15 years, concurrent with the recovery of pre-dredging sediment grain sizes (Waye-Barker et al. 2015).

2.4.3 Drawing Analogous Benthic Impacts from Bottom Trawl Fisheries

Because of the parallels of marine dredging and the disturbance of substrate from bottom trawling conducted by commercial fisheries, we provide a brief overview of bottom trawling impacts to benthic communities and the potential for cumulative impacts. Bottom trawling is similar to dredging in that substrate is disturbed, but trawling does not result in removal of substrate like dredging does. Hiddink et al. (2017) conducted a meta-analysis and found that the depletion of fauna was highly correlated with the trawl type and its penetration into the seabed. For example, otter trawls with a penetration depth of 2.4 cm removed an average of 6% of organisms, while hydraulic dredges used for fishing had a penetration depth of 16.1 cm and removed 41% of organisms per pass (Hiddink et al. 2017). They estimated median recovery times to be 1.9–6.4 years. In a meta-analysis of bottom trawling studies, Hiddink et al. (2018) showed that the shortest-lived organisms (< 1-year lifespan) increased in abundance immediately after trawling and were similar to reference conditions. In their study, organisms with a > 1 year lifespan decreased and were less abundant than reference locations. The abundance of organisms with lifespans of > 10 years declined the most (-37%) (Hiddink et al. 2018). Additionally, sessile (e.g., bivalves) and low mobility fauna are expected to take the longest to recover (Sciberras et al. 2018). Therefore, substrates with a relatively high proportion of long-lived organisms are expected to be the most sensitive to bottom trawling disturbances (Hiddink et al. 2018).

Despite the link of benthic invertebrates to fish, few studies have examined the effect of substrate disturbance on fish. At Georges Bank offshore of New England, Smith et al. (2013) found total abundance and biomass of benthic epifauna was greater at sites undisturbed by fisheries bottom trawling, although results were inconsistent by site. Smith et al. (2013) showed that several fish species had a greater length at undisturbed sites, and these species included haddock, longhorn sculpin, Atlantic cod, and winter flounder. In contrast, little skate and longhorn sculpin were found to have greater lengths at a disturbed site; notably, the difference was marginal and only occurred in one year (Smith et al. 2013). Additionally, fish diet dissimilarities among sites corresponded to epibenthic fauna that were sensitive to bottom disturbance (Smith et al. 2013).The authors further acknowledge that the fish sampled were not restricted to undisturbed waters and less mobile species might show greater differences than mobile species.

2.4.4 Factors Promoting Physical Shoal Recovery

The extent, duration, and severity of the impact of substrate removal via dredging will depend on the recovery time needed to restore the substrate and its fauna. Rutecki et al. (2014) reviews the geological underpinnings of sand shoal evolution and growth, including the role of storms, cold fronts, waves, and other hydrodynamics. These fundamental dynamics ultimately influence sediment transport, and thus, the potential for shoal growth and recovery following dredging operations.

Borrow design, size, location on a shoal, depth of excavation, water depth, and orientation of the excavation site will all influence recovery rates (Xu et al. 2014). For example, Xu et al. (2014) reported that a borrow site designed as a wide pit with shallow sloping walls did not accumulate mud sediments in the borrow area, presumably because water regularly flushed out these fine sediments before they could settle. Dibajnia and Nairn (2011) proposed using a relative shoal height ratio (shoal relief: depth of the base) to determine potential for shoal growth and concluded that shoals in water depths of > 30 m would not grow or recover following dredging because of the importance of wave-induced currents (Dibajnia and Nairn 2011). Although recommendations are difficult to generalize across regions, Rutecki et al. (2014) summarized recommendations from CSA International Inc. et al. (2010) and Dibajnia and Nairn (2011) as follows.

Promoting physical recovery of sand shoals may occur by the following:

- Extracting sand from depositional areas, the leading edge, or downdrift margin of a shoal
- Avoiding dredging in upstream erosional areas that feed the depositional areas
- Shallow dredging spread out over a larger area rather than deep dredging in a smaller area
- Alternating dredged versus undredged areas down the longitudinal axis of the shoal crest
- Excavation in the higher portions of the shoal that are exposed to wave-generated turbulence

Further recommendations summarized by Rutecki et al. (2014) include the following:

- Shoals with a base depth deeper than 30 m of water should not be dredged because deeper shoals have limited potential to grow after dredging.
- Shoals with a relative shoal height (defined as height/base depth, or height divided by base depth) of less than 0.5 should not be dredged because shoals with a smaller H/BD ratio are not likely to recover after dredging.
- If shoal recovery to its pre-dredge height is desired, then only shoals that have reached their maximum relative shoal height, where $(\text{height/base depth})_{\max} = (\text{base depth}-5)/\text{base depth}$, are recommended for dredging.
- For shoals with a base depth of 21 m (as determined from the modeling of Isle of Wight Shoal), dredging from the shoal crest is not recommended. When dredging from the top of the shoal, relative shoal height should not be reduced to less than 0.65 (i.e., removal of more than 1.3 m) after dredging, or the shoal will not re-grow to the same pre-dredge height.
- Sand should not be removed from the entire length of the shoal, i.e., dredging along the axis of the shoal, because it affects wave-focusing processes and the shoal does not recover to the same pre-dredge height.
- For the Mid-Atlantic, it is recommended that sand be dredged from the SW side of the shoal, because a) wave-focusing is concentrated on the NE side of the shoal and b) overall shoal migration is toward the southwest.

3 Review of Fish Habitat Associations in Relation to Geomorphology, Oceanographic Conditions, and Other Factors

In this section, we review the literature on habitat associations of federally managed fish. The review is organized following US Fishery Management Council regions of the GoM, South Atlantic, Greater Atlantic (collectively Mid-Atlantic and New England), and then highly migratory species are addressed together for all regions. Because of the large number of federally managed species encompassing EFH designations of "reef fish," sharks, and other species groups, we limited the scope of our review in several ways.

First, our literature review is restricted to research within our three study areas: Greater Atlantic, South Atlantic, and GoM. Although some species have a worldwide distribution (e.g., shark species), habitat associations have the most relevance when applied to the area where observations have been made. For example, water temperatures or depths used by blacktip shark are likely to differ between North America and Australia.

Second, we limited our review to species that have designated EFH that commonly overlap with sand shoals. To provide this context, we provide tables that quantify how much of the EFH of each species, or species group, overlaps with the three study areas (Federal waters, ≤ 50 m depth). Specific methods included using the ArcMap (ESRI, Redlands, CA) clip tool to reduce EFH polygons to the respective study area. Then, this remaining EFH area (in km^2) was divided by the total study area. We also focused on species documented to use sand shoals, particularly those listed by Rutecki et al. (2014).

Thirdly, we do not attempt to characterize the natural history of species unless there is direct relevance to the fish's habitat use or seasonality. Such descriptions are deliberately kept brief; EFH documents and other sources should be consulted for such information. Likewise, we do not determine what constitutes a particular life stage (e.g., juvenile vs. adult) because measures may differ by study. Each study is reported upon as it is presented by the author. The focus here is on habitat associations. A summary of basic habitat associations (sand affinity, temperature, depth, salinity) is provided in *Volume 4: ShoalMATE (Shoal Map Assessment Tool for EFH).* We report the units given by each study, with the exception of salinity, which had a simple conversion from parts per thousand (ppt) to practical salinity units (psu).

Fourth, the literature review relies primarily on peer-reviewed science. In this manner, we are limiting speculation or data that has not been thoroughly evaluated. Overall, the following sections provide a foundation of understanding specific habitat associations of species. Although we attempted to focus on fish-geomorphology relationships, such relationships are often poorly studied or results are nonexistent for species. Therefore, there is no section on geomorphology relationships. On the occasions when fish-geomorphology relationships were observed, we do clearly identify such associations with the word "geomorphology" or "substrate" to allow for searches of such information throughout these sections. The habitat use of each species, or species group, is organized by life stage in the marine environment in the order of juvenile (if relevant), adult, spawning adult, larvae, eggs, and then any related spatial mapping studies. We focus on life stages within the marine environment and provide only brief overviews of other life stages (e.g., juveniles that inhabit estuaries).

To better document the knowledge gap of marine fish and relationships with geomorphology, we conducted an international literature review in later sections. The focus of this search was on spatially explicit habitat associations, as spatial mapping is an important aspect of defining EFH. Here, our goal was to quantify how species, categorized into functional groups (e.g., sharks, demersal fish, hard bottom fish, medium pelagics), relate to geomorphology, physical oceanography, fish-based oceanography, geographic, and biological predictors. The goal of this international review was to highlight our state-of-knowledge and quantify knowledge gaps.

Overall, the habitat associations addressed within this literature synthesis have the goal of describing how each species is distributed and what particular habitats are important or are not important. The reader may use this knowledge combined with knowledge of the effects of dredging in previous sections of this report to assess environmental impacts. Such an assessment is beyond the scope of our review here and may be highly site-specific.

3.1 GoM Fish Habitat Associations and Seasonality

Key Points and Knowledge Gaps *(gaps are in italics)*

- In shallow waters of the Gulf of Mexico, federally managed species both of direct socio-economic importance and likely to be common on sand shoals include red drum; red snapper; Spanish mackerel; king mackerel; several shark species; and brown, pink, and white shrimp.
- Seasonal hypoxia, defined as waters with ≤ 2 mg L^{-1} of dissolved oxygen, may cause fish and shrimp to aggregate at the edge of hypoxic waters or move vertically in the water column. Windswept sand shoals are suspected to act as temporary refugia for fish and invertebrates where oxygen is available, even when surrounding waters are hypoxic.
- *Red drum are poorly studied in ocean habitats.* Spawning aggregations are known to occur near inlets and estuaries where larvae disperse to juvenile habitats.
- Juvenile reef-associated fish use sand shoal habitats; red snapper and lane snapper have been the two most common species observed. Adult reef-associated fish that use artificial structures may forage in surrounding soft substrates, but the use of defined shoals remains uncertain.
- *The shallow water habitats of coastal migratory pelagic species are unstudied.* Of relevance to shoals, Spanish and king mackerel are piscivores whereas cobia primarily feed on demersal crustaceans and fish.
- Brown shrimp are positively related to mud substrate, whereas pink shrimp are positively associated with sand and rock. Other habitat associations of these species include depth, salinity, water temperature, proximity to shoreline, and dissolved oxygen. Detailed white shrimp habitat relationships remain largely undocumented.

3.1.1 Introduction to the GoM Physical Setting and Fish

The GoM ecosystem includes 1.6 million km^2 of coastal and marine waters and is considered the ninth largest body of water in the world (Karnauskas et al. 2013). In the northern Gulf of Mexico (nGoM), shallow waters are strongly influenced by riverine inputs and deeper waters are strongly influenced by the Loop Current. The Loop Current enters the GoM through the Yucatan Channel, goes northward in the basin, and then exits via the Straits of Florida. Importantly, the Loop Current produces spin-off eddies with typical diameters of 300–400 km and with a current velocity of 1–2 m s^{-1}; these eddies usually take a westward path originating generally near the Mississippi Delta (Johnson et al. 2017) (**Figure 3-1**). To a lesser extent, wind stress curl, or the influence of wind on water vertical structure, also contributes to the vorticity of eddies (Ohlmann et al. 2001). The end result is an exchange of shelf waters and deeper waters (Johnson et al. 2017; Ohlmann et al. 2001). The interaction of eddies with the shelf can produce upwelling (in cold-core eddies) and downwelling (in ware-core eddies), which are important for biological productivity (Spies et al. 2016).

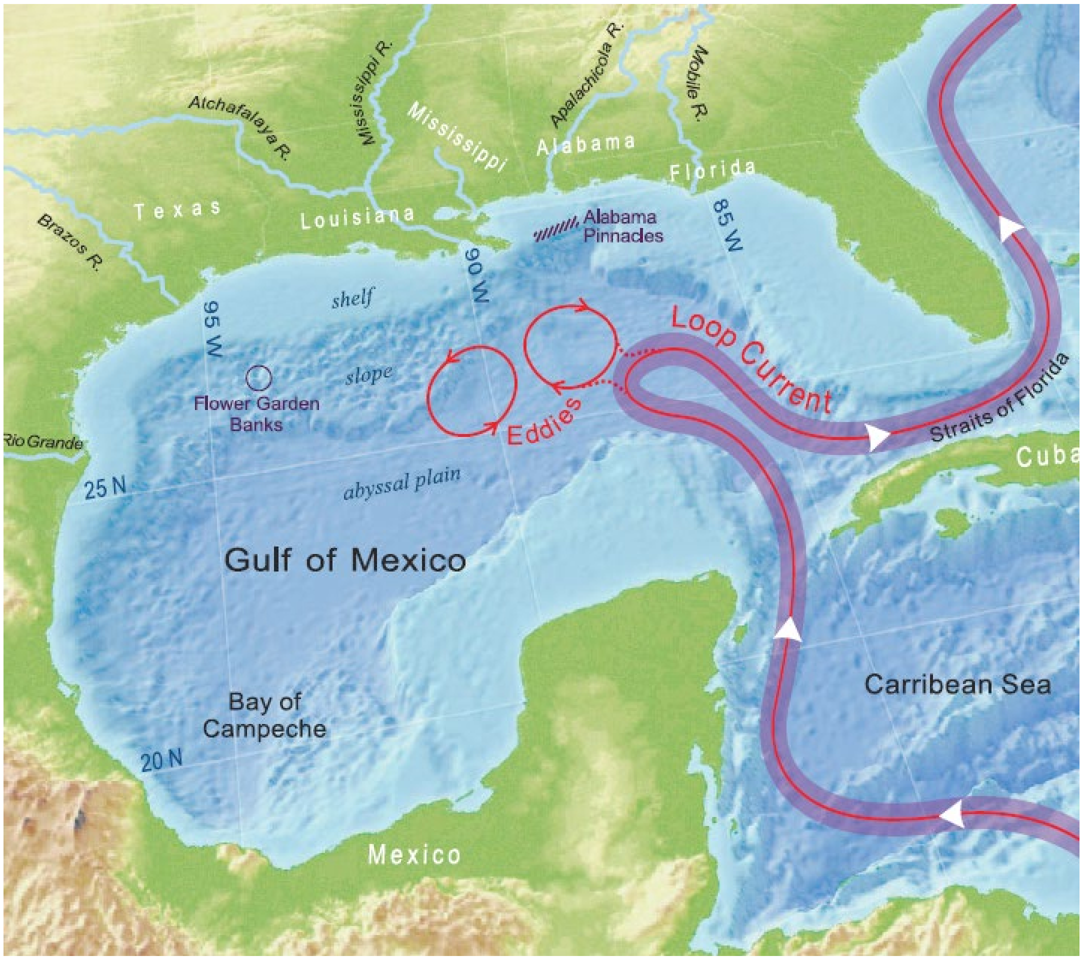


Figure 3-1. GoM's Loop Current, spin-off eddies, major rivers, and select geological features.
Source: Adapted from Spies et al. (2016). Reproduced with permission from the University of Southern Mississippi, Gulf and Caribbean Research.

The majority of sediments on the continental shelf west of the W 88º longitude (near Mobile Bay, Alabama) are mud with some sand interspersed; east of this longitude are predominately sands. Offshore of the Florida peninsula, designated EFH for corals depict substrate with an interspersion of sand and corals. Ship Shoal, Sabine Shoal, St. Bernard Shoal, and Heald Bank Shoal are examples of isolated shelf shoals, and bedform shoals are common on the west Florida shelf (Rutecki et al. 2014). Estuaries, subaquatic vegetation (SAV), oyster reefs, and coastal wetlands in the nGoM also contribute toward offshore marine productivity by providing habitat for estuarine-dependent life stages and producing common prey species of the marine environment (e.g., menhaden, crabs) (Spies et al. 2016). GoM reef fish habitats have changed dramatically with the introduction of > 3,900 oil platforms and > 20,000 artificial reefs established since the 1940s (Shipp and Bortone 2009).

Over 150 rivers contribute freshwater to the nGoM with the Mississippi River contributing over half of the annual total volume (Spies et al. 2016). The westward direction of Mississippi River plume lowers salinity in the nearshore of central and western Louisiana. Of particular importance, the nGoM hypoxic zone is a prominent feature in the summer season, and it generally ranges from 15,100–18,000 km^3 (Obenour et al. 2013) (**Figure 3-2**). Hypoxia is generally defined as waters with a DO < 2 mg L^{-1}. This "dead zone" is caused by excessive nutrients and the subsequent proliferation of phytoplankton near the mouth of the Mississippi River. DiMarco et al. (2010) found the distribution of hypoxic waters were associated with winds, freshwater flow, bathymetry, and they suggest the area between shoals undergoes frequent hypoxic events because of the change in water flow due to topography. Shoals themselves contribute to mixing and tend to have fewer hypoxic events (DiMarco et al. 2010). Additionally, studies have examined the effects of the hypoxia on shrimp (Craig et al. 2005), shrimp trawlers (Purcell et al. 2017), reef fish (Reeves et al. 2017a), and forage fish (Craig and Crowder 2005). Together, these studies have found species tend to either aggregate at the edges of hypoxic zones or move their depth in the water column to avoid hypoxic waters.

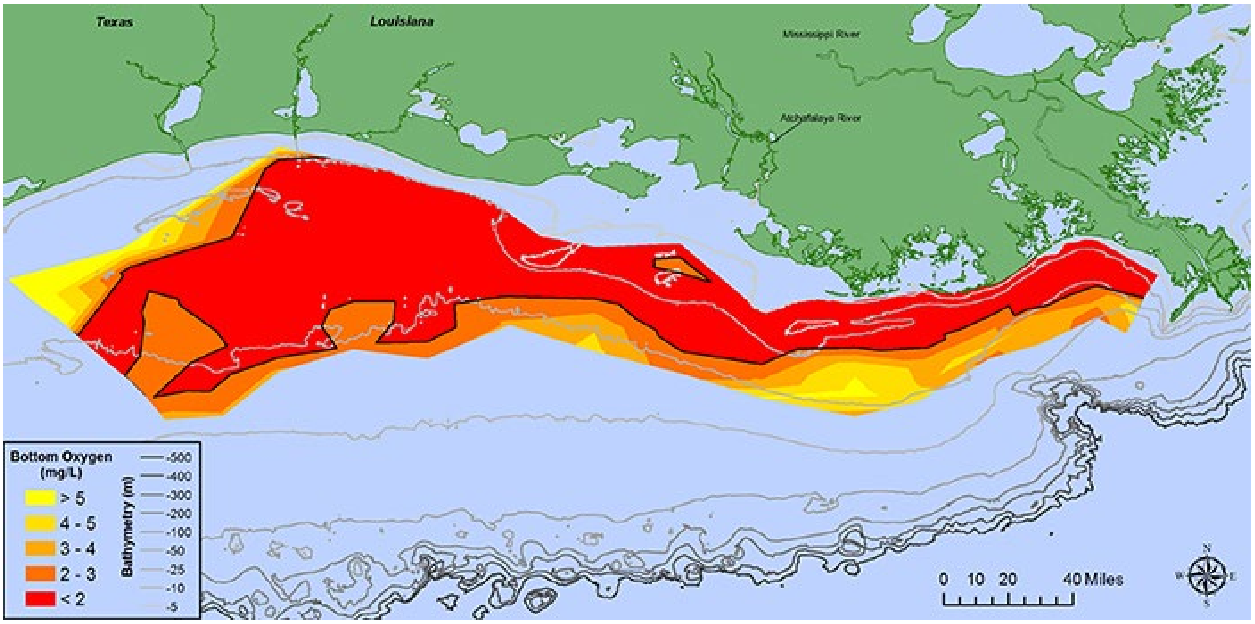


Figure 3-2. Map of GoM hypoxic zone in July 2017. The black line shows the area where DO is < 2 mg L-1, which is considered the threshold for hypoxia.
Source: NOAA, https://www.noaa.gov/media-release/gulf-of-mexico-dead-zone-is-largest-ever-measured (accessed 2019 Aug 8) and is courtesy of N. Rabalais, LSU/LUMCON.

Tarnecki et al. (2016) described the current state-of-knowledge of food webs in the GoM. They showed species, or species groups, that are animal prey for many other functional groups include bivalves, brown shrimp, pink shrimp, other shrimp, carnivorous macrobenthos, crabs and lobsters, infaunal mesobenthos, zooplankton, small demersal fish, other demersal fish, Sciaenidae, seatrout, sessile filter feeders, sheepshead, small pelagic fish, small reef fish, and squid. In particular, Gulf menhaden has been a focus of food web studies (Geers et al. 2016; Robinson et al. 2015), and the Gulf menhaden fishery is one of the largest by volume in US with a mean of 447,000 metric tons harvested annually from 2005–2009 (Karnauskas et al. 2013).

For commercial fisheries of the nGoM, key federally managed species include red snapper, grouper species, shrimp, spiny lobster, and tuna (National Marine Fisheries Service 2017). Other key species listed include blue crab, menhaden, and mullets. For recreational fisheries, federally managed species of economic importance include red drum, porgies (particularly sheepshead), red snapper, and Spanish mackerel (National Marine Fisheries Service 2017). In individual states, federally managed species of economic importance include sharks, vermilion snapper, gag, red grouper, gray snapper, and king mackerel (National Marine Fisheries Service 2017). As of 2018, overfished stocks in the nGoM include greater amberjack, gray snapper, and lane snapper (NOAA Fisheries 2019). Spawning seasons for GoM fish span much of the year, but are particularly concentrated June 1 through mid-October (Ainsworth et al. 2015) (**Figure 3-3**).

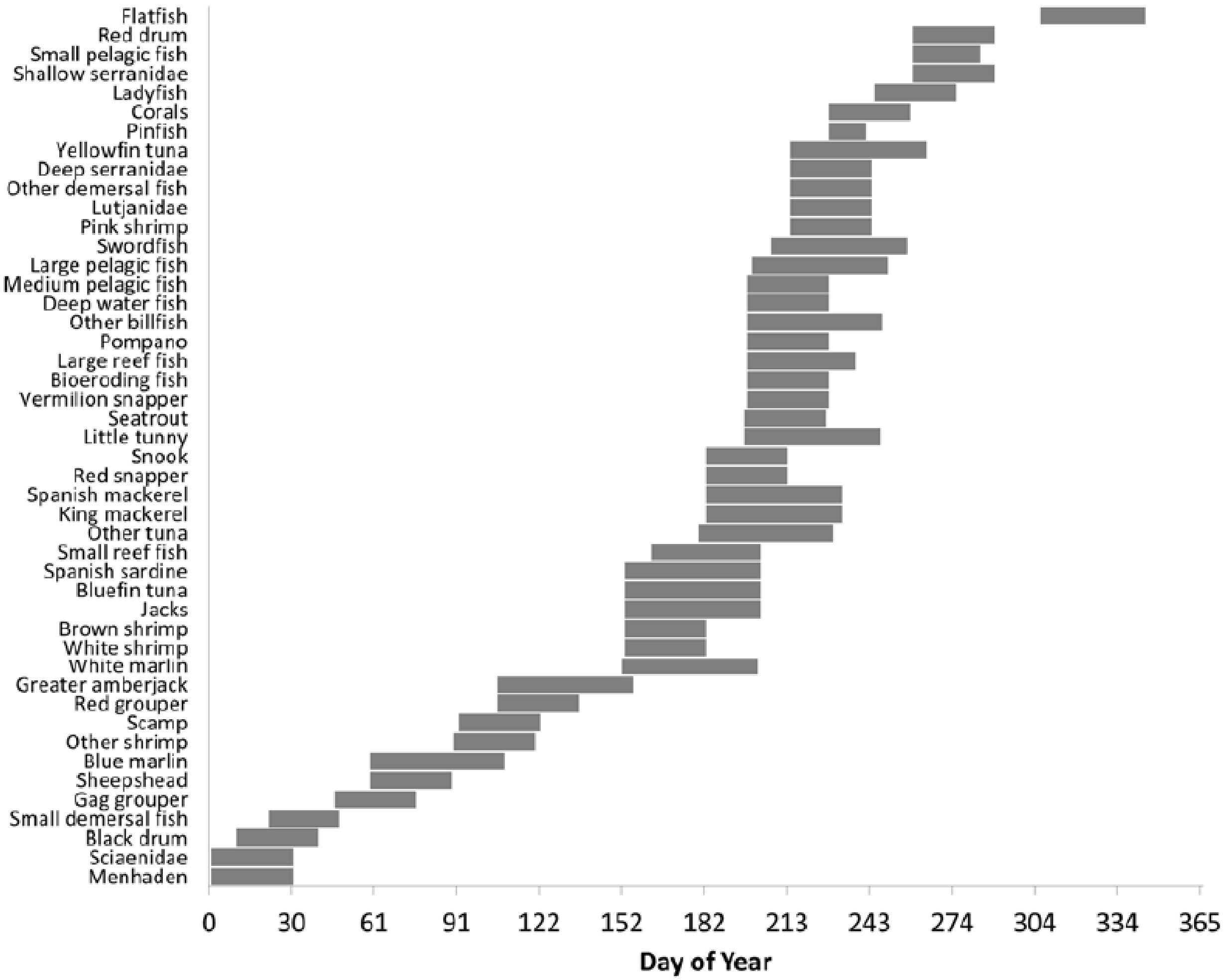


Figure 3-3. Seasonality of GoM spawning seasons.
Source: Adapted from Ainsworth et al. (2015). Reproduced with permission from the Creative Commons Attribution License.

The EFH designations for GoM federally managed fish show a strong overlap with Federal waters that are ≤ 50 m in depth (**Table 3-1**). Although red drum EFH appears to have a relatively low overlap with these waters, designated EFH overlaps with shoals commonly used for sand dredging offshore of Louisiana (e.g., Ship Shoal, St. Bernard Shoals), Mississippi, Alabama, and the peninsula of Florida.

Table 3-1. Species, or species groups, designated with EFH in the GoM.

Species or EFH species group	Proportion of GoM study area designated as EFH	Fishbase description of habitat
Red drum	0.22	Demersal ≥ 10 m
Reef fish (49 species)	0.98	Reef-associated
Coastal migratory pelagics - Spanish mackerel - King mackerel - Cobia	0.98	Pelagic
Shrimp - Brown shrimp - White shrimp - Pink shrimp - Royal red - Rock shrimp - Seabob shrimp	0.80	NA
Corals	0.18	NA
Spiny lobster	0.06	NA

Notes: Those overlapping with > 50% of the study area are bolded. Study area is defined in **Figure 1-2.** All life stages were included.

3.1.2 Red Drum in the GoM

Red drum are a demersal fish that use estuarine waters as juveniles, and then leave those areas at maturity (~3.5 years old) for offshore marine waters (Matlock 1987). Post-settlement and juvenile red drum in the nGoM use a variety of estuarine habitats, including seagrass, bare substrate, oyster reef, rivers, habitat edges, and, in particular, waters in close proximity to salt marsh (*Spartina alterniflora*) (Dance and Rooker 2016; Matlock 1987; Moulton et al. 2017; Williams et al. 2016). In contrast to the somewhat well-studied estuarine habitats of juvenile red drum, adult red drum habitat use of marine environments is mostly unknown.

Adult red drum were present, but rarely caught in a study of Sabine and Trinity Shoals offshore of Texas (Brooks et al. 2005). Powers et al. (2012) analyzed adult red drum surveys in state and Federal waters east of Louisiana and south of the Alabama-Florida border. They reported red drum are year-round residents of the sampling area (water temperatures ranging 12.3–31.7°C), but catch per unit effort (CPUE) peaked in March, April, and November. The majority of red drum collected by longline were in waters < 20 m in depth, and the maximum depth was 63 m (Powers et al. 2012). We note that sampling was concentrated in inshore waters with more shallow depths compared to offshore environments. Matlock (1987) speculates that most red drum are within 16 km of Texas shoreline, but also describes red drum catches near oil platforms in Louisiana and reefs in Texas as far as 113 km offshore.

Spawning has long been speculated to be in nearshore waters and in close proximity to inlets or passes (Matlock 1987), and recent evidence has confirmed this notion. Powers et al. (2012) reported aerial surveys that found red drum spawning aggregations near inlets from Ship Island (Mississippi) to Dauphin Island (Alabama) and surrounding Louisiana's Chandeleur Islands. Lowerre-Barbieri et al. (2016) quantified spawning aggregations in the nearshore marine environment just offshore of Tampa Bay, Florida, and southward to Charlotte Harbor, Florida. Passive acoustic monitoring provided evidence that most adult red drum moved elsewhere before and after the spawning season (Lowerre-Barbieri et al.

2016). The protection of spawning habitats are key to the current management strategy of red drum (Powers et al. 2012); therefore, these habitats should be further examined. Across the nGoM, red drum physiology indicates a spawning season ranging from mid-August to early October (Wilson and Nieland 1994); this was further evidenced by acoustic arrays in nearshore waters of Florida with first and last detection of adults being August 26 and October 15, respectively (Lowerre-Barbieri et al. 2016).

3.1.3 Reef Fish in the GoM

Numerous reef fish species use artificial reefs, including oil platforms, which are sometimes located on shoals in the nGoM. However, reefs and hard bottom are particularly vulnerable to dredge impacts, and if these habitats are present, a range of buffer distances are implemented to designate where dredging cannot occur. To address general reef fish habitat associations on shoals, we first address species that are found within or in close proximity to artificial reefs on shoals. Secondly, we provide a case study of fish associated with artificial reefs on and near Ship Shoal. Third, we address reef fish that use unconsolidated substrates as habitat in their juvenile life stage. In particular, we use red snapper as a case study to represent reef fish that have a juvenile life stage over soft sediment substrates. Red snapper adults also use soft sediment substrates for foraging as they venture away from artificial reefs and oil platforms. These associations are also reviewed.

Murawski et al. (2018) assessed bottom longline surveys of adult reef fish in the nGoM and reported the median, interquartile range, 5th and 95th percentiles of depth and temperature for common fish species (**Figure 3-4**). The following results were observed:

- Red grouper, sharpnose shark, red snapper, and scalloped hammerhead were primarily in waters with relatively higher temperatures and shallower depths. Each of these species had their median depth intersect with waters ≤ 50 m in depth and temperatures ≥ 19°C.
- Golden tilefish and yellowedge grouper were primarily found at greater depths and cooler temperatures (≥ 100 m depth and < 19°C).
- Scamp, greater amberjack, and red porgy were found at intermediate depths and temperatures.

In order of the most commonly sampled species via fishery-independent longline, the snapper/grouper complex included red snapper, red grouper, yellowedge grouper, wenchmen, red hind, scamp, snowy grouper, silk snapper, gag, speckled hind, vermilion snapper, blackfin snapper, coney, yellowtail snapper, warsaw grouper, and yellowmouth grouper. All other snapper/grouper complex species had a sample size < 10 individuals (Murawski et al. 2018). For reef fish, hypoxia may cause fish to avoid bottoms and vertical changes in depth of fish may be observed (Reeves et al. 2017a). Switzer et al. (2015) noted that little is known about fine-scale dynamics of hypoxia and refugia may exist where reef fish congregate.

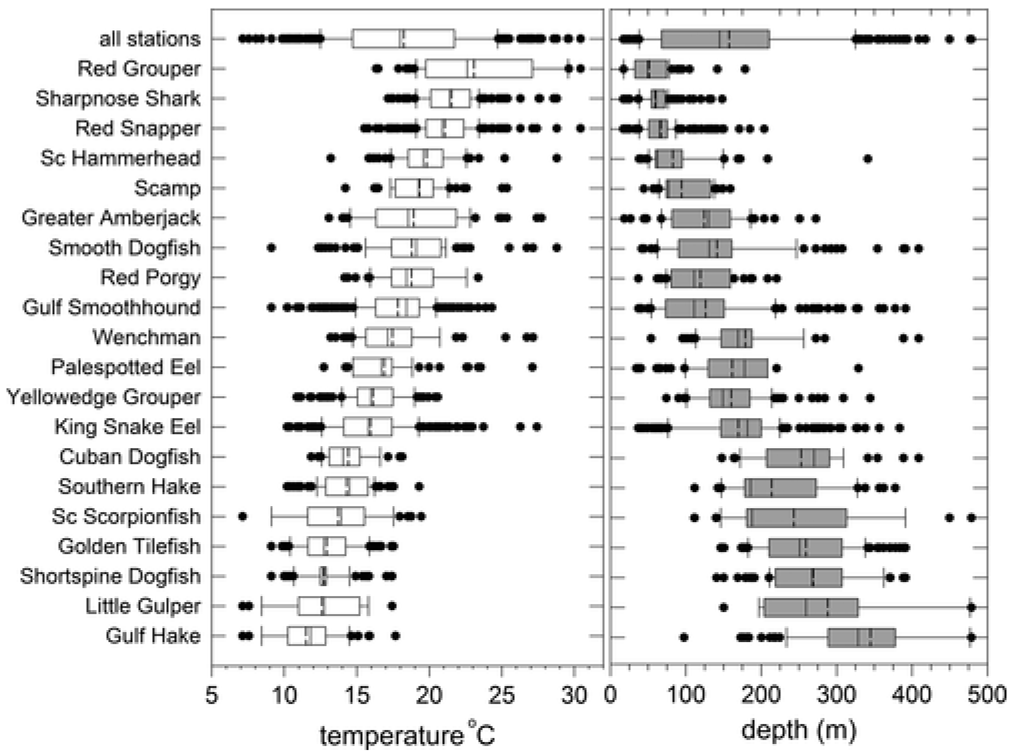


Figure 3-4. GoM fish habitat associations with water temperature and depth.
Source: Adapted from Murawski et al. (2018) with open access permissions.

Gruss et al. (2018; 2017) used a coarse interpolation modeling method to estimate the spatial distribution of red grouper, gag, juvenile red grouper, juvenile gag, and juvenile red snapper in the nGoM. They based their model on locations of known samples and assumed the grid cells closer together are more similar than those cells farther apart. We caution that this type of modeling omits the effects of fine-scale features, like geomorphology, or substantial changes in environmental conditions. The distribution of several other species and functional groups of interest were estimated for the peninsula of Florida, including amberjack, reef carnivores, and reef omnivores (Gruss et al. 2018).

In regard to fish shifts in habitat use, Gruss et al. (2017) showed gag and red grouper males, nonspawning females, spawning females, and juveniles had differing spatial distributions. Adults were generally more widespread, while juveniles were found in more focused areas. Habitat associations were not quantified in their mapping, but geostatistical modeling of fish sampling locations showed a trend of red grouper and gag being commonly found at depths < 60 m (Gruss et al. 2017). Red grouper and gag were most common in the northeast section of the nGoM, with a relatively high probability of encounter offshore of west Florida shelf waters (Gruss et al. 2017). Juvenile red snapper were most abundant west of Tallahassee, Florida (Gruss et al. 2018); a finer scale of distribution was not discernible.

3.1.4 A Case Study of Adult Reef Fish on Artificial Reefs On and Adjacent to Ship Shoal, Offshore of Louisiana

Reeves et al. (2017a) studied reef fish inhabiting oil platforms at Ship Shoal (mean = 8 m depth) and an oil platform 7.5–15 km seaward of the shoal (mean = 15 m depth). The most common species documented include Atlantic spadefish (100% of surveys), sheepshead (93%), gray snapper (90%), blue runner (85%), Atlantic bumper (73%), red snapper (65%), Bermuda chub (63%), and horse-eye jack (43%) (Reeves et al. 2017a). Ship Shoal only had one hypoxic (DO of < 2 mg L^{-1}) event over a 2-week timeframe in the summer of 2016, while seaward platforms were consistently hypoxic from June 27 to

August 22 (Reeves et al. 2017a). They found a total of 29 fish species, and fish were generally not observed > 12 m in depth when hypoxia was present at seaward platforms. Prior to hypoxic conditions, shoal platforms had more red drum, black drum, and yellow jack; during and after hypoxic conditions more gray triggerfish were observed at Ship Shoal. Some fish, such as Atlantic bumper, were more abundant in seaward waters, presumably because of the deeper depths. Importantly, juvenile gray snapper abundance increased by a factor of 10 on Ship Shoal as the season progressed, while the seaward area remained at a stable level. Aggregation of gray snapper may occur because they are benthic invertebrate feeders and their prey may be more abundant on the shoals compared to the surrounding hypoxic waters (Gelpi et al. 2009; Reeves et al. 2017b). Overall, shoal and non-shoal oil platforms differed in their fish species assemblages, but hypoxia did not appear to affect the abundance of most species.

3.1.5 Shoals as Habitat for Juvenile Reef Fish in the GoM

Juvenile reef fish have been documented in surveys specifically focused on sand shoals. In trawl surveys conducted at, or near, a sand borrow site offshore of Florida, Zarillo et al. (2008) commonly found juvenile grunt species and sand seabass. Other reef-associated species found included lane snapper, white grunt, red grouper, dwarf sand perch, tomtate, and gray snapper (Zarillo et al. 2008).

Brooks et al. (2005) examined fish species at Heald Bank, Sabine Bank, Trinity Shoal, and Tiger Shoal offshore of Louisiana. Reef-associated fish species observed included lane snapper, rock seabass, dwarf sand perch, sand perch, and red snapper. Offshore of Texas and Galveston Bay, Heald Bank, Sabine Bank, and Freeport Banks have been documented as important nursery habitat for lane snapper (Mikulas and Rooker 2008). Peak densities of juvenile lane snapper occurred August 4 to September 1. In regard to geomorphology, a higher abundance of year-0 lane snapper occurred at offshore mud and shell ridges compared to inshore mud sediments (Mikulas and Rooker 2008). In an investigation of shoals offshore of Texas, Wells et al. (2009) found juvenile red snapper and lane snapper occurred in > 50% of trawl surveys; dwarf sand perch were also common. As evidenced by the temporal variability of lane snapper, Wells et al. (2009) speculates that shoals in their study area may provide an important link between estuarine and offshore habitats.

3.1.6 Red Snapper in the GoM

Red snapper in the nGoM are relatively well studied and provide a case study for both juvenile reef fish that utilize unconsolidated substrates and for adult reef fish that use these substrates for foraging near reefs. Gallaway et al. (2009) summarizes the life history of red snapper in the nGoM, and we provide a synopsis here **(Figure 3-5)**. The red snapper is a demersal species that matures by two years of age. Major life stages include the following:

- Pre-recruit (< 50 mm total length [TL]) stages of egg, larvae, and post-settlement juveniles. At 50 mm, they become susceptible to the shrimp fishery gear as bycatch.
- Post-recruit (> 50 mm TL) stages are juvenile (0 and 1 year), young adults (2–7 years), and mature adults ≥ 8 years).

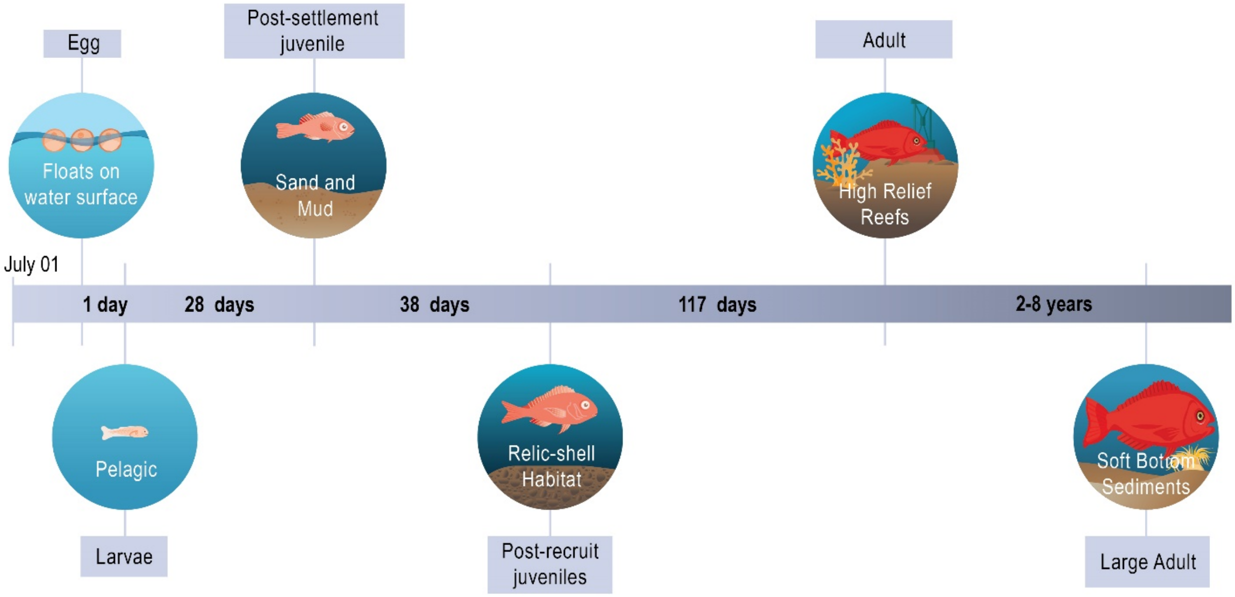


Figure 3-5. Summary of red snapper life stages and broad description of habitats used. Dates for larvae and juvenile stages depict an example with an egg hatching July 1st.

Newly settled red snapper move to habitats with some structure, and juvenile red snappers have been documented to have relationships with geomorphology and selection for shell, low-relief shell, and substrate complexity:

- An experimental study of age-0 red snapper showed a preference for shell substrate compared to sand (Szedlmayer and Howe 1997).
- Rooker et al. (2004) found abundance to be greater at low-relief, relic-shell habitats compared to mud bottoms.
- An experimental study spanning August–November showed that 1-m^2 artificial reefs attracted more juvenile red snapper, and a greater diversity of juvenile reef fish, when the substrate was more complex (Lingo and Szedlmayer 2006).
- Szedlmayer et al. (2014) found a greater abundance of juvenile red snapper at a site with a silt substrate compared to sand and coarse sand at other sites.
- Juvenile red snapper have been documented using muddy substrates offshore of Texas (Rooker et al. 2004) and Alabama (Szedlmayer and Lee 2004).
- Powers et al. (2018) suggested that juvenile abundance was greater in a region with more muddy sediments compared to waters with sandy sediments. However, they did not formally examine relationships with sediment grain size.

Gallaway et al. (1999) developed a habitat suitability index for age-0 and age-1 red snapper based on trawl surveys. They found that red snapper selected for depths of 28–37 m (range: 18–64 m), bottom temperatures of 22–29°C (range: 20–29°C), salinities of 34–35% (range: 30–37%), dissolved oxygen of $\geq$ 5 mg L^{-1}; and that the number of oil platforms were negatively related to both age classes of red snapper. The relationship with higher platform density may be because of predators at these sites or because these fish are already recruiting to platforms where trawls cannot logistically survey (Gallaway et al. 1999). In a study of the artificial reef zone offshore of Alabama, Powers et al. (2018) found juvenile red snapper had the highest abundance in the 20–40 m depth class, compared to deeper waters. Juvenile red snapper are also affected by hypoxia. Relative abundance of age-0 and age-1 red snapper offshore of Louisiana were reduced during years with severe hypoxia (i.e., hypoxic conditions over > 20,000 km^2), and juveniles moved to deeper, colder, and higher salinity waters during these years (Switzer et al. 2015). Juvenile red snapper still maintained similar peaks in abundance by depth (15–25 m for age-0; 35–55 m for age-1), but more juvenile red snapper were found in depths between 30–45 m (age-0) and 45–70 m (age-1) during years with severe hypoxia. Additionally, Szedlmayer et al. (2014) found hypoxic waters caused nearly a complete loss of juvenile red snapper, and Gallaway et al. (1999) found few red snapper in hypoxic waters.

Gallaway et al. (2009) suggested that age-0 red snapper begin moving to reefs of intermediate relief (1 m^2) when near 100 mm TL and are entirely at these reefs by December of their age-0 year. This is supported by Wells et al. (2008), who showed red snapper juveniles gradually changed their habitat use from sand and low-relief shell (0–0.5 years) to low- and high-relief shell (1 year). After reaching 18 months of age, red snapper recruit to natural outcroppings, oil platforms, and artificial reefs (Gallaway et al. 2009). This conclusion is generally supported by video surveys reported by Powers et al. (2018), who found juvenile red snapper began to move to natural and artificial reefs at 200 mm TL and were fully recruited to reefs by 280 mm.

Young red snapper adults of 2–8 years of age primarily occur on high-relief reefs, particularly oil platforms and various types of artificial reefs (Gallaway et al. 1999). At ≥ 8 years of age, fully mature adult red snapper move to open bottom habitat away from reefs or artificial structures, presumably because predation is no longer a threat (Gallaway et al. 1999). Red snapper may not reach their full reproductive potential as adults until age14–15 (Cowan 2011). Powers et al. (2018) used vertical longline to sample 2- to 8-year-old red snapper at artificial and natural reefs. The highest CPUE was on artificial reefs (0.27 fish/hook/5 min.) compared to natural reefs (0.07 fish/hook/5 min) or unstructured bottoms (0.01 fish/hook/5 min.). Similarly, Karnauskas (2017) showed artificial reefs and oil platforms had 16–20 times the abundance of red snapper at depths ≤ 50 m. The mean age of red snapper collected by habitat were as follows: artificial reefs = 4.9 years, natural hard bottom = 6 years, and unstructured bottoms = 8.6 years. Bottom longline used away from reefs sampled larger and older fish with a mean age of 9.25 years (range 5–42 years) (Powers et al. 2018). They found no difference regarding depth of habitat use for these older fish. Froehlich et al. (2019) tracked adult red snapper for ~ 92 days at an artificial reef offshore of Texas. They found depth used by red snapper increased when temperatures decreased, particularly during cold fronts when temperatures were < 20°C (Froehlich et al. 2019). Emigration from reefs has also been reported following cold fronts elsewhere and fish returned when waters were > 20°C (Topping and Szedlmayer 2011).

Given that red snapper adults use unconsolidated substrates for foraging near artificial reefs, we briefly review movements and foods of adult red snapper. We found the following in regard to movements:

- Red snapper move away from oil platforms and artificial reefs at night and move between platforms (Peabody and Wilson 2006; Topping and Szedlmayer 2011; Williams-Grove and Szedlmayer 2016).
- Monitoring by acoustic receivers showed 72% of adult red snapper stayed at least one year at particular artificial and natural reefs (Topping and Szedlmayer 2011). Movements away from reefs tended to occur at night, and regular movements spanned 2–8 km (Topping and Szedlmayer 2011).
- Froehlich et al. (2019) reported maximum distances moved from capture locations ranged 1–1,038 m. Kernel density home ranges (95%) averaged 77.9 km^2, which is likely to be an underestimate because the spawning season was not included.
- Summarizing long-term movements of tagged subadult and adult red snapper, Patterson et al. (2007) showed mean movements among studies ranged from 0.3–30.9 km with maximum distances of 5–558 km.

In regard to food consumption, Schwartzkopf et al. (2017) examined stomach contents and stable isotopes of red snapper at oil platforms and natural reefs offshore of Louisiana. They found individuals using oil platforms fed primarily in the surrounding soft bottom habitats and in the water column, while red snapper at natural reefs fed at the reefs. Although the reason is unknown, these differences may be due to the underlying substrates, as artificial reefs were surrounded by mud, whereas natural reefs were a patchwork of corals and sand (Schwartzkopf et al. 2017). Soft bottom prey species of red snapper were dominated by fish, including lizardfish and searobin (Schwartzkopf et al. 2017), which have been documented as common species on sand shoals (Rutecki et al. 2014). At artificial reefs offshore of Texas, a comparative analysis of red snapper and gray triggerfish provides additional evidence that red snapper may depend more on non-reef prey associated with mud or sand bottoms (Streich et al. 2018). The results showed red snapper eat diverse foods from site to site, including fish, crabs, gastropods, and stomatopods (Streich et al. 2018). In addition, Szedlmayer and Lee (2004) showed red snapper diet shifted with age toward more reef-associated prey, although some similarities between age groups still existed.

Spawning of red snapper in the nGoM spans from April–Sept 30, with a peak June–August. Highest larvae abundances have been observed at 50–100 m depths, but red snapper larvae have consistently been observed at 100–200 m depths as well (Lyczkowski-Shultz and Hanisko 2007). Larvae are also much more common in waters west of the Mississippi River (Lyczkowski-Shultz and Hanisko 2007).

The spatial distribution of red snapper has been modeled by three primary studies.

1) Most recently, Dance and Rooker (2019) modeled the distribution of red snapper separately for the western and eastern nGoM, delineated by the Mississippi River Delta. They found the following relationships for red snapper in the eastern nGoM:
    - The CPUE of age-0 red snapper was highest at the longitude near Mobile Bay, Alabama and declined eastward, declined with a higher DO level, had an uneven relationship with temperature, peaked at depths of 15–30 m, and declined in close proximity to artificial reefs.
    - The CPUE of age-1 red snapper was highest at the longitude near Mobile Bay, Alabama, and declined eastward, had both positive and negative relationships with DO and bottom temperature, peaked at depths of 15–50 m, and increased in close proximity to artificial reefs.
    - The CPUE of adult red snapper was related to longitude, peaked at depths of 15–60 m, increased in a close proximity to artificial reefs, and increased in close proximity to natural reefs.

    And in the western nGoM, Dance and Rooker (2019) found the following relationships:

    - The CPUE of age-0 red snapper was related to latitude, longitude, increased with higher DO, increased with higher bottom temperature, declined with depths > 50 m, and was lower in close proximity to artificial reefs.
    - The CPUE of age-1 red snapper was related to latitude and longitude, increased with higher DO, increased with higher bottom temperature, peaked at depths of 30–65 m, declined in close proximity to artificial reefs, and was higher in close proximity to natural reefs.
    - The CPUE of adult red snapper was related to latitude, increased with higher DO, increased with greater depth, and was higher within 40 km of artificial reefs.

2) Gruss et al. (2018) modeled the presence/absence of red snapper in the nGoM with four predictor variables: depth, terrain ruggedness index (i.e., geomorphology), percent sand (for juveniles), and percent hard bottom (for adults). Adult red snapper were modeled only for the Florida continental shelf, while juveniles were modeled for the nGoM. The following summarize the findings:

- In contrast to previous studies (see above), Gruss et al. (2018) showed both juveniles and adult red snapper had peak abundance near 100 m in depth and declined substantially at 150–300 m depths.
- Adults and juveniles were both negatively related to terrain ruggedness. This is the opposite of expectations given their life history. This may be a function of including deeper waters over the continental shelf or from missing variables in the analysis (e.g., artificial reefs).
- Juveniles selected for a greater percent of sand substrate, and adults selected for less hard bottom. Given red snapper life history, the relationship with hard bottom is the opposite of expectations.

3) Karnauskas et al. (2017) examined habitat relationships of age-1 and older red snapper. Bottom longline was used to assess non-reef habitats and vertical line surveys estimated relative abundance at artificial reefs and oil platforms (Karnauskas et al. 2017). Depth, longitude, and type of reef (i.e., oil platform, artificial reef, no reef) were used to model the distribution of red snapper. The authors note that the variance explained by the model is quite low. Other habitat factors may have been important, or red snapper may be responding to habitat at a finer spatial scale than the 10 km^2 cell size that they used (Karnauskas et al. 2017). For the nGoM, they estimated that 13.3% of the number of red snapper and 7.8% of the biomass were on artificial structures (**Figure 3-6**).

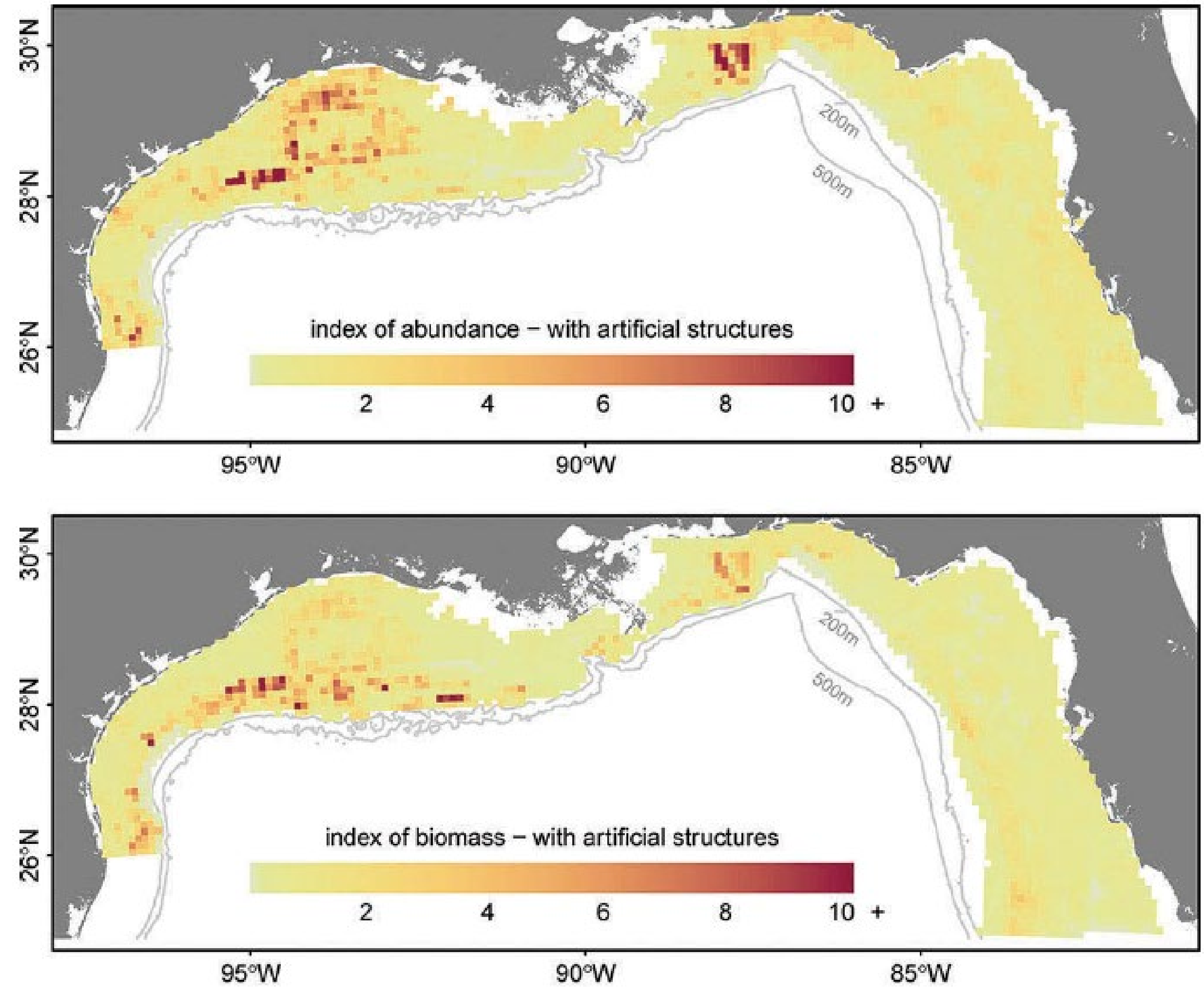


Figure 3-6. Predicted red snapper abundance and biomass for the GoM based on models of abundance accounting for artificial structures.
Source: Adapted from Karnauskas et al. (2017). Reproduced with permission from the Creative Commons Attribution License.

3.1.7 Coastal Migratory Pelagics in the GoM: Spanish Mackerel, King Mackerel, Cobia

3.1.7.1 Spanish Mackerel

The Spanish mackerel is a piscivorous pelagic species that generally uses depths of 10–35 m (Froese and Pauly 2018), although a paucity of published papers exist for them in the GoM. Spanish mackerel in the nGoM likely winter in south Florida or the Campeche-Yucatan region near Mexico, migrate in early spring to the nGoM, and then migrate back to their wintering grounds in the fall (Sutherland and Fable 1980).

Confirming the neritic (i.e., shallow) habitat use of Spanish mackerel, Gruss et al. (2018) analyzed data from offshore of western Florida and found that both Spanish mackerel juveniles and adults had the greatest probability of presence near the shoreline, particularly < 50 km of the shoreline, and that the probability of presence declined steadily moving farther from the shoreline. Schrandt et al. (2016) showed the seasonality of subadult and adult Spanish mackerel, as they were present April–November 30 along shorelines and estuaries spanning from Horn Island, Mississippi to Pensacola, Florida. Juveniles were caught April–August with most caught in May (Schrandt et al. 2016). They also found that 1- to 3-year-old Spanish mackerel used estuarine waters with a wide range of environmental conditions, including sea surface temperatures (SSTs) of 15–34°C, salinities of 0–31%, DO of 2.8–10.8 mg L^{-1}, and depths of 1.8–9 m.

Larvae of Spanish mackerel have been collected offshore of Mississippi and Alabama May–August 31 (Ransom et al. 2016) and offshore of Texas May–September 30 (McEachran et al. 1980). McEachran et al. (1980) also found Spanish mackerel larvae were more abundant in waters < 50 m in depth compared to deeper waters; there was no difference in larvae abundance between depths above or below 35 m. Larvae were captured within a water temperature range of 19.6–29.8°C and salinity range of 28.3–37.4 psu (McEachran et al. 1980). However, we note that no analyses were conducted to determine environmental preferences for temperature or salinity.

3.1.7.2 King Mackerel

The king mackerel is a piscivorous species generally known to use a depth range of 5–140 m (Froese and Pauly 2018), although a paucity of data exists for the GoM. King mackerel in the nGoM likely winter in south Florida from Cape Canaveral to the Florida Keys, migrate in early spring along shallow waters into the nGoM, and then migrate back to their wintering grounds in the fall (Sutherland and Fable 1980; Sutter et al. 1991).

Juvenile and adult king mackerel offshore of western Florida have the highest probability of presence with SSTs of 14–23°C and a sharp decline in presence at SST > 23°C (Grüss et al. 2018). Wall et al. (2009) found the CPUE of king mackerel caught in recreational fishing tournaments was positively related to temperature fronts within 10 km, as well as chlorophyll concentration. In a study of fish otoliths, Dzaugis et al. (2017) showed king mackerel growth rates were highest in years when the spring season warmed rapidly, as evidenced by relatively warm SST in March and winds from the south and east during this time.

Serial spawning of king mackerel occurs from May to early October, with a peak in September in the nGoM (Grimes et al. 1990; McEachran et al. 1980). Offshore of Texas, McEachran et al. (1980) analyzed ichthyoplankton surveys ranging depths of 12–139 m. They found king mackerel larvae were more abundant at ≥ 35 m depths compared to depths of < 35 m; larvae abundance did not differ above or below 50 m depths, indicating that larvae were relatively common at 50–183 m depths. Furthermore, larvae were captured within a water temperature range of 19.6–29.8°C and salinity range of 28.3–37.4 psu (McEachran et al. 1980). However, we note that no analyses were conducted to determine environmental preferences for temperature or salinity.

3.1.7.3 Cobia

Cobia migrate from their wintering grounds off south Florida into the northeastern GoM during early spring (Dippold et al. 2017; Franks and Brown-Peterson 2002). More specifically, they occur offshore of northwest Florida, Alabama, Mississippi, and southeast Louisiana from late March through October (Biesiot et al. 1994; Franks et al. 1999). In stomach content analyses, cobia have been found to have demersal feeding habits. Meyer and Franks (1996) found diets of cobia captured between Louisiana and Florida were dominated by crustaceans (79% of stomachs). Portunid crabs were the most frequently found prey (73% of stomachs) followed by hardhead catfish *Arius felis* (24% of stomachs), eels (18% of stomachs), Sicyoniidae (prawns) and Penaeidae (shrimp) (9.6% of stomachs), and stomatopods/Squilla spp. (mantis shrimp) (6.9% of stomachs) (Meyer and Franks 1996).

Most recently, Gruss et al. (2018) showed cobia offshore of western Florida had the greatest probability of presence in shallow waters (0–50 m being the highest) and probability declined steadily at greater depths; peak presence was found when chlorophyll-*a* concentrations were ~3–10 mg m^{-3}.

As summarized by Franks and Brown-Peterson (2002), cobia spawning season in the nGoM has been reported as April–September 30 with populations near Texas possibly starting to spawn by May. Cobia spawn on multiple occasions per year (Biesiot et al. 1994), yet spawning locations are unknown for the GoM (Franks and Brown-Peterson 2002). Cobia larvae collected by Ditty and Shaw (1992) throughout the GoM were found in both estuary and offshore marine environments with SSTs ranging 24–32℃ and depths ranging of 3–300 m. We note that the sampling they used was limited and opportunistic. Observations from the Atlantic population may provide further clues regarding spawning locations. Observations have suggested that cobia spawn at the mouth of Chesapeake Bay and immediately offshore of Virginia Capes (Joseph et al. 1964), generally near inlets (Smith 1995), inshore (Lefebvre and Denson 2012), and offshore (Shaffer and Nakamura 1989).

3.1.8 Shrimp in the GoM

In order of economic importance (greatest to least), shrimp species in the nGoM Shrimp Fishery Management Plan (1981) include brown, white, pink, royal red, rock, and seabob shrimp. Of these, the Penaeid species of brown, pink, and white shrimp represent the bulk of the commercial shrimp landings. These species all have estuary-dependent life stages and complex life cycles. For these species, growth and survival of shrimp within estuaries is primarily dependent on salinity and temperature; as shrimp grow, they shift to deeper waters and become more predatory (Gulf of Mexico Fishery Management Council 1981). The timing of emigration to the nGoM is dependent on size, tide, and temperature (Gulf of Mexico Fishery Management Council 1981). Growth continues rapidly under ideal temperatures and spawning probably occurs before shrimp are 1 year old (Gulf of Mexico Fishery Management Council 1981). Because of the complexity involved in the various life stages of Penaeid shrimp, we have detailed their general habitat use and notable dates of each life stage (**Figure 3-7**, **Table 3-2**).

The importance of estuarine habitats to shrimp production is exemplified by Turner (1977), who found strong relationships between commercial yields of brown shrimp harvested in inshore waters and hectares of intertidal wetlands. Zimmerman et al. (2002) has since reviewed studies of brown and white shrimp estuarine relationships and found that marsh-edge habitat does support a greater abundance of shrimp compared to nonvegetated waters. However, densities of shrimp in submerged aquatic vegetation (SAV) have been documented as being lower, higher, or similar to marsh-edge habitats (Clark et al. 2004; Glancy et al. 2003; Zimmerman et al. 2002). Therefore, both SAV and marsh edges are likely important to shrimp in estuaries; the importance of estuarine habitat may be geography specific.

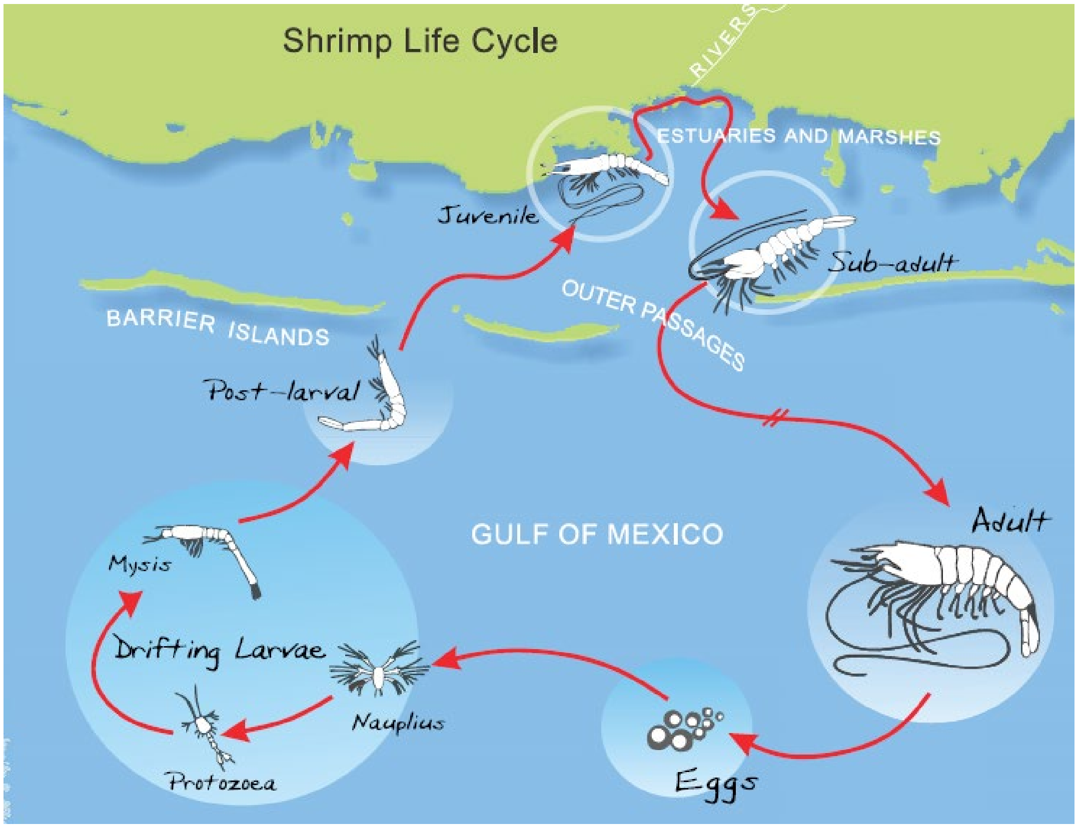


Figure 3-7. Life cycle of brown and white shrimp in the GoM.
Source**: A**dapted from Spies et al. (2016) and diagram by J.R. Allen. Reproduced with permission from the University of Southern Mississippi journal, *Gulf and Caribbean Research*.

Gruss et al. (2018) modeled the distribution of shrimp as a single group, which included pink, brown, white, and rock shrimp, plus five other species. Shrimp had a higher probability of presence with less sandy sediments and peaked when mud was 20–40% of the sediments. Shrimp presence increased as depth increased (highest increase was 0–50 m, but shrimp still increased up to 300 m); abundance peaked when bottom temperatures were 10–15°C, although abundances were still high at 15–30°C (Grüss et al. 2018).

3.1.8.1 Brown Shrimp

Montero et al. (2016) developed a species distribution model for brown shrimp using depth, percent mud, salinity, SST, DO, season, latitude, and longitude. In order of importance (highest to lowest), the species-habitat relationships showed a greater CPUE of brown shrimp with a greater proportion of mud sediments, depths of 20–100 m, bottom salinity of 10–20 psu, SST of 23–30°C, and DO of 2–7 ppm (Montero et al. 2016). Seasonality did not affect CPUE, but latitude and longitude were substantial factors. The predictive model showed a much higher brown shrimp abundance offshore of Mississippi, Louisiana, and Texas compared to the shelf of Alabama and Florida. During summer, a higher density was observed nearshore, while fall populations were more often further offshore (Montero et al. 2016). The authors note that this pattern might be because of emigration of late juveniles from the estuaries during the summer months. The offshore habitat use in the fall might characterize the species' spawning habitat because peak spawning is thought to be September–November 1 in water depths of 27–100 m(Renfro and Brusher 1982). The association with a relatively low salinity is related to the proximity to estuaries and major river outflows (Montero et al. 2016).

Table 3-2. Federally managed shrimp species and life history as depicted from the Gulf of Mexico Fishery Management Plan (1981).

Species	Life stage	Ecosystem	Depth and geomorphology relationships	Notable dates
Brown shrimp	Spawning	GoM	Depths: 18–110 m Substrate: mud, silt, or mud/sand/shell (TX–AL)	Spawning peaks: Sept–Nov 1 (major) & April–June 1 (minor)
Brown shrimp	Free-swimming larvae	GoM	-	Peak: Aug–Nov 1 (TX)
Brown shrimp	Postlarvae	Recruits to estuaries	-	Peaks: March–April 15 & June–Sept 30 (TX); Feb–March 1 (LA)
Brown shrimp	Adult	Moves to GoM	May stopover in open bays on way to GoM	Peaks: May–Aug 31 (TX); June–July 30, range of May 15–Nov 1 (LA)
White shrimp	Spawning	GoM	Depths: 7–31 m, possibly near inlets Substrate: mud, silt, clay/shell, sand/shell (TX–AL)	Spring through fall (multiple spawning events)
White shrimp	Free-swimming larvae	GoM	-	
White shrimp	Postlarvae	Recruits to estuaries	-	Peaks: May–Oct 31 (MS) June–Aug 1 (LA) May–Oct 31 (TX)
White shrimp	Adult	Moves to GoM	-	Sept–Nov 30 OR Oct–Dec (fall spawning, with cold fronts)
White shrimp	Adult (for small, late spawned individuals)	Moves back to estuary, then returns to GoM to spawn	-	Estuary in early spring; Returns to GoM spring/early summer
Rock shrimp	All life stages	GoM	Depths: 18–82 m Substrate: calcareous sediments (FL); sandy bottoms	Spawning continuous; peaks Oct–Jan 31 (FL)
Pink shrimp	Spawning	GoM	Depths: 22–48 m Substrate: calcareous sediments (FL); firm mud, silt, sands with shells	FL Keys: most intense spring to fall (year-round) FL / Tampa: summer
Pink shrimp	Free-swimming larvae	GoM	-	-
Pink shrimp	Postlarvae	Recruits to estuaries	-	FL Keys: Peaks April–June 1 & July–Oct 31 (year-round) MS: May–Dec 31 TX: peak Aug–Sept 1
Pink shrimp	Adult	Moves to GoM	-	-
Seabobs	All life stages	GoM	Depths: 0–9 m, mostly at 1.8–3.6 m Substrate: mud, silt, silt/s and bottoms (TX–AL)	Spawning July–Dec 1

Note: See **Figure 3-7** for a visual of the life stages.

Craig (2012) investigated the effect of hypoxic conditions on brown shrimp. During hypoxic conditions (DO < 2.0 mg L^{-1}), brown shrimp were concentrated along the edge of the hypoxic zone (i.e., farther inshore and offshore), particularly within 0–5 km of the hypoxia. The shrimping fishery also showed a pattern of moving farther inshore and offshore near the edge of the hypoxic zone (Purcell et al. 2017). Craig (2012) also showed the strength of association between DO and brown shrimp decreased in years with less hypoxia. Notably, strong gradients of DO were found across relatively short distances of 1–5 km, as oxygenated waters near shoals were surrounded by hypoxic waters (Craig 2012). During years of moderate hypoxia, Craig and Crowder (2005) found brown shrimp abundance peaked with the following: depths of 40–100 m, in relatively close proximity to the shoreline, temperatures of 18–27°C, DO >2 mg L^{-1}, and salinities >32 psu. For years with severe hypoxia, brown shrimp habitat associations changed. They shifted in latitude to a more narrow depth range (peaking at 30–70 m), to both cooler and warmer waters, and to salinities < 20 psu (i.e., characterizing inshore movements) (Craig and Crowder 2005).

In addition to annual variation resulting from hypoxic conditions, brown shrimp habitat use has been shown to vary with their density offshore of Texas (Craig et al. 2005). In years with relatively high abundance, Craig et al. (2005) found brown shrimp used waters farther offshore, which extended their distribution beyond their more consistently used shallow waters. Furthermore, annual SST may affect brown shrimp abundance. Li and Clarke (2005) found a strong relationship of annual trawl catches and SST of shelf waters in April and May, as brown shrimp catches increased in years with relatively high temperature during this time. They concluded the SST anomalies measured in April and May likely affect brown shrimp because of its high variability and its link to growth and survivorship of brown shrimp in estuaries.

3.1.8.2 Pink Shrimp

Drexler and Ainsworth (2013) modeled the distribution of pink shrimp for the nGoM. Importantly, they found that aggregating the predictions into broad polygons, rather than the finer resolution of the original data, improved model validation. Drexler and Ainsworth (2013) documented the following habitat relationships:

- Peak abundance of pink shrimp was at bottom temperatures of 17–32°C; few samples were beyond this range, but a negative association was observed at < 15°C.
- Sand and rock substrates had greater densities of pink shrimp compared to mud or gravel.
- A slight peak in abundance was observed at depths of ~25–40 m, abundance was relatively high at 41–119 m, then a sharp decline in abundance was predicted at depths > 120 m.
- There was a slight decrease in predicted pink shrimp abundance as chlorophyll increased, then a sharp decline was predicted when chlorophyll was > 15 mg m^{-3}.
- An increase in abundance was predicted when DO was > 5 mL L^{-1}, but very few samples were observed in this range.

Rubec et al. (2016) created qualitative habitat suitability maps based on pink shrimp fishing vessels on the western Florida continental shelf during the spring and fall. Depths up to 50 m were trawled. They noted that shrimping mostly occurred on offshore sand ridges because those areas were most trawlable; non-trawlable substrates included hard bottom or mixed hard bottom habitats. Relatively high pink shrimp CPUE occurred in waters with a northwest origin of bottom currents in the spring, higher current velocities, and bottom types of fine sand and mud (compared to medium sand, coarse sand, and gravel). In support of the relationship with current velocity, simulations of pink shrimp larvae transport have found that larvae and postlarvae migration to estuaries is likely dependent on a combination of tidal and wind-driven currents (Criales et al. 2006). Zink et al. (2017) reviewed the relationship of salinity to pink shrimp postlarval, juveniles, and subadults; they found a wide tolerance to salinity. Pink shrimp survivorship remained high at 15–40 psu and a majority of the reviewed studies reported maximal abundance between ~20–35 psu (Zink et al. 2017).

3.1.8.3 White Shrimp

Adult white shrimp habitat relationships are poorly studied in the GoM, although white shrimp have been found to be abundant on and adjacent to sand shoals offshore of the Atlantic Coast of Florida (Zarillo et al. 2009). In a study offshore of Louisiana, Diop et al. (2007) analyzed white shrimp landings over time and found landings were positively, but weakly, correlated with late juvenile abundances. Furthermore, a moderately strong, negative relationship was found between adult white shrimp CPUE and temperature during the late juvenile phase (Diop et al. 2007). Other relationships discovered included a weak relationship of a reduced adult CPUE when river flow was high and when fishing effort was high (Diop et al. 2007). Overall, the availability and condition of juvenile habitats may affect offshore abundance and subsequent habitat use of white shrimp.

The density of juvenile white shrimp are similar at marsh edges and subaquatic vegetation habitats (Rozas and Minello 2006). In terms of juvenile white shrimp density, growth, and secondary production, an analysis of estuarine habitats showed that edges of saline and brackish marshes of Louisiana had a higher value (per area) compared to oligohaline marsh edges (Mace and Rozas 2017). As expected, density of white shrimp was highest ≤ 1 m from the marsh edge and declined as distance from the edge increased (Mace and Rozas 2017). Baker et al. (2014) analyzed a stage-based population model for white shrimp and concluded that the population was most sensitive to survival rates in the early life stages.

3.1.8.4 Royal Red, Seabob, and Rock Shrimp Habitat Associations

Royal red, seabob, and rock shrimp do not have an estuary-dependent life stage. Royal red shrimp are found and harvested from approximately 180–730 m in depth (Gulf of Mexico Fishery Management Council 1981; Stiles et al. 2007). Because this depth range is well beyond our study area (≤ 50 m), we do not provide further information on royal red shrimp.

Rock shrimp are primarily harvested from Florida’s sandy bottoms, although their range extends across the entire nGoM (**Figure 3-8**) (Gulf of Mexico Fishery Management Council 1981; Stiles et al. 2007). They are nocturnal and present year-round but are suggested to be most available for harvest July–November 30, with the greatest abundance being in September (Stiles et al. 2007). After a particularly large harvest of rock shrimp in 1996, trawling vessels began to enter juvenile grounds, including the deep-sea coral ecosystem of Oculina Reef in the South Atlantic (Stiles et al. 2007). At this time, much of the juvenile population was thought to have been harvested, and damage has occurred to the deep-sea reef (South Atlantic Fishery Management Council 1996).

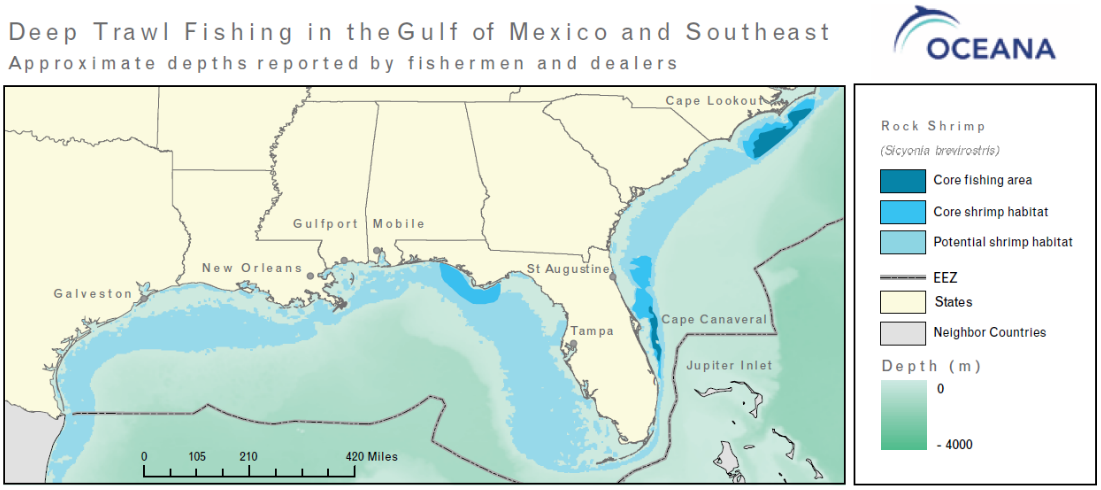


Figure 3-8. Potential distribution of rock shrimp based on interviews, FMC documents, and scientific literature where available.
Source: Adapted from Stiles et al. (2007) with open access permissions.

Seabob shrimp are harvested alongside white shrimp October–December 31 as they migrate from deeper waters to the shoreline in response to cold fronts (Gulf of Mexico Fishery Management Council 1981). Little is known about seabob shrimp. In the southern GoM, seabob shrimp were found in the shallowest of waters spanning ~0–20 m (Castrejon et al. 2005).

3.2 South Atlantic Fish Habitat Associations and Seasonality

Key Points and Knowledge Gaps *(gaps are in italics)*

- In the South Atlantic, federally managed species that are socio-economically important and likely to be common on sand shoals include king mackerel, Spanish mackerel, summer flounder, shark species, black sea bass, and possibly other hard bottom fish.
- Most research on South Atlantic marine fish has focused on depth and latitude, whereas relationships with geomorphic features and oceanographic conditions are poorly known. *Because of the lack of surveys over soft substrates, the differences in fish species composition between hard bottom and soft substrates has not been well documented for reef fish.*
- Juvenile reef-associated fish do use unconsolidated sediments extensively, although *research and monitoring specific to shoals is limited*. Black sea bass, bank sea bass, and grunts have been documented in unconsolidated sediment habitats. *Little is known about the use of sandy habitats by juvenile red snapper and lane snapper, which have been linked to those habitats in the Gulf of Mexico.*
- Adult reef-associated fish that use natural and artificial structures likely forage in surrounding soft substrates, but the *extent of this remains uncertain*.
- *The shallow water habitats of the coastal migratory pelagic species (Spanish and king mackerel) are primarily unstudied.*

3.2.1 Introduction to the South Atlantic Physical Setting and Fish

Two broad geographies are recognized in the South Atlantic: the South Atlantic Bight extends from Cape Hatteras, North Carolina to Cape Canaveral, Florida; and the southeastern Florida Coast extends from Cape Canaveral to the Florida Keys (Dame et al. 2000). The most prominent feature of the South Atlantic is the Gulf Stream current (**Figure 3-9**), which creates a cross-shelf mixing of waters and strong water stratification (Castelao 2011).The Gulf Stream can intrude upon shelf waters by surface water intrusion, interlayering, or via a bottom water intrusion (Atkinson 1977). Surface and bottom water intrusions are most frequent in the summer months, are influenced by wind stress, interact with salinity, and are more frequent at the extreme northern and southern waters of the South Atlantic Bight (Castelao 2011).

The coast of the South Atlantic has several series of barrier islands with associated inlets, estuaries, and wetlands that contribute to the high productivity of the coastal environment (Dame et al. 2000). These estuarine waters support marine fish by acting as a nursery and as juvenile habitats for species such as sharks (Castro 1993; Curtis et al. 2013; McCallister et al. 2013), red drum, shrimp, and flounder. Estuaries are also used by prey species like crabs and menhaden. Estuarine waters in the South Atlantic are threatened by modifications to freshwater inflow, runoff, and pollution (Dame et al. 2000).

Sediments in the shallow waters (≤ 50 m) of the South Atlantic primarily range from very fine sand to very coarse sand (Conley et al. 2017). Shoals and shoal complexes of the South Atlantic include large cape-associated shoals such as Cape Lookout Shoals (NC), Frying Pan Shoals (NC), and Canaveral Shoals (FL) (Rutecki et al. 2014) (see Volume 2 for details of geoform classification). Hundreds to thousands of bedform shoals are also present in the region. Natural hard bottom reefs, or live bottom reefs, consist of rocky outcrops that support sessile invertebrates such as sponges and sea fans (Miller and Richards 1980). These habitats support a diverse reef fish assemblage.

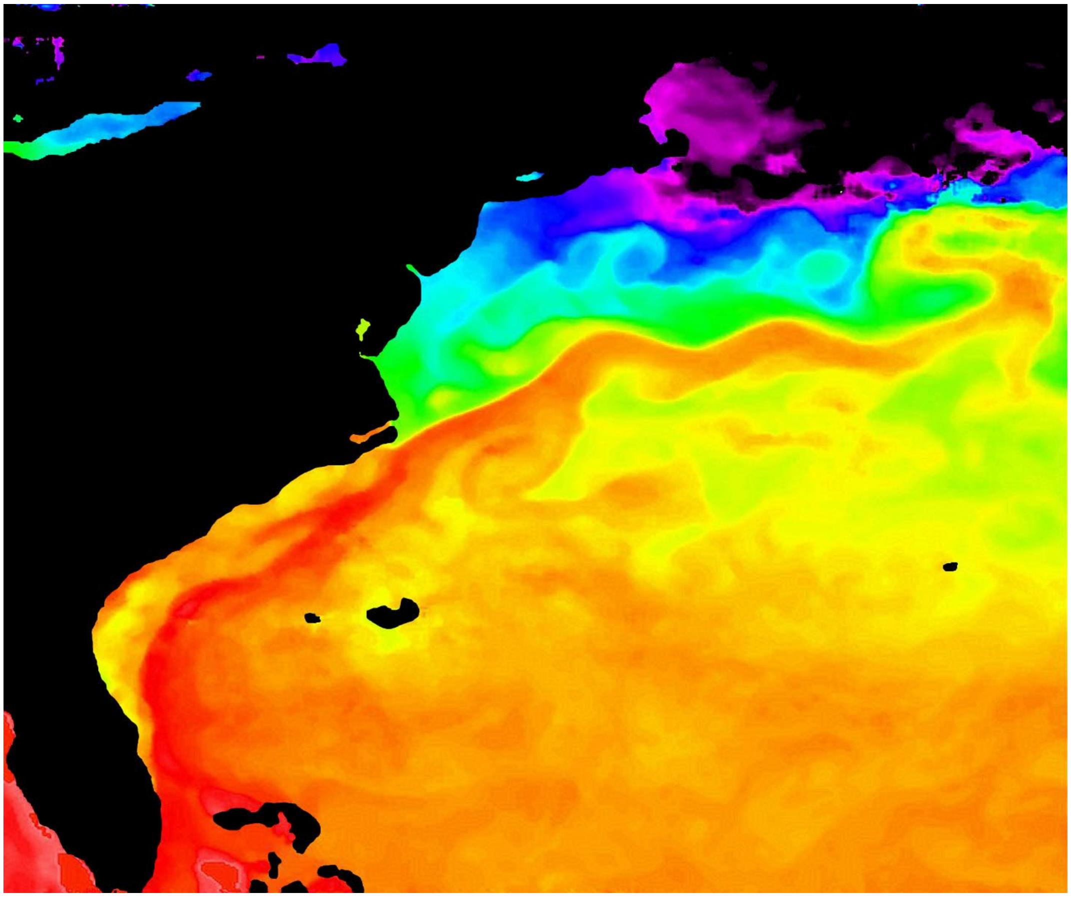
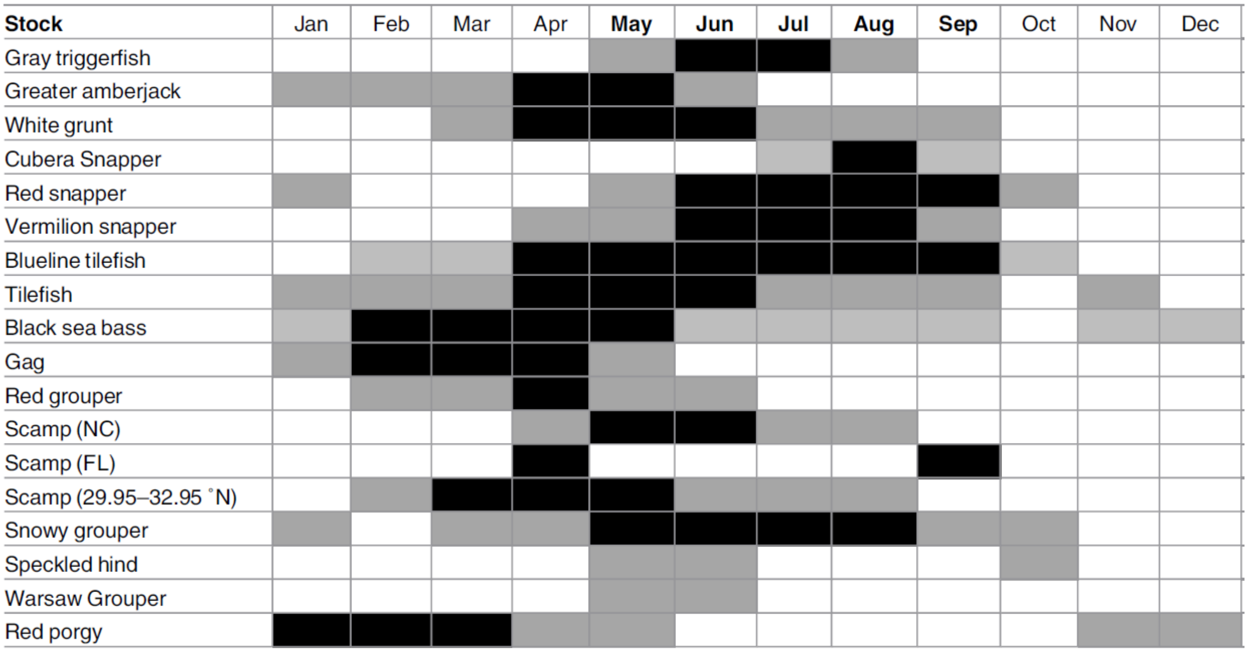


Figure 3-9. The Gulf Stream current depicted in red moving from southern Florida toward the northeast.
Source: Adapted from NASA Jet Propulsion Laboratory.

In regard to prey species, Okey et al. (2014) simulated changes in forage species, or species groups, that were identified through a South Atlantic Marine Bight "Ecopath with Ecosim" food web model. They tested the effect of changes in anchovies, Atlantic menhaden, Atlantic silverside, halfbeaks, mullets, sardines, scads, shad, thread herring, pelagic-oceanic planktivores, squid, and shrimp. The results showed Atlantic menhaden and squids had the largest effects on higher trophic levels, although the effects varied by predatory species (Okey et al. 2014). As an example, they found an increase in Atlantic menhaden was projected to strongly increase biomass of striped bass, bluefish, large coastal sharks, small coastal sharks, and highly migratory species.

The National Marine Fisheries Service (2017) provides a fisheries summary for the South Atlantic region, and we provide a synopsis of the report here. For commercial fisheries of the South Atlantic, key federally managed species include groupers, snappers, flounders, king mackerel, swordfish, and tuna. For recreational fisheries, federally managed species of economic importance include black sea bass, bluefish, dolphinfish, king mackerel, sharks, sheepshead porgy, and Spanish mackerel. Within individual states, additional federally managed species of economic importance include gray snapper, porgies (sheepshead), summer flounder, striped bass, and tilefish. As of 2018 in the South Atlantic, the stocks, or stock complexes, of red snapper, red porgy, red grouper, snowy grouper, Warsaw grouper, speckled hind, hogfish (southeast Florida), tilefish, and blueline tilefish are considered overfished, or overfishing is occurring (NOAA Fisheries 2019). The seasonality of reef fish spawning is highly variable, but peak spawning is usually April–August (**Figure 3-10**). The overlap of EFH with our South Atlantic study area is summarized in **Table 3-3** and includes the broad snapper-grouper complex.

In addition to mapped EFH, the South Atlantic FMC has designated the federally managed dolphinfish and wahoo as having EFH that includes the Gulf Stream, Charleston Gyre, Florida Current, and pelagic *Sargassum*. These features are not within our specific study area and do not overlap with past sand dredging locations. Therefore, we not review dolphinfish and wahoo further.



Stock	Jan	Feb	Mar	Apr	**May**	**Jun**	**Jul**	**Aug**	**Sep**	Oct	Nov	Dec
Gray triggerfish					▨	■	■	▨				
Greater amberjack	▨	▨	▨	■	■	▨						
White grunt			▨	■	■	■	▨	▨	▨			
Cubera Snapper							░	■	░			
Red snapper	▨				▨	■	■	■	■	▨		
Vermilion snapper				▨	▨	■	■	■	▨			
Blueline tilefish		░	░	■	■	■	■	■	■	░		
Tilefish	▨	▨	▨	■	■	■	▨	▨	▨		▨	
Black sea bass	░	■	■	■	■	░	░	░	░		░	░
Gag	▨	■	■	■	▨							
Red grouper		▨	▨	■	▨	▨						
Scamp (NC)				▨	■	■	▨	▨				
Scamp (FL)				■					■			
Scamp (29.95–32.95 ˚N)		▨	■	■	■	▨	▨	▨				
Snowy grouper	▨		▨	▨	■	■	■	■	▨	▨		
Speckled hind					▨	▨				▨		
Warsaw Grouper					▨	▨						
Red porgy	■	■	■	▨	▨						▨	▨

Figure 3-10. Timing of spawning (gray) and peak spawning (black) for selected species in the snapper-group EFH group of the southeastern Atlantic Ocean of the US.
Source: Adapted from Farmer et al. (2017). Reproduced with open access permission from PlosOne.

Table 3-3. For federally managed species within the South Atlantic, the proportion of area designated as EFH within each study area.

Fishery Management Council	Species or EFH species group	Proportion of South Atlantic study area designated as EFH	Fishbase description of habitat
South Atlantic	Snapper-grouper	**0.95**	reef-associated
South Atlantic	Corals	0.08	NA
South Atlantic	Spiny lobster	**0.95**	NA
South Atlantic	Coastal migratory pelagics	0.48	-
South Atlantic	- Spanish mackerel	-	pelagic 10–35 m
South Atlantic	- King mackerel	-	reef-associated 5–140 m usually 5–15 m
South Atlantic	- Cobia	-	reef-associated 0–1,200 m
Mid-Atlantic	Atlantic butterfish	0.24	benthopelagic 15–420 m usually ≤55 m
Mid-Atlantic	Atlantic mackerel	0.07	pelagic 0–1,000 m usually 0–200 m
Mid-Atlantic	Atlantic surfclam	0.04	NA
Mid-Atlantic	Black sea bass	0.15	reef-associated ≥1 m
Mid-Atlantic	Bluefish	**0.95**	pelagic 0–200 m
Mid-Atlantic	Longfin inshore squid	0.17	NA
Mid-Atlantic	Northern shortfin squid	0.03	NA
Mid-Atlantic	Ocean quahog	0.01	NA
Mid-Atlantic	Scup	0.17	demersal ≥15 m
Mid-Atlantic	Spiny dogfish	0.10	benthopelagic 0–1,460 m usually 50–300 m
Mid-Atlantic	Summer flounder	**0.92**	demersal 10–183 m usually ≤37 m

Notes: Species with EFH overlapping with > 50% of the study area are bolded. Study area is defined in **Figure 1-3**. All life stages were included. NA = information not applicable

3.2.2 Snapper-Grouper Complex of the South Atlantic

Containing 55 species of snapper, grouper, and related species, the snapper-grouper complex (also known as “reef fish”) is most closely associated with hard bottom and artificial reefs in the South Atlantic. With hard bottom scattered from nearshore to the outer slope, the EFH designations include a broad geographic coverage.

Walsh et al. (2006) studied reef fish habitat use of unconsolidated sediment substrates by conducting trawl surveys offshore of Georgia. They classified 121 of 181 fish species as being a juvenile life stage, and they suggest that unconsolidated sediments of the continental shelf are important for early life stages of reef-associated species. In particular, Walsh et al. (2006) reported consistent occurrence of bank sea bass sand perch, and *Stenotomous* spp. (scup, porgy). Reef-associated species with irregular use of soft sediments included black sea bass, snowy grouper, short bigeye, Atlantic bigyeye, and mutton snapper. In the sampling of 0–70 m depths, the most common federally managed species had the following depth and season associations:

- Bank sea bass were associated with 40–70 m depths during spring, 0–70 m depths during summer, and 40-70 m depths during winter. They were not observed in fall. Among those recorded, juveniles were common.
- Black sea bass were associated with 0–40 m depths during spring and 0–70 m depths during summer. They were not observed in fall or winter. Juveniles and adults were recorded.
- *Stenotomus* spp. (scup or porgy) were observed over 0–40 m depths during spring, 0–70 m during summer, and 20–40 m depths during fall and winter. Among those recorded, juveniles were common.
- Mutton snapper were only observed during summer, were observed over the entire 0–70 m depths, and were mostly composed of juveniles.

Zarillo et al. (2009) conducted trawl surveys within and adjacent to sand shoals offshore of Florida's Atlantic Coast. The most common (≥ eight occurrences) federally managed snapper-grouper species were rock sea bass, porgy, juvenile grunt, and Atlantic spadefish. The next most common species (two to three occurrences) were white grunt and black sea bass (Zarillo et al. 2009). Gilmore (2008) notes that goliath grouper have been observed in spawning aggregations on shoals or adjacent to shoals. From wide-ranging fish sampling, Miller and Richards (1980) note that subtropical reef-associated fish species extend in a narrow productive zone of 33–40 m depths from North Carolina to Florida. They suggested that both inshore and farther offshore cooling occurred in winter and limited species' distributions. Their data showed catch of commercial hard bottom fish species occurred primarily at 33–41 m depths (69% of total weight) with 24–32 m depths having a high catch rate as well (16% of total weight).

Two separate stocks of black sea bass have been identified, with the population splitting into a Mid-Atlantic/northeast Atlantic population and a South Atlantic population that occurs south of Cape Hatteras, North Carolina (McCartney et al. 2013; Roy et al. 2012). In an extensive analysis of trap data, Bacheler and Ballenger (2015) found small and large black sea bass were similarly related to depth (peak CPUE at < 30 m) and bottom temperature (peaked at 12–15°C, but had a broad temperature range). However, the proportion of small individuals was greater in the most shallow waters. Although surveys were concentrated solely on hard bottom habitats (i.e., no locations surveyed were away from hard bottoms), black sea bass were not associated with rugosity, slope of slope, or available hard bottom maps (Bacheler and Ballenger 2015).

The red snapper has been poorly studied in the South Atlantic, as 94% of red snapper studies are from the GoM, and no research on juveniles has been published in the US Atlantic waters (Rindone et al. 2015). Regarding the nGoM, sand shoals and unconsolidated substrates with some structural complexity provide habitats where juvenile red snapper can be abundant (see Gulf of Mexico section above). In the South Atlantic, the red snapper fishery has been closed since 2010. From trap and camera data of hard bottom locations offshore of Florida and Georgia, Coggins Jr et al. (2014) found red snapper were related to depth and latitude. In their study, red snapper were estimated to be present at 45% of the sites surveyed, were most abundant at 20 m depths, and declined in abundance through 60 m depths. The predictive model showed probability of presence and abundance to be highest surrounding Cape Canaveral (and its shoals) and northward to the Florida-Georgia state boundary (Coggins Jr et al. 2014).

Fish habitat use of sand sediments adjacent to reef structures was examined offshore of southeast North Carolina (Rosemond et al. 2018). Rosemond et al. (2018) found that daytime fish habitat use of sand substrate declined in terms of abundance, species richness, and biomass as distance to reef increased up to a maximum of 90 m from the reef. Community composition changes were due to decreases of common planktivores and increases in transient pelagic predators. Benthic carnivores included black sea bass, scup, and slippery dick (Rosemond et al. 2018). Regarding the effect of survey timing, Wenner (1983) found twice as many reef-associated species were captured at night over sandy substrate compared to daytime surveys. Species such as tomtate moved away from live bottom habitats during night hours. Black sea bass have been primarily caught at night as well (Sedberry and Van Dolah 1984).

Bacheler et al. (2016) examined hard bottom fish in the South Atlantic, and the most common fish surveyed by video were white grunt, black sea bass, red snapper, red porgy, Vermilion snapper, Almaco jack, gray triggerfish, greater amberjack, scamp, lionfish, gag, gray snapper, and hogfish. All other species were observed in < 5% of surveys. Relationships among fish species presence were quantified with the variables of longitude/latitude, depth, amount of hard bottom present in videos, water clarity, and current direction in relation to the camera (Bacheler et al. 2016). Latitude/longitude (factor in 69% of fish species' statistical models), depth (66%), and amount of hard bottom (72%) were most commonly factors in species' distributions; currents (38%) and water clarity (22%) were less common factors (Bacheler et al. 2016). Of particular relevance to unconsolidated sediment habitats, the following trends were observed:

- Black sea bass were most common with depths of < 30 m, but were present in waters up to 80 m in depth.
- Red snapper were found at all depths surveyed (10–110 m) with higher abundances with < 60 m depths.
- White grunt were most common with 20–40 m depths and only ranged 10–60 m depths.
- Lane snapper were much more common with depths < 30 m but were present in depths up to 60 m.

Spatial models were developed using only the latitude/longitude variables, as the other variables were not spatially explicit. The deviation explained varied widely from 4.2% to 75% of variation explained in fish species distribution (Bacheler et al. 2016).

Farmer et al. (2017) synthesized spawning reef fish locations and timing in the South Atlantic (**Figure 3-10**, **Table 3-4**). Most of the variability in the prediction of spawning locations was explained by the temporal variables of month, year, and lunar phase. Red snapper had 5% of the variation explained by substrate curvature, and black sea bass were strongly related to temperature and latitude (58% deviance explained). White grunt spawning was related to latitude and depth (23% deviance explained).

Table 3-4. Summary statistics of spawning females of reef-associated species in the South Atlantic

Species	Mean depth (range)	Mean salinity (range)	Mean water temperature (range)
Black sea bass	25.4 m (15–66 m)	35.7 psu (34–40 psu)	19.5°C (11–27°C)
Red snapper	43.2 m (23–66 m)	36.2 psu (35–37 psu)	22.6°C (17–28°C)
White grunt	33.3 m (22–52 m)	36.1 psu (35–37 psu)	23.7°C (18–27°C)

Notes: Reported from Farmer et al. (2017).

3.2.3 Coastal Migratory Pelagics of the South Atlantic: Spanish Mackerel and King Mackerel

3.2.3.1 Spanish Mackerel

The Spanish mackerel is a piscivorous, pelagic species that commonly use depths of 10–35 m (Froese and Pauly 2018), although only a few studies have focused on them in the South Atlantic. As in the GoM population, Spanish mackerel in the South Atlantic likely winter in south Florida. Schmidt et al. (1993) showed the reproductive chronology of female Spanish mackerel, as sampled from Beaufort, North Carolina, to Riviera Beach, Florida. They found the spawning season ranged primarily from May–August 31. Similarly, Spanish mackerel are common in the Chesapeake Bay from late April to early October (Chittenden et al. 1993). In the South Atlantic, the Marine Resources Monitoring, Assessment, and Prediction (MARMAP) program (Collins and Stender 1987) sampled ichthyoplankton in depth ranges spanning 9 to 3,490 m and found Spanish mackerel larvae in waters ranging 11–29 m in depth. In their limited sampling of Spanish mackerel larvae, there was no overlap with the smallest of king mackerel larvae. No latitudinal changes in abundance were observed in the South Atlantic (Collins and Stender 1987). Adding to the sparse data on habitat use, larvae and juvenile Spanish mackerel have been captured at Breach Inlet near Charleston, South Carolina, and along the nearshore of North Carolina, South Carolina, and Georgia from May–October (Peters and Schmidt 1997). Studies of more specific South Atlantic habitat occurrence of Spanish mackerel juveniles and adults are lacking.

3.2.3.2 King Mackerel

The king mackerel is a piscivorous species that generally uses a depth range of 5–140 m (Froese and Pauly 2018). In their wintering grounds, Atlantic king mackerel overlap with GoM king mackerel beginning near St. Augustine, Florida, and extending southwest of Naples, Florida (Clardy 2008). Studies of more specific habitat use of king mackerel juveniles and adults are lacking.

King mackerel tagged near Ft. Pierce, Florida, provided evidence that king mackerel migrated northward from Florida as far as North Carolina (Sutter et al. 1991). Primarily supporting these results, Trent et al. (1987) examined CPUE of king mackerel from charter boats and did not observe a temperature-dependent migration. Instead, they showed the approximate peak captures as represented in **Figure 3-11**.

In the South Atlantic, serial spawning occurs from April through early October, with a peak in September (Collins and Stender 1987; Finucane et al. 1986; McEachran et al. 1980). Larvae have also been collected in November in these waters (Collins and Stender 1987). Through the MARMAP program, king mackerel larvae were more abundant at depth ranges of 21–200 m compared to more shallow waters (only 2 of 175 surveys in depths $\leq$ 20 m had king larvae). Larvae were also abundant between the latitudes of 32–33ºN, which may be attributed to a region of upwelling produced by the “Charleston Bump” topographic ridge (Collins and Stender 1987).

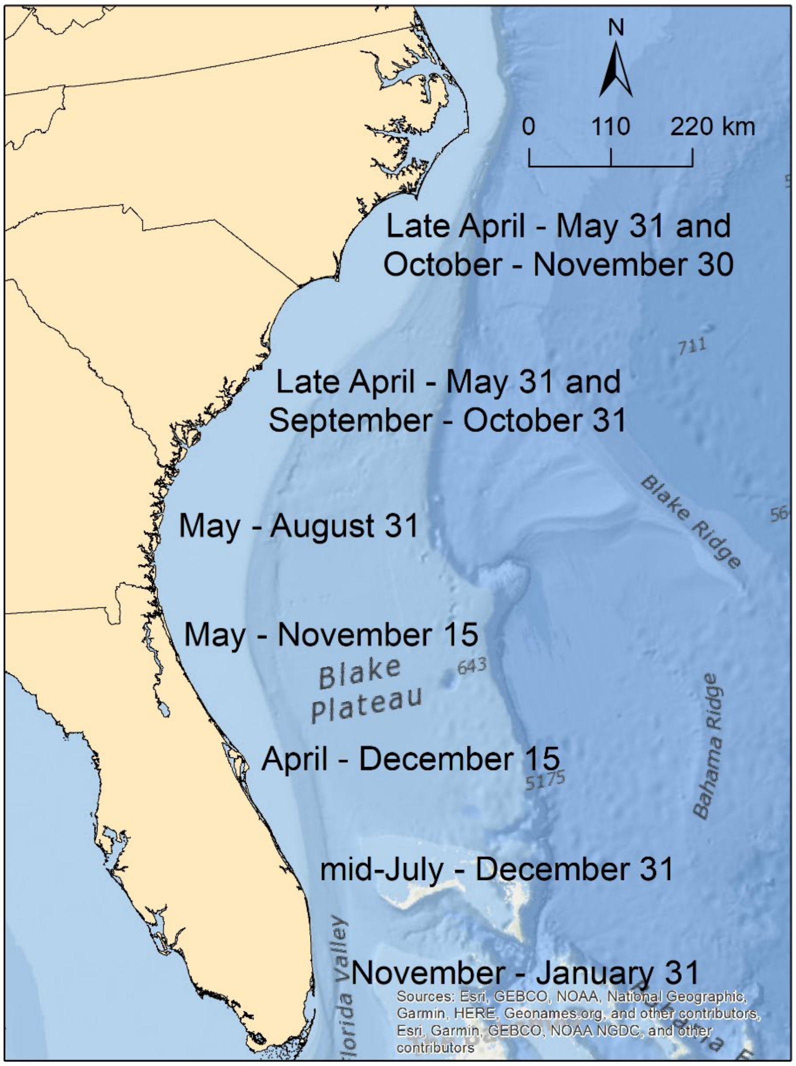


Figure 3-11. King mackerel seasonality with dates matching to approximate locations depicted by Trent et al. (1987).

3.3 Greater Atlantic Fish Habitat Associations and Seasonality

3.3.1 Introduction to the Greater Atlantic Physical Setting and Fish

The Greater Atlantic region can be described in three major divisions: the Mid-Atlantic Bight (Cape Hatteras, North Carolina–New Jersey), Southern New England (New York–southern Massachusetts), and the Gulf of Maine (north Massachusetts–Maine) (Greene et al. 2010). Each division differs in their major features, oceanographic characteristics, and geological origins. As described by Greene et al. (2010) these features include:

1) Mid-Atlantic Bight: The Chesapeake Bay and Delaware Bay provide extensive estuarine habitats. Warm water intrusions from the Gulf Stream move across the shelf and creates upwelling that enhances productivity. A series of barrier island inlets also provide spawning and estuarine habitats.
2) Southern New England: Rivers are an important aspect of this geography because of their historical support of large populations of American shad, Atlantic salmon, and eel. Shallow estuaries and sounds are important as well as the shellfish industry.
3) Gulf of Maine: One of the most productive marine ecosystems on Earth. These waters are near both the Labrador Current to the north and the Gulf Stream to the south. When these currents meet, ideal conditions are met for productivity of phytoplankton.

In terms of temporal variability, the North Atlantic Oscillation Index and the Atlantic Mulidecadal Oscillation both drive SST dynamics at annual and decadal timeframes (Ecosystem Assessment Program 2009). Climate change is affecting the present and future of fish in the Greater Atlantic. The Greater Atlantic is projected to undergo the most dramatic shifts in fish species compared to the South Atlantic and GoM (Morley et al. 2018). Kleisner et al. (2017) showed fall SSTs in the Greater Atlantic have increased since 1968, and water temperatures are projected to keep increasing. An analysis of select demersal and pelagic fish shows the thermal shift will result in species moving northward or to deeper waters, as some species lose habitat and the range of other species expands (Kleisner et al. 2017). For example, black sea bass are projected to lose habitat, while summer flounder are projected to gain habitat. Northern species, such as Atlantic cod, haddock, American plaice, and thorny skate are projected to lose habitat (Kleisner et al. 2017). The center of biomass for black sea bass and scup have also moved northward due to temperature changes (Bell et al. 2014). The effects of a warming ocean are important in the Greater Atlantic, but we do emphasize that our review is focused on species-habitat associations rather than any evaluation of future change.

In terms of sand resources, the Greater Atlantic region has cape-associated and sorted bedform shoals. Research has focused on various aspects of sand shoals in the Mid-Atlantic (Diaz et al. 2004; Maa et al. 2004; Slacum Jr et al. 2010a). New England sand shoals have been recognized in the literature (Smith 1969; Twichell 1983), but these shoals are rarely explicit in fisheries research. Sediments in shallow waters of the Greater Atlantic are diverse, ranging from silt to sand and gravel (Greene et al. 2010). Hard bottoms have a patchy distribution throughout much of the Greater Atlantic but are common in shallow waters of the Gulf of Maine (unpublished data, Matthew Poti, NOAA NCCOS).

The importance of prey species is highly varied in the Greater Atlantic. Bluefin tuna have been identified as feeding on species such as silver hake, Atlantic herring, and sand lance (Estrada et al. 2005). Phytoplankton, zooplankton, and benthic invertebrates have been proposed as important prey for fisheries (Ecosystem Assessment Program 2009). Bowman et al (2000) assessed stomach contents of 180 species and found common prey were sand lance, hakes, herrings, mackerels, butterfish, anchovies, scup, flatfishes, sculpins, longfin inshore squid, and northern shortfin squid.

The National Marine Fisheries Service (2017) provides a fisheries summary for the New England (Connecticut–Maine) and the Mid-Atlantic (Virginia–New York), and we provide a synopsis of the report here. For New England commercial fisheries, key federally managed fish are Atlantic herring, Atlantic mackerel, cod, haddock, various flounders, and monkfish (also referred to as goosefish). Although our focus is not on invertebrates, key federally managed species are quahog clam, sea scallop, and squid. For New England recreational fisheries, federally managed species of socioeconomic importance are Atlantic cod, Atlantic mackerel, bluefin tuna, bluefish, little tunny, scup, and summer flounder. Concerning Mid-Atlantic commercial fisheries, the key federally managed fish is summer flounder and key invertebrates are Atlantic surfclam, quahog clam, sea scallop, and squid. Concerning Mid-Atlantic recreational fisheries, key federally managed fish are black sea bass, bluefish, scup, summer flounder, and winter flounder.

The overlap of EFH with our Greater Atlantic study area is summarized in **Table 3-5**. As of 2018, overfishing is occurring for summer flounder in the Mid-Atlantic (NOAA Fisheries 2019). The following stocks, or stock complexes, have the status of being overfished or have overfishing occurring in New England (NOAA Fisheries 2019):

- Atlantic cod – Georges Bank
- Atlantic cod – Gulf of Maine
- Windowpane – Gulf of Maine/ Georges Bank
- Witch flounder
- Yellowtail flounder – Cape Cod/Gulf of Maine
- Yellowtail flounder – Georges Bank
- Yellowtail flounder – Southern New England/Mid-Atlantic
- Thorny skate – Gulf of Maine
- Atlantic halibut
- Atlantic salmon
- Atlantic wolfish
- Ocean pout
- Winter flounder – Southern New England
- Red hake – Southern Georges Bank/Mid-Atlantic
- Atlantic mackerel – Gulf of Maine/ Cape Hatteras

Given that the Greater Atlantic has 38 federally managed species and each have their individually designated EFH, we selected a subset of these to review the literature on habitat associations. We selected species based on the amount of overlap between a species' EFH designation and our study area, plus the commercial economic value of species. Of the highest ranked species, we selected:

- Atlantic herring
- Summer flounder
- Monkfish
- Black sea bass
- Atlantic surfclam
- Longfin inshore squid

3.3.2 Review of Select Finfish Species in the Greater Atlantic

3.3.2.1 Atlantic Herring

Pelagic fish are not well sampled with typical bottom trawl surveys (Wang et al. 2018); therefore, habitat relationships of these fish are poorly known. Wang et al. (2018) used fishery-dependent data to investigate Atlantic herring in the Greater Atlantic. Their presence-only modeling produced monthly distribution models, and showed the following habitat relationships with Atlantic herring:

- The effect of environmental variables on Atlantic herring differed by month.
- SST of the current and previous month had the strongest influence overall. In particular, these variables accounted for 78% of the model explanatory power in spring (March–May).
- Chlorophyll-*a* concentration most influenced Atlantic herring in June–August, and chlorophyll-*a* had a moderate influence September–November.
- Bathymetry accounted for a moderate level of explanatory power overall. Bathymetry was most influential in the winter and spring when the species was in deeper waters.
- Geomorphology habitat variables were not tested.
- Offshore of New Jersey, Palamara et al. (2012) investigated Atlantic herring habitat with a multivariate approach. They found Atlantic herring (winter to spring seasons) were related to the combined environmental variables of more shallow depths, cooler SST anomalies, cooler bottom temperatures, coarse sediment grain size, and a remote sensing variable similar to chlorophyll.
- Atlantic herring spawn in the Gulf of Maine July–November; spawning then initiates progressively to the south (Sinclair and Tremblay 1984). The species tends to be located relatively close to the shoreline in the spring, farthest inshore in summer, and are found over a wide-ranging area in the fall/winter (Wang et al. 2018).

Table 3-5. For federally managed species within the Greater Atlantic, the proportion of area designated as EFH within the study area.

Fishery Management Council	Species or EFH species group	Proportion of Greater Atlantic study area designated as EFH	Fishbase description of habitat
Mid-Atlantic	Atlantic butterfish	**0.87**	benthopelagic 15–420 m usually ≤55 m
Mid-Atlantic	Atlantic mackerel	**0.87**	pelagic 0–1,000 m usually 0–200 m
Mid-Atlantic	Atlantic surfclam*	**0.64**	NA
Mid-Atlantic	Black sea bass*	**0.80**	reef-associated 1 m–unknown
Mid-Atlantic	Bluefish	**0.82**	pelagic 0–200 m
Mid-Atlantic	Golden tilefish	0.00	demersal 80–540 m
Mid-Atlantic	Longfin inshore squid*	**0.79**	NA
Mid-Atlantic	Northern shortfin squid	0.24	NA
Mid-Atlantic	Ocean quahog	0.42	NA
Mid-Atlantic	Scup	**0.89**	demersal 15 m–unknown
Mid-Atlantic	Spiny dogfish	**0.87**	benthopelagic 0–1,460 m usually 50–300 m

Fishery Management Council	Species or EFH species group	Proportion of Greater Atlantic study area designated as EFH	Fishbase description of habitat
Mid-Atlantic	Summer flounder*	**0.92**	demersal 10–183 m usually ≤37 m
New England	Acadian redfish	0.02	pelagic 100–1,000 m usually 100–500 m
New England	American plaice	0.05	demersal 10–3,000 m usually 90–250 m
New England	Atlantic cod	**0.54**	benthopelagic 0–600 m usually 150–200 m
New England	Atlantic halibut	0.00	demersal 50–2,000 m
New England	Atlantic herring*	**0.86**	benthopelagic 0–364 usually 0–200 m
New England	Atlantic wolffish	0.11	demersal 1–600 m 18–110 m
New England	Barndoor skate	0.03	demersal 0–750 m usually 0–150 m
New England	Clearnose skate	0.42	demersal 0–330 m usually 0–50 m
New England	Haddock	0.30	demersal 10–450 m usually 10–200 m
New England	Little skate	**0.55**	demersal 0–329 m
New England	Monkfish*	**0.81**	bathydemersal 20–1,000 m
New England	Ocean pout	**0.34**	demersal 0–388 m
New England	Offshore hake	0.01	bathydemersal 80–1,170 m usually 160–640 m
New England	Pollock	0.20	demersal 37–364 m
New England	Red hake	**0.67**	demersal 35–1,152 m usually 110–130 m
New England	Rosette skate	0.00	reef-associated 55–530 m
New England	Sea scallop	**0.61**	NA
New England	Silver hake	0.35	demersal 55–914 m
New England	Smooth skate	0.00	bathydemersal 46–914 m
New England	Thorny skate	0.01	demersal 20–1,000 m usually 50–100 m
New England	White hake	0.16	demersal 100–1,000 m usually 100–247 m

Fishery Management Council	Species or EFH species group	Proportion of Greater Atlantic study area designated as EFH	Fishbase description of habitat
New England	Windowpane flounder	**0.89**	demersal 55–73 m
New England	Winter flounder	0.43	demersal 5–143 m
New England	Winter skate	**0.62**	demersal 0–120 m
New England	Witch flounder	**0.54**	demersal 18–1,570 m usually 45–366 m
New England	Yellowtail flounder	**0.67**	demersal 27–364 m usually 36–91 m

Notes: * = Literature reviews were conducted on these species
Those overlapping with > 50% of the study area are bolded. Study area is defined in **Figure 1-4**. Includes all life stages.

3.3.2.2 Summer Flounder

In a Mid-Atlantic Bight study, Manderson et al. (2011) investigated relationships of summer flounder with a plethora of habitat characteristics. In comparison to other species, they found summer flounder were more responsive to benthic characteristics and prey abundance. Manderson et al. (2011) found the following habitat associations:

- Peak summer flounder abundance with a bottom temperature of 7–12.5°C. Summer flounder were rare in waters of < 6°C and temperature selection did not change with season.
- Highest abundance with depths of < 150 m (winter–spring).
- A positive relationship with standard deviation of depth (winter–spring); in our study, shoals have a high standard deviation of depth compared to flat substrates (also similar to rugosity measures).
- A positive relationship with a divergence index (i.e., vertical water velocity), which indicates potential upwelling. The authors suggest that upwelling and enhanced productivity near estuarine plumes entering the ocean may serve as valuable spawning sites.
- A positive relationship with Simpson's potential energy anomaly (strength of stratification) .
- High summer flounder abundance with a frontal index.
- A negative correlation with chlorophyll-*a* (winter–spring) and a positive correlation with chlorophyll-*a* in the fall.
- Summer flounder were positively correlated with a potential prey, squid, in the winter–spring, but a negative correlation was found in the fall.

Offshore of New Jersey, Palamara et al. (2012) investigated summer flounder habitat with a multivariate approach. They found summer flounder (winter to spring seasons) were related to the combined environmental variables of more shallow depths, cooler SST anomalies, cooler bottom temperatures, coarse sediment grain size, and a remote sensing variable similar to chlorophyll. In waters near Maryland, Slacum Jr et al. (2008) focused on substrate associations with summer flounder and found poor correlations. Summer flounder were often, but inconsistently, found in the 1–20 m depth range, which was often represented as the troughs of sand shoals (Slacum Jr et al. 2008).

Summer flounder spawning peaks in the fall for the Mid-Atlantic Bight waters, and spawning ranges from September–January 31 (Able et al. 1990). Able et al. (1990) found larvae abundance peaked in November, and Smith (1973) reported spawning waters were 12–19°C. Migration corridors between estuaries and the continental shelf may be important. Recent analyses have shown summer flounder juveniles (year 2 and 3) emigrate to the shelf in the fall and return to the estuary in the spring (Sackett et al. 2007). Even young-of-the-year summer flounder migrate out of estuaries in the late fall to the shallow continental shelf (Able et al. 1990).

3.3.2.3 Monkfish

The monkfish, also referred to as goosefish, is a benthic species of the Greater Atlantic (Armstrong et al. 1992). Monkfish ecology has been poorly studied (Richards et al. 2008), but some insights can be gained from known prey species, depth, and water temperatures used by the species. Teleost fish are an important prey across all monkfish size classes, but smaller size classes frequently prey on crustaceans (Armstrong et al. 1996). Sand lance, long-finned inshore squid, red hake, and little skate (for the largest monkfish) were the primary fish species fed on across all size classes (Armstrong et al. 1996). Richards et al. (2008) provides a detailed analysis of monkfish in relation to depths and seasonal distributions. They found the following:

- Across all seasons, 90% of monkfish in trawl surveys were in depths of 32–339 m (surveyed waters included 24–346 m depths).
- Winter and spring monkfish were in relatively deeper water; in the summer, they were equally widespread across depths.
- Fall distribution of monkfish showed no depth preference in the North Atlantic, but were in relatively deeper water in the Mid-Atlantic Bight.
- Compared to depths, monkfish were more selective of bottom temperatures. Across all seasons, 90% of monkfish were found with bottom temperatures of 4.5–13.0°C (surveyed waters included 3.8–19.3°C). This resulted in their distribution within relatively warmer waters for winter/spring and relatively cooler waters in summer/fall.

3.3.2.4 Black Sea Bass

Black sea bass is a demersal, temperate reef-associated species that associates with hard bottom structures such as rock outcroppings, reefs, and artificial reefs (Fabrizio et al. 2013). The black sea bass north of Cape Hatteras, North Carolina, is a separate stock from the other populations in the Atlantic (McCartney et al. 2013; Roy et al. 2012). The species overwinters in waters of the middle- to outer continental shelf and then migrates inshore as temperatures increase in the spring (Moser and Shepherd 2009; Musick and Mercer 1977). In the Mid-Atlantic, black sea bass spawn April–October ranging from depths of 18–45 m (Musick and Mercer 1977), while the southern New England population spawns May–June (Steimle 1999). The Greater Atlantic black sea bass population has been shown to be moving northward in response to warming water temperatures (Bell et al. 2014), and their range is projected to contract in the future (Kleisner et al. 2017).

Miller et al. (2016a) studied black sea bass in relation to oceanographic variables in the Mid-Atlantic Bight and southern New England. Juveniles and adults had similar habitat characteristics with the following results:

- Juveniles and adults together selected salinities ranging 32.6–35 psu.
- A positive relationship was found with temperature and juveniles.
- Adults black sea bass selected bottom temperatures of > 7.9°C.

Fabrizio et al. (2013) studied black sea bass near a dredge disposal site. They found black sea bass started to move away from an inshore site beginning in June. By early November, 74% of individuals had presumably moved to offshore locations; this generally coincided with the time when shallow, inshore waters were becoming cooler than offshore waters (Fabrizio et al. 2013). Habitat use of black sea bass showed a selection for coarse substrate material, depth, and variance of the slope (i.e., substrate complexity) (Fabrizio et al. 2013). However, they found that black sea bass did not select for substrates, or substrate complexity, when waters were > 27.5 m in depth.

3.3.2.5 Atlantic Surfclam

The Atlantic surfclam is a filter-feeding bivalve that is common in sandy sediments of the northeast Atlantic (Weinberg 2005), although most of its habitat associations are related to oceanographic conditions. Changes in the distribution of Atlantic surfclam have been attributed to climate change and the associated increase in bottom temperatures (Weinberg 2005; Weinberg et al. 2002). Atlantic surfclam are typically in waters < 50 in depth (Weinberg and Helser 1996). A study in the southern portion of their range (offshore of Virginia) showed the species declining, but the decline was lessened in deeper waters (Weinberg 2005). The authors suggest this is because warming was more severe in shallow waters. Likewise, a physiology-based simulation study for the Mid-Atlantic Bight showed that years of above average water conditions (+2°C) would result in starvation mortality and declines in the population (Narváez et al. 2015). Overall, the southern portion of the Atlantic surfclam range is likely the most at-risk. However, a study of Georges Bank (offshore Massachusetts) also shows a movement into deeper waters (Powell et al. 2017).

In a study encompassing a shoal complex (Beach Haven Ridge) offshore of New Jersey, Savage et al. (1976) found Atlantic surfclam had a much greater abundance in troughs compared to shoal crests. We do note that substantial numbers were still found on crests (Savage 1976). In an experimental study of larvae settlement, Snelgrove et al. (1998) found larvae selected sand substrates over mud when water flow was present. In still water experiments, Atlantic surfclam larvae had inconsistent selection, possibly because their movement was limited (Snelgrove et al. 1998).

3.3.2.6 Longfin Inshore Squid

Longfin inshore squid (hereby, "squid") migrate between lower latitudes or farther offshore wintering habitats to higher latitudes and inshore waters during summer (Manderson et al. 2011). A detailed assessment of these shifts in abundance was described by Hatfield and Cadrin (2002), who provides detailed information on squid frequency by depth zones and broad latitudinal zones. In a comprehensive study of the Mid-Atlantic Bight, Manderson et al. (2011) quantified habitat relationships of longfin inshore squid. They found squid distribution was best explained by remote sensing oceanographic data followed by *in situ* oceanographic variables and then benthic habitat data (Manderson et al. 2011). The habitat variables that explained > 5% of the null deviance are given below with a qualitative interpretation of the relationship (1 being the most important variable, 5 the least important):

1) Squid were most abundant with bottom temperatures > 7°C (fall–spring).
2) Squid were most abundant at ≤ 200 m depths (winter–spring).
3) A positive relationship with standard deviation of depth was found (winter–spring). In our study, shoals have a high standard deviation of depth compared to flat substrates.
4) Squid were related to water mass classifications (unsupervised classification characterized by moderate temperature, salinity, and primary productivity over intermediate depths).
5) Squid were negatively related to Simpson's potential energy anomaly (strength of stratification) in the fall.

3.4 Shark Habitat Associations

3.4.1 Introduction to Sharks

Key Points ***(Gaps are in italics)***

- Shark species can be categorized into demersal, pelagic, benthopelagic, and reef-associated, as well as coastal or oceanic pelagic.
- Commonly derived habitat associations of sharks include depth, water temperature, chlorophyll, dissolved oxygen, and salinity. *Although infrequently* tested, sharks are also associated with SST gradients, fronts, and eddies. *In turn, these oceanic features are likely related to prey, but this connection has been poorly studied.*
- Sharks may migrate vast distances but use relatively localized areas, where they either stopover briefly during migration or reside for weeks to months at a time. *In case studies, migratory and wintering areas are used consistently, but our knowledge of such areas among shark species is sparse*.
- Sharks may interact with sand shoals because of their narrow migration paths along nearshore waters.
- Waters near Cape Hatteras, North Carolina, and Cape Canaveral, Florida, are particularly important nearshore areas for a variety of shark species. These locations offer the northernmost waters where temperate and subtropical temperatures, respectively, are consistently maintained.

As predators, sharks are vulnerable to overfishing of prey species, mortality as part of fisheries bycatch, and direct harvests (Musick et al. 2000). Shark populations are particularly vulnerable because of their natural history characteristics of having a late age to maturity and low fecundity (Musick et al. 2000). In a study including the Atlantic Ocean of North America, GoM, and Caribbean Sea, Baum et al. (2003) found that all coastal and oceanic shark species recorded in longline fisheries (except makos) had decline by > 50% in the last 8 to 15 years. For commercial fisheries, the blacktip shark and sandbar shark are the most valuable (Castro 1996), although the sandbar shark fishery is now closed. Other shark species are harvested, but the value of individual species are not well documented.

Generally, sharks can be divided into oceanic and coastal species. The coastal species have a relatively high proportion of their EFH designations in our study areas (waters ≤ 50 m), while more oceanic species have a lower proportion of their distribution in the study areas (**Table 3-6**). Oceanic species also tend to be larger. Furthermore, sharks are classified into those that are reef-associated, pelagic, demersal, or benthopelagic (**Table 3-6**). We note that the Fishbase classifications of "reef-associated" is particularly broad; species like blacknose shark, spinner shark, and blacktip shark are not strongly associated with reefs in the Atlantic. Benthopelagic species are defined as those that live and feed near the bottom, in midwaters, or near the surface (Froese and Pauly 2018). Although our focus is on shark-habitat relationships, we note that density dependence can regulate shark populations; both positive and negative interactions occur among shark species (Peterson et al. 2017). Importantly, recent studies have shown that sharks are not simply transient in nature and often reside in specific waters for several weeks or months at a time (Conrath and Musick 2008; Haulsee et al. 2018; Reyier et al. 2014); this has even been demonstrated for highly pelagic species (Lea et al. 2015).

Table 3-6. For federally managed shark species, the proportion of area designated as EFH within each study area.

Species	Proportion of GoM study area designated as EFH	Proportion of South Atlantic study area designated as EFH	Proportion of Greater Atlantic study area designated as EFH	Fishbase description of habitat
Angel shark	0.03	0.11	0.48	Bathydemersal 1–1,375 m usually ≤128 m
Atlantic sharpnose shark	**1.00**	**0.97**	0.24	Demersal 0–280 m usually 0–10 m
Basking shark*	0.00	0.05	0.49	Pelagic 0–2,000m
Bignose shark	0.02	0.17	0.16	Reef-associated 12-810 m usually 80–220 m
Bigeye thresher	0.00	0.00	0.00	Pelagic-oceanic 0–730 m
Blacknose shark	**0.76**	**0.66**	0.00	Reef-asscoiated 9–64 m usually ≥ 9 m
Blacktip shark	**0.58**	**0.61**	0.04	Reef-associated 0–100 m usually 0–30 m
Blue shark	0.00	0.07	**0.55**	Pelagic 1–1,000 m usually 1–220 m
Bonnethead shark	0.15	0.27	0.00	Reef-associated 10–80 m usually 10–25 m
Bull shark	**0.74**	0.32	0.00	Reef-associated 1–152 m usually 1–30 m
Caribbean reef shark	0.03	0.04	0.00	Reef-associated 1–65 m usually 1–35 m
Common Thresher shark	0.00	0.35	**0.82**	Pelagic 0–650 m usually 0–200 m
Dusky shark	0.04	**0.75**	**0.88**	Reef-associated 0–400 m usually 200–400 m
Finetooth shark	0.23	0.33	0.00	Demersal ≤10 m
Great hammerhead shark	**0.72**	**0.72**	0.24	Pelagic 1–300 m usually 1–100 m
Lemon shark	0.48	0.23	0.00	Reef-associated 0–92 m
Longfin mako	0.00	0.00	0.00	NA
Night shark	0.00	0.00	0.00	benthopelagic 0–600 m 50–100 m
Nurse shark	0.45	0.26	0.00	reef-associated 0–130 m 1–35 m
Oceanic whitetip shark	0.00	0.00	0.00	pelagic 0–230 m usually 0–152 m

Species	**Proportion of GoM study area designated as EFH**	**Proportion of South Atlantic study area designated as EFH**	**Proportion of Greater Atlantic study area designated as EFH**	**Fishbase description of habitat**
Porbeagle shark	0.00	0.01	0.10	pelagic 0–715 m
Sandbar shark	0.43	**0.93**	**0.86**	benthopelagic 0–500 m usually 20–65 m
Sand tiger shark	0.00	0.49	**0.55**	reef-associated 1–191 m usually 15–25 m
Scalloped hammerhead shark	**0.64**	**0.93**	**0.71**	pelagic 0–1,000 m usually 0–25 m
Shortfin mako shark	0.00	0.12	0.81	pelagic 0–750 m usually 100–150 m
Silky shark	0.21	**0.66**	0.16	reef-associated 0–4,000 m usually 0–500 m
Spinner shark	**0.71**	**0.70**	0.18	reef-associated 0–100 m
Tiger shark	0.49	**0.97**	**0.82**	benthopelagic 0–800 m usually 0–140 m
Whale shark	0.04	0.00	0.00	pelagic 0–1,928 usually 0–100 m
White shark	0.16	0.44	**0.68**	pelagic 0–1,200 m usually 0–250 m

Notes: * The GoM and South Atlantic both extend through the Florida Keys. The Greater Atlantic extends from Maine to Cape Hatteras, North Carolina.
Those overlapping with > 50% of the study area are bolded. Study areas are defined in **Figures 1-2, 1-3, and 1-4**. All life stages were included.

Because our focus is on offshore marine habitats, we do not directly address the rivers, estuaries, and bays that commonly act as nursey habitat for juvenile sharks (Heupel et al. 2007). For further information on these topics, resources are available for the GoM and Atlantic coasts (e.g., Curtis et al. 2013; Froeschke et al. 2010; Heupel et al. 2007; McCallister et al. 2013; Ulrich et al. 2007).

3.4.2 GoM Sharks

Of the 30 highly migratory Atlantic shark species with designated EFH, seven species had EFH overlapping with > 50% of our GoM study area (**Table 3-6**). These species were the Atlantic sharpnose, blacknose, blacktip, bull, great hammerhead, scalloped hammerhead, and spinner sharks. These species commonly overlapped with sand dredging locations offshore of Louisiana and Florida. Additionally, lemon, nurse, sandbar, and tiger sharks had EFH overlapping with 40–50% of the study area, and EFH overlap for them was primarily on the western shelf of Florida**.** Eight shark species' EFH designations had no overlap with the GoM study area, and we did not consider these species further for the GoM. Below, we synthesize species-habitat relationships of relevant shark species.

Drymon et al. (2013) tested the relationships of shark abundance (CPUE) with depth, water temperature, chlorophyll, dissolved oxygen, salinity, crustacean biomass, and fish biomass over the entire nGoM. They found blacknose shark distribution was best explained by temperature, although temperature was negatively correlated with depth; this generally supports a depth preference of 10–30 m for blacknose shark (Drymon et al. 2013; Drymon et al. 2010). Atlantic sharpnose shark distribution was best correlated with chorlophyll concentration, and blacktip shark distribution was best explained by crustacean biomass (Drymon et al. 2013). However, blacktip sharks prey primarily on fish, so the relationship was likely a proxy for another environmental or biotic factor (Drymon et al. 2013). They also found Atlantic sharpnose and spinner sharks were negatively related to dissolved oxygen for unknown reasons. Spinner shark were correlated with a combination of depth and chlorophyll. Bull sharks were not well correlated with the variables tested.

Drymon et al. (2010) examined the relationship between shark species CPUE and water depth (**Table 3-7**). Additional findings included:

- Atlantic sharpnose sharks were more abundant at 10–29.9 m depths compared to depths of > 30 m.
- Atlantic sharpnose females were disproportionately more abundant than males in waters > 30 m depth, whereas males were more common in waters of 0–9.9 m depths. Likewise, Parsons and Hoffmayer (2005) captured 718 males and only 9 female sharpnose sharks in shallow waters, presumably because females do not use shallow waters for mating or pupping.
- Blacknose shark males were disproportionately more abundant than females in waters of 0–29.9 m depths.
- Blacktip sharks were more abundant at 0–9.9 m depths compared to waters of > 30 m depths. These sharks were biased toward mature females at 0–29.9 m compared to depths of > 30 m.

Data for other species were not robust enough for statistical analyses, but the authors noted that scalloped hammerhead, sandbar, and silky sharks were rarely encountered at shallow or mid depths but were found in waters with depths of > 30 m. Finetooth sharks were mostly observed at shallow depths. Of species caught in nearshore surveys (in the vicinity of classified sand shoals), 10 of 12 species occurred during the spring, summer, and fall. Bonnethead and nurse sharks were absent during the spring months. Peak periods for Atlantic sharpnose, finetooth, and spinner sharks occurred during the fall (Drymon et al. 2010).

Table 3-7. Reported CPUE (sharks hooks^{-100} h^{-1}) across depth categories (range 2–366 m) for sharks on GoM surveys 1995–2008.

Species	Depth: 0–9.9 m	Depth: 10–29.9 m	Depth: > 30 m
Atlantic sharpnose shark	2.01	2.82	5.22
Bignose shark	0	0	0.01
Blacknose shark	0.77	2.11	0.51
Blacktip shark	1.36	1.01	0.53
Bonnethead shark	0.01	0.01	0.01
Bull shark	0.33	0.15	0.08
Dusky shark	0	0	0.01
Finetooth shark	0.24	0.02	0
Great hammerhead	0.04	0.08	0
Night shark	0	0	0.01
Nurse shark	0.01	0.07	0.02
Sandbar shark	0.01	0.04	0.22
Scalloped hammerhead	0.02	0.06	0.22
Shortfin mako	0	0	0.01
Silky shark	0	0	0.11
Spinner shark	0.31	0.36	0.23
Tiger shark	0.02	0.1	0.06

Only sharks with designated EFH are shown. Source: Modified based on information from Drymon et al. (2010).

Gruss et al. (2018) grouped all large coastal sharks together (sandbar, blacktip, silky, tiger, bull, spinner, lemon, nurse, smooth hammerhead, scalloped hammerhead, and great hammerhead) and found the following habitat relationships for nGoM waters offshore of Florida:

- Peak probability of presence occurred at depths of 75–150 m.
- Probability of large shark occurrence decreased as distance from shoreline increased.
- Peak probability of occurrence occurred at 15–22°C.
- Bottom salinity differences of < 2 psu were identified as having an extremely large effect on shark distribution for unknown reasons.

In the same study, an analysis of small coastal sharks as a group (Atlantic sharpnose, blacknose, finetooth, and bonnethead) showed the following relationships (Grüss et al. 2018):

- High probability of presence at ≤ 50 m of depth
- High probability of presence with lower chlorophyll concentration
- High probability of presence with surface salinity > 35 psu

Wells et al. (2018) tracked scalloped hammerheads throughout the GoM, and they selected for relatively high salinity waters (~35 psu), depths of < 1,500 m, less chlorophyll concentration, relatively high sea surface height anomalies, and for waters within 20 km of artificial reef or natural hard bottom. In their study, artificial reefs included oil and gas platforms, and the substrate variables had a much stronger influence than the oceanographic variables (e.g., SST was not a factor). Movements of scalloped hammerheads showed discrete areas were used for foraging and only a few movements were purely transitory (Wells et al. 2018). In their study, the 50% kernel density home ranges described overlap with Ship Shoal, St. Bernard Shoals, and other shoals near Mississippi and Alabama.

Dusky sharks captured and tagged near the Mississippi River Delta moved 9–31 km per day but did not appear to track chlorophyll concentrations (Hoffmayer et al. 2014). The depth of dusky shark individuals within the water column ranged from 0–573 m, with high frequency of use for 20–50 m depths in the water column. The maps generally showed dusky sharks using waters > 50 m in total depth, although those measures were not reported. Temperatures of the water column utilized by dusky sharks ranged from 9–32°C, with 24–26°C being the most selected (Hoffmayer et al. 2014).

Subadult bull sharks offshore of Louisiana and Florida were initially captured in water depths ranging 5–96 m with a mean depth of 12 ± 14 m (Carlson et al. 2010). Some sharks remained in the general area of capture, while others migrated long distances (range of daily movements of individuals: 0.1–27 km). Water temperature used by bull sharks ranged from 16 to > 32°C, with a peak at 26–33°C. Water depths of bull shark ranged from 2 to > 50 m. Of 15 bull sharks analyzed, five spent all of their time at < 20 m depths of the water column, although specific depths of the location were not reported.

3.4.3 South Atlantic Sharks

Of the 30 shark species with designated EFH, 10 species had EFH overlapping with > 50% of the South Atlantic study area (**Table 3-6**). These species were Atlantic sharpnose, blacknose, blacktip, dusky, great hammerhead, sandbar, scalloped hammerhead, silky, spinner, and tiger shark. They were strongly associated with South Atlantic waters, where sand dredging has occurred. Notably, finetooth shark EFH overlapped by only 33% of the study area, but the overlap occurred in locations with past dredging events. Bull, common thresher, sand tiger, and white sharks had EFH overlapping with 30–50% of the study area, and these areas commonly overlapped with past sand dredging events. Of the remaining shark species, nurse and lemon sharks are geographically limited to Florida, but overlap strongly with the study area there. Because of the highly migratory nature of sharks moving between the Greater Atlantic and South Atlantic regions, the section below on Greater Atlantic sharks covers migratory movement and wintering studies that overlap both the South Atlantic and Greater Atlantic regions.

Studies conducted in the shallow waters offshore of North Carolina (Thorpe et al. 2004) and South Carolina (Ulrich et al. 2007) investigated waters of 3–15 m depths and found the following federally managed shark species to be common:

- Atlantic sharpnose
- Blacknose
- Blacktip
- Bonnethead
- Finetooth
- Sandbar
- Spinner

In a study of tiger, blue, shortfin mako, and great hammerhead sharks, Queiroz et al. (2016) found predictable hotspots of high habitat use based on a coarse selection for SST and a fine-scale selection for strong SST/productivity gradients (i.e., fronts). Blue and shortfin mako sharks were wide-ranging along the North American, Atlantic Coast in spring/summer, but were primarily found in waters near Florida and deep offshore waters of the central Atlantic during the fall and winter. Hammerhead sharks remained on the continental shelf year-round, and tiger sharks used the Gulf Stream during the warmer months. Specific aggregations of multiple species highlighted use of the Gulf Stream and the North Atlantic Current/Labrador Current convergence zone; the low productivity Sargasso Sea was generally absent of locations.

At hard bottom survey sites, Bacheler et al. (2016) observed the following depth associations:

- Atlantic sharpnose shark had no association with depth and were present at 10–60 m depths.
- Tiger shark showed no association with depth and were present at 10–79 m depths.
- Sandbar shark had no association with depth and spanned 20–69 m depths.
- Nurse shark were most common at depths of 10–19 m and declined in occurrence in deeper water with a maximum depth of 59 m.

Cape Canaveral waters are considered a winter nursery area for juvenile lemon sharks (Reyier et al. 2008; Reyier et al. 2014). These sharks aggregate with densities as high as 21.8 sharks km^{-1} of survey in the waters near Cape Canaveral and its associated shoals (Reyier et al. 2008). Monitoring the movements of 54 juvenile lemon sharks with a telemetry array, Reyier et al. (2014) observed the following:

- Strong site fidelity of individuals at Cape Canaveral spanning November 23–February 28 with some movements to another nearby aggregation.
- Juvenile lemon sharks inhabited waters ranging 12–30°C, but spent >70% of their time at 15–20°C.
- Greater abundance was observed with shorter day lengths and cooler water temperatures, which suggests Cape Canaveral is a warm water refuge in the region. During one cold spell when water temperature averaged < 16°C, individuals temporarily moved southward 62–191 km.
- Sand shoals were observed to deflect south-flowing nearshore currents to the east, which allowed warmer north-flowing currents to flow to the nearshore. Water temperatures were 2–3°C warmer because of this phenomenon.
- During late February through April, juvenile lemon sharks migrated northward to waters near Georgia and Charleston Harbor, South Carolina.
- In addition to juveniles, Cape Canaveral monitoring has observed 60 tagged adult lemon sharks migrating through in spring and several adults that remained into the summer.

Overall, sand shoals are thought to be either a refuge from predators or a productive feeding ground for sharks in the Cape Canaveral region (Reyier et al. 2008; Reyier et al. 2014). Observations of juvenile lemon sharks offshore of Brazil showed foraging by "substrate inspection" near rocky and reef bottoms, whereas adults foraged for sardines (Garla et al. 2017). Such behavior may explain the association with Cape Canaveral's shoals. In addition to lemon sharks, the nearshore shoal habitats surrounding Cape Canaveral appear to be nursery areas for scalloped hammerhead (Adams and Paperno 2007). For scalloped hammerheads, Adams and Paperno (2007) found neonates in water depths of 3.8–9.7 m from late May through June. From February to June, they also captured juvenile nurse sharks, juvenile blacktip sharks, neonate/juvenile/adult Atlantic sharpnose, and juvenile/adult bonnetheads. The authors suggest that the shallow waters may protect these sharks from large predators (Adams and Paperno 2007).

Sandbar sharks are common from Long Island to West Palm Beach, Florida, during the summer and range from the Carolinas to the southern tip of Florida during winter (Springer 1960). Conrath and Musick (2008) tracked sharks offshore of eastern Virginia during the summer, and all seven individuals wintered offshore of North Carolina. The authors suggest that central North Carolina could be important waters for wintering sharks because of its proximity to the warm Gulf Stream. Conrath and Musick (2008) reported the following habitat associations:

- During the summer, 80% of sandbar shark locations in the water column were < 12 m in depth (range of 0–24 m), but winter observations ranged 0–172 m. Total depth of the water column was not reported.

- Water temperatures that sharks experienced ranged from 10–26°C during winter with peak use at 18–22°C.
- During summer, sharks tended to be in waters of 20–28°C.

3.4.4 Greater Atlantic Sharks

Of the 30 shark species with designated EFH, nine species had EFH overlapping with > 50% of our Greater Atlantic study area (**Table 3-6**). These species were blue, common thresher, dusky, sandbar, sand tiger, scalloped hammerhead, shortfin mako, tiger, and white sharks. Because current sand dredging is limited to the Mid-Atlantic, blue shark EFH does not overlap with current sand dredging leases. In addition, angel, Atlantic sharpnose, great hammerhead, and spinner shark EFH did overlap with past sand and gravel leases in the Mid-Atlantic.

Sand tiger sharks (hereby, sand tigers) near Fenwick Island, Delaware, and its shoals, selected waters that were closer to shore (range 0–20 km), less saline (< 32 psu), and higher in dissolved organic matter compared to other areas (Haulsee et al. 2015). Water temperature, DO, and chlorophyll were not a predictor of habitat use in their study. In addition, sand tigers were found to migrate northward along a narrow, nearshore band of shallow water in the spring (Haulsee et al. 2015). Haulsee et al. (2018) further suggest that sand tigers may use the shoreline and shoals as landmarks for their migration in the fall. In the same region, adult and juvenile sand tiger sharks were related to depth, day of year, and raw reflectance from satellite imagery (related to chlorophyll). Habitat selection of sand tiger sharks showed peak use at 10–23 m in depth in this study.

The seasonality of sand tiger shark distribution provides a case study for other sharks that remain unstudied in the Mid-Atlantic. Juvenile sand tigers are known to use waters extending from Cape Hatteras, North Carolina, northward to New England from June to October 1, and then can be found only south of Cape Hatteras to central Florida December to April (Kneebone et al. 2014). Kneebone et al. (2014) showed juvenile sand tiger sharks used water temperatures of 9.8–26.9°C, with the most common temperatures being 12–20°C. They also found juveniles used depths of 0–80 m and were frequently found at depths < 35 m. Similar to juvenile migration patterns, adult sand tigers often arrive offshore of Delaware Bay by May, and they migrate southward by mid-October (Haulsee et al. 2018). Importantly, Kneebone et al. (2014) concluded the following:

- Twenty-seven sand tigers (43% of individuals studied) were detected briefly near Cape Hatteras, North Carolina. This is likely a consistent part of their migration route.
- During winter (December–April), approximately 20% of individuals had a period of residency in the vicinity of Cape Hatteras, North Carolina, and another 9% had residency in Florida, particularly near Cape Canaveral.
- Because of the locations of passive telemetry locators, sand tiger sharks were known to use waters < 10 m in depth along their migration paths. The lack of passive transmitters in deeper waters means a comparison could not be made.

Furthermore, a study in Delaware Bay showed all seven tagged male sand tigers migrated south to the vicinity of Cape Hatteras, North Carolina, whereas females moved outward to the edge of the continental shelf (Teter et al. 2015). As noted with sandbar sharks (Conrath and Musick 2008), this area is thought to be important to wintering sharks because of relatively high temperatures of the Gulf Stream. Five of the seven male sand tigers also spent 2–4 weeks at “rest-stops” offshore of the North Carolina/Virginia border (Teter et al. 2015). During migration, sand tigers used depths of 18–73 m with larger sharks using deeper waters (Teter et al. 2015).

Basking sharks, a planktivorous species, were captured and satellite tagged offshore of Cape Cod, Massachusetts, from June to October, and were found to migrate southward in late autumn (e.g., Sept 30);

individuals spent the winter in a range spanning from South Carolina to Brazil (Skomal et al. 2009). Eighty-one percent of the tracked sharks overwintered in tropical waters. Skomal et al. (2009) note that the North Atlantic waters undergo dramatic seasonal temperature fluctuations, whereas other studies of basking sharks in more stable environments have not shown such long-distance movements.

For white shark, Curtis et al. (2014) and Skomal et al. (2017) showed white shark spend winter (January–March) restricted to waters south of Cape Hatteras, North Carolina, and extending into the eastern GoM. Curtis et al. (2014) also found the following:

- During summer (July–September), white sharks were primarily observed north of Virginia.
- During spring and fall, white shark were spread throughout the Atlantic and eastern GoM.
- White sharks migrated northward when SST was > 14°C.
- Young-of-the year and neonates were most frequently observed between the central coast of New Jersey to Massachusetts Bay.

Skomal et al. (2017) tagged white sharks near Cape Cod, Massachusetts, and Jacksonville, Florida. They found juveniles and most subadults spent their time in coastal environments; some subadults and most mature adults spent at least some time much farther offshore. Although the total depth of the waters used were not reported, white sharks themselves spent almost all of their time < 50 m in depth and more than half of their time was ≤ 20 m deep the water column (Skomal et al. 2017). White sharks were observed in waters ranging from 4–28°C, indicating that movements are likely based on foraging or reproductive potential rather than temperature tolerance (Skomal et al. 2017). Near the Gulf Stream, mature white sharks have been reported as focusing on the interior of clockwise-rotating anticyclonic eddies characterized as warm temperature anomalies (Gaube et al. 2018).

Shortfin mako were tagged near Long Island, New York and the Yucatan Peninsula of Mexico for studies of horizontal and vertical movements (Vaudo et al. 2017; Vaudo et al. 2016). Shortfin mako were found in water temperatures ranging 5.2–31.1°C, and frequently spent time in waters 22–27°C (Vaudo et al. 2016). In this study, shortfin mako adjusted to temperature changes by diving less deeply in colder waters and more deeply in warmer waters. Depths of shortfin mako ranged from 28–866 m, and waters of < 15°C created a lower depth limit to diving behaviors (Vaudo et al. 2016). During summer and autumn, shortfin mako had core home ranges extending along the US Atlantic Coast north of the Gulf Stream from the Carolinas, USA to Newfoundland, Canada (Vaudo et al. 2017). Winter and spring distributions expanded toward the south into the Caribbean Sea and northern South America, where offshore waters were frequently used (Vaudo et al. 2017).

For the blue shark, Howey et al. (2017) tagged individuals offshore of Massachusetts. They found blue sharks were on the continental shelf May–November 1 and used a maximum daily water temperature range of 12.1–23.1°C and an average daily maximum depth of 46 m (Howey et al. 2017). In addition Vandeperre et al. (2016) observed the following habitat associations:

- For small juveniles of both sexes, few blue sharks were observed with SST of < 15°C and in waters with the lowest primary productivity; more individuals were observed relatively close to the shelf break.
- Large juveniles/subadult females were associated with 1) a combination of warm SST in areas of high primary productivity and 2) areas of high primary productivity in a close proximity to the shelf.
- Large juvenile male blue shark were associated with 1) distance to 1-km isobaths and 2) a combination of warm SST plus high primary productivity.

3.5 Tuna, Swordfish, and Billfish Habitat Associations

The designated EFH for the Atlantic highly migratory species of tuna, swordfish, and billfish overlap only small amounts with our study areas in the GoM and South Atlantic. Therefore, species in those regions are not further addressed here. In contrast, skipjack tuna, albacore tuna, yellowfin tuna, and bluefin tuna do have substantial overlap in the Greater Atlantic study area. However, direct linkages of these species to shoal habitats are not well defined (Rutecki et al. 2014), and we found in next section that few studies of large pelagic fish have considered testing for geomorphology relationships. Because of these characteristics, plus the complexity involved with research on these species, we have limited our review here to bluefin tuna in the Greater Atlantic. The EFH designation of bluefin tuna overlaps with 99% of our Greater Atlantic study area (**Table 3-8**). More specifically, designated EFH for juvenile bluefin tuna is throughout our Greater Atlantic study area, and EFH for adults is restricted to offshore Virginia, northern New Jersey, as well as waters east of Long Island Sound, New York.

Table 3-8. For Highly Migratory Species (excluding sharks), the proportion of area designated as EFH within each study area.

Species group	Species	Proportion of GoM study area designated as EFH	Proportion of South Atlantic study area designated as EFH	Proportion of Greater Atlantic study area designated as EFH	Fishbase description of habitat
Tuna	Albacore tuna	0.00	0.06	**0.52**	pelagic 0–600 m
Tuna	Bigeye tuna	0.00	0.09	0.14	pelagic 0–1,500 m usualy 0–500 m
Tuna	Bluefin tuna	0.00	0.23	**0.99**	pelagic 0–985 m usually 0–100 m
Tuna	Skipjack tuna	0.01	0.14	**0.82**	pelagic 0–260 m
Tuna	Yellowfin tuna	0.00	0.23	**0.60**	pelagic 1–250 m usually 1–100 m
Swordfish	Swordfish	0.02	0.29	0.16	pelagic 0–2,878 m usually 0–550 m
Billfish	Blue marlin	0.00	0.00	0.00	pelagic 0–1,000 m
Billfish	Longbill spearfish	0.00	0.06	0.17	pelagic 0–200 m usually ≥ 100 m
Billfish	Roundscale spearfish	0.00	0.00	0.00	pelagic 0–200 m
Billfish	Sailfish	0.05	0.36	0.08	pelagic 0–200 m usually ≥ 30 m
Billfish	White marlin	0.00	0.00	0.00	pelagic 0–150 m usually 0–100 m

Notes: Those overlapping with > 50% of the study area are bolded. Study areas are defined in **Figures 1-2, 1-3, and 1-4**. All life stages were included.

Within the US, bluefin tuna primarily spawn in the deeper waters of the GoM and their main feeding areas are the extremely productive Atlantic shelf (Druon et al. 2016). Worldwide, bluefin tuna seasonal migrations track changes in chlorophyll, SST, and temperature fronts (Druon et al. 2011; Royer et al. 2004). Research has also found that bluefin tuna in the North Atlantic are associated with anticyclonic eddies that downwell water and effectively mix the water column (Hsu et al. 2015). Within the North Atlantic region, Walli et al. (2009) found bluefin tuna have long residence times (mean= 167 days per year), and their diving behavior was correlated with the depth of the thermocline (i.e., deeper dives were made if thermocline was deeper). Marcek et al. (2016) also found juvenile bluefin tuna diving behavior was related to thermocline depth. Water temperatures of bluefin tuna range widely as 0–31℃, but 87% of their time were in waters 10–23℃ (Walli et al. 2009). Walli et al. (2009) and Galuardi et al. (2012) both found that bluefin tuna used waters near the surface (< 20 m depth) the vast majority of time, but the depth of the water column was not reported.

After a literature review of habitat associations that affect bluefin tuna distribution, Druon et al. (2016) developed a habitat suitability index model based the range of four habitat variables (**Table 3-9**).

Table 3-9. Parameters used to develop habitat suitability models for bluefin tuna.

Size class of bluefin tuna	Variable	Minimum value	Intermediate value	Maximum value
Small (5–25 kg)	Chlorophyll (mg m^{-3})	0.10	0.25	1.95
Small (5–25 kg	Gradient of chlorophyll (mg m^{-3} km^{-1})	0.0008	0.0030	NA
Small (5–25 kg	SST (℃)	13	NA	26.1
Small (5–25 kg	Sea surface height anomaly (m)	NA	NA	-0.10
Large (5–25 kg)	Chlorophyll (mg m^{-3})	0.14	0.25	4.42
Large (5–25 kg)	Gradient of chlorophyll (mg m^{-3} km^{-1})	0.0008	0.0030	NA
Large (5–25 kg)	SST (℃)	7.5	NA	24
Large (5–25 kg)	Sea surface height anomaly (m)	NA	NA	-0.10

Source: Druon et al. (2016)

In regard to seasonal movements in the Atlantic, juvenile bluefin tuna were often located south of Delaware waters during November–April 30, and then moved northward May–October 31 (Galuardi and Lutcavage 2012). The diet of bluefin tuna does provide some inference into their habitat use. Chase et al. (2002) found sand lance, Atlantic herring, Atlantic mackerel, squid, and bluefish were the most common prey of juvenile and adult bluefin tuna.

4 Where are the Marine Fish? A Literature Review of Spatially Explicit Habitat Associations and Models of Fish Distribution

Key Points and Knowledge Gaps (*gaps are in italics*)

- Spatially explicit predictive models of marine fish have increased rapidly in recent years. Worldwide, only two such studies were published in 2007, and the trend increased to an annual high of 42 predictive studies published in 2018.
- A total of 7 predictive marine fish studies have been published in the Gulf of Mexico, 4 in the South Atlantic, and 19 in the Greater Atlantic.
- Nonlinear statistical techniques are common, showing that species-habitat relationships are often complex.
- Oceanographic and water chemistry habitat characteristics tend to dominate modeling, while *substrate associations are poorly studied for 8 of 10 fish guilds. The effects of soft bottom complexity on fish distribution remains largely unknown.*
- *Further progress can be made between marine fish ecology and predictive modeling to determine where spatial models can be improved for management and conservation applications*.

4.1 Introduction

Species distribution models, and similar methods of spatial modeling, have proliferated in the last two decades due to rapid improvements in Geographic Information Systems (GIS) technologies, remote sensing, availability of spatial data, and computation capacity. However, modeling of marine species has lagged far behind the terrestrial counterpart (Robinson et al. 2011). For terrestrial ecosystems, landscape ecology has been defined as the study of heterogeneous spatial patterns and processes, and it has a history of merging the fields of ecology and geography (Turner 1989). These concepts are now being applied to marine ecosystems in the form of “seascape ecology” (Pittman 2011). The distribution of marine fish are particularly important because of their commercial and recreational economic value, as well as their value for subsistence fishing. Recent distribution models in marine ecosystems have been applied to ecosystem-based management (Gruss et al. 2018; Moore et al. 2016), marine spatial planning (Hattab et al. 2013), scenario assessments (Delevaux et al. 2018), stock assessments (Saul et al. 2013), and climate change scenarios (Morley et al. 2018; Su et al. 2013).

As marine spatial modeling continues to expand, a synthesis of studies has the potential to inform best modeling practices, identify knowledge gaps, assess research trends, identify the most useful datasets, and improve the efficiency of future studies by identifying environmental variables commonly invoked. Here, we focus on marine fish species because they are economically and socially important, represent diverse marine environments, occupy multiple trophic levels, and likely to respond to a similar suite of environmental characteristics. Spatial modeling approaches have changed dramatically from habitat suitability indices initially developed in the 1970s (U.S. Fish and Wildlife Service 1981) to correlative species distribution models (Guisan and Zimmermann 2000), individual-based models (Okunishi et al. 2009), and other data-driven techniques. Simultaneously, statistical methods in ecology have continued to evolve, as machine-learning techniques and flexible general additive models (GAMs) continue to gain popularity in ecology.

A vast of array of predictor variables have now been developed for marine environments. For example, the online database “Bio-Oracle” provides worldwide data on 23 types of predictor variables (Tyberghein et al. 2012). Predictive modeling studies have spanned from those using no predictor variables by applying kriging techniques (Rambo et al. 2017) to using > 30 types of predictor variables (Bouchet et al. 2017; Manderson et al. 2011). Additionally, each variable may be computed at multiple spatial (e.g., Pittman et al. 2009) and temporal scales (Mannocci et al. 2017), which can result in a bewildering array of options. In regard to spatial scale of analyses, Mannocci et al. (2017) recognizes a hierarchy where marine prey patches determine fine-scale habitat associations (1 m–1 km), eddies/fronts/oceanographic features occur at an intermediate scale (1–100 km), and water masses/currents dictate broad-scale (100–1,000 km) distributions of organisms. However, the mobility and guild of fish species are likely to play a role in their responses to scale. Robinson et al. (2011b) suggests that pelagic species may be well represented by coarse data because heterogeneity of their environment occurs at a broad scale (e.g., 10–100 km scale), whereas coastal and benthic species may respond to a relatively fine scale because of the existence of local heterogeneity. Therefore, predictors of pelagic fish species are likely to differ from demersal or reef-associated species. Understanding these differences and patterns will be help further develop models for fish species and will help elucidate knowledge gaps.

Given the nascent nature of marine spatial modeling, a complex array of potential predictor variables, high diversity of marine fish species and data sources, and evolving statistical methods, we sought to provide clarity to the state of predictive modeling of marine fish. We conducted a literature review with the following objectives:

- Identify the most and least frequently studied geographies and fish functional groups
- Describe the frequency of various statistical techniques
- Quantify differences in fisheries data sources used for fish functional groups
- Determine how predictor variables that are tested differ by functional group
- Identify and discuss knowledge gaps

4.2 Literature Search Methods

We conducted a literature search of peer-reviewed research articles with the Web of Science database. Articles from 2007–2018 were included, and the search was conducted January 16, 2019. The following keywords were searched within the title and abstract: TS = ("species distribution" OR "ecological niche*" OR "bioclimatic envelope" OR "habitat suitability" OR "habitat model*" OR "spatial distribution" OR "seascape") AND TS = fish AND TS= ("marine" OR "ocean"). After finding notable articles on shark spatial distribution modeling were missing, we conducted a second search by simply replacing “fish” with “shark.” From the two searches, a total of 1,648 articles were obtained.

4.3 Review Scope and Protocol

The scope of the review included only fish species; therefore, taxonomic groups such as squid, shrimp, crabs, other crustaceans, corals, and bivalves, are not covered here. We limited the review to those research articles that resulted in a spatial prediction of fish distribution beyond fish survey locations. Although localized studies may greatly improve our understanding of fish ecology and distribution, ultimately we were interested in predicting and mapping the distribution of fish over relatively broad areas. No limitations were placed on the dependent variables, as predictions of individual species presence/relative abundance/relative biomass, species richness, species diversity, or other measures of community composition were all considered relevant. Given our focus and objectives, we removed the following types of studies from the initial keyword search:

- Studies that did not include predictions of fish distribution (e.g., whales, invertebrates, seabirds)

- Studies focused solely on the distribution of fishing effort
- Genetic or evolutionary studies that addressed long-term, broad-scale connectivity
- Study areas focused solely on salt marsh, mangrove, intertidal, estuary, or freshwater environments (< 8 ppt)
- Simulations of larval dispersal; studies of larvae were included when spawning was inferred
- Review or discussion articles
- Center of gravity studies, as they do not predict the full distribution of species
- Reserve design or conservation planning studies that used existing models
- Marine Protected Area (MPA) studies were examined to ensure those that predicted the distribution of fish were included. Those studies that compared MPA with non-MPA sites were often excluded as spatial predictions and analyses were not developed

The geography of each study area was recorded as its ocean, nearest continent (hereby, "continent"), nearest nation (hereby, "nation"), specific geography name (if given), and year published. For each study, we categorized the type of fish species analyzed into one of 10 guild categories **(Table 4-1)**. A few studies included only the juvenile stage of fish, and thus, these studies were characterized by the habitat of those juvenile fish regardless of adult stage habitat associations. To improve our sample size of shark studies, the four studies that included both sharks and other species (elasmobranch diversity, grouper, and large pelagics) were categorized as shark studies. Otherwise, we used a "general" category to include studies that included more than one fish guild. These general studies often examined species richness, species diversity, or all of the species captured in a particular survey. Fish data sources were categorized as fishery independent, fishery dependent, both fishery independent and dependent, international database, previous research/museum specimens, and interviews. The specific method of fishery-independent surveys was also recorded. Fishery-dependent data included logbooks, landings, catch observations, incidental catch observations, and recreational catch. International databases included "Sea Around Us," "SeaLifebase" geographic range polygons as well as databases of raw species data such as the "Ocean Biogeographic Information System."

For statistical methods, we recorded the type and total number of statistical methods applied for each study and the total number of machine-learning methods. Machine-learning methods are relatively new statistical methods in ecology and are based on iterative learning. These techniques included artificial neural networks, classification and regression trees, random forest, boosted regression trees, and multivariate adaptive regression splines. We also summarized the different methods that applied bioclimatic envelopes (surface range envelopes, AquaMaps, Sea Around Us, bioclimatic envelope). We did not distinguish models with only fixed effects and those with random effects included. Spatial autocorrelation was not documented, though geographic variables were included with the variables tested.

We recorded each variable tested, or otherwise utilized (e.g., in habitat suitability indices), in models of fish distribution. However, we did not include temporal factors (year, month, day of year, day/night) or factors primarily affecting detectability (moon phase, lunar illumination, wind, clouds, precipitation) because of our primary interest in habitat relationships and best predictors for mapping fish distribution. We further summarized variables into categories of fish-based oceanographic, physical oceanographic, geography, substrate, and biological. Fish-based oceanographic variables included those that have direct influence on fish, such as temperature, salinity, and nutrients. In contrast, physical oceanographic factors are hypothesized to have an indirect effect on fish through enhanced productivity via sea-level anomalies, chlorophyll-*a*, upwelling events, and ocean currents. Geographic depicted variables focused on location or proximity to surrounding ecosystems, such as latitude/longitude, distance to shoreline, distance to shelf, and distance to other ecosystems like mangroves or estuaries. Substrate variables characterized components like sediment type, bottom types, and topography.

Table 4-1. Fish functional group, examples of common species, and the number of predictive modeling papers obtained from the Web of Science database from 2007–2018.

Fish functional group	Common example species	Number of studies	Fishery independ-ent	Fishery dependent	Fishery independ-ent and dependent	International database	Museum or previous research
Demersal (not associated with hard bottom)	cod, stingrays, hake, whiting, sole, flounder, halibut, and other flatfish	59	55 (93%)	5 (8%)	5 (8%)	4 (7%)	2 (3%)
General	species richness, species diversity, or a combination of the groups; seahorse, aquaculture species	40	15 (38%)	6 (15%)	1 (3%)	17 (43%)	2 (5%)
Large pelagics	tuna, swordfish, marlin, billfish, sailfish, and ocean sunfish	29	10 (34%)	15 (52%)**	0	1 (3%)	1 (3%)
Sharks	-	21	10 (48%)	9 (43%)**	2 (10%)	2 (10%)	2 (10%)
Forage fish	Atlantic herring, capelin, sardine, anchovy, lanternfish, and sand lance	18	11 (61%)	4 (22%)	2 (11%)	2 (11%)	0
Coral reef	-	16	13 (81%)	1 (6%)	1 (6%)	1 (6%)	2 (13%)
Hard bottom	grouper, snapper	14	14 (100%)	3 (21%)	3 (21%)	0	0
Medium pelagics	mackerel, dolphinfish, eel species, yellow kingfish	15	3 (20%)	10 (67%)**	0	2 (13%)	0
Diadromous	shad, salmon	7	4 (57%)	2 (29%)	1 (14%)	1 (14%)	1 (14%)
Invasive	-	7	2 (29%)	0	0	3 (43%)	2 (29%)
Totals	-	**226**	**137 (61%)**	**55 (24%)**	**15 (7%)**	**33 (15%)**	**12 (5%)**

Notes: The source of fish data is described as a percent of studies within each defined guild. The category of fisheries-independent + fisheries-dependent data is not exclusive of individual categories.

4.4 Statistical Analysis

We reviewed 226 peer-reviewed scientific papers. We tested the trend in number of studies across years with a generalized linear model (GLM) with a Gaussian distribution. We used a multinomial logistic regression with fish functional group as a dependent variable and separately tested for differences in data sources (simplified to fishery-independent, fishery-dependent, and international databases), statistical method, and type of predictor variables (fish-based oceanographic, physical oceanographic, geographic, substrate, and biological). For multinomial logistic regression, we used the R package “nnet” and Wald tests to calculate p-values ($\infty = 0.05$). Because demersal species were the most common fish functional group, we used them as a reference group in the analyses. A single study, Gruss et al. (2018), was not included in the analyses of habitat variables because they studied 51 functional groups of fish and invertebrates and each one had a differing, consolidated number of predictor variables. All values reported are ± 1 standard error (SE).

To further consider habitat variables tested by functional group, we invoked linear discriminant analysis (LDA), which is a statistical, machine-learning method for identifying linear combinations of variables that maximizes the separation of known data groupings (Rao 1948; Ripley and Hjort 1996). Here, we utilized LDA to identify the linear combinations of study characteristics that separate the specified fish functional groups. We performed LDA with the “MASS” package “lda” function (Ripley 2002; Ripley and Hjort 1996) in RStudio (Version 1.1.383). Additional packages used “irr” (Gamer et al. 2019) and “scatterplot3d” (Ligges and Mächler 2002). For this analysis, we discarded those variables in < 1% of studies. We condensed the classes of medium and large pelagics into “pelagics” and hard bottom and coral reef fish into a category of “reef fish.”

We first fit an LDA model with a set seed that considered the set of habitat characteristics and examined the percentage of the trace explained by each of the linear discriminants. We included the appropriate number of discriminants to maintain interpretability and still lead to good separation of the groupings by explaining a sum of at least 75% of the trace. Next, we examined the separation of the groups from plots of the discriminants to understand where the groups lie based on the linear combinations of discriminants. We examined the discriminant coefficients to identify the study characteristics that are likely to lead to this separation. Finally, we examined agreement statistics including percent agreement and Cohen’s kappa between the estimated and true groupings to understand how well the model fits the data.

4.5 Results and Discussion

The number of marine fish predictive modeling papers had a strong upward trend from 2007–2018 (GLM, t-value = 7.91, $\beta = 3.02$ (± 1 SE, 0.38), $p < 0.0001$) (**Figure 4-1**). Only two such studies were published in 2007, but 42 predictive modeling papers were published in 2018. Of the studies focused on a single ocean, the Atlantic Ocean was most studied (88 studies), followed by the Pacific (71), Mediterranean Sea (27), Indian (12), Southern (8), and Arctic (2) **(Figure 4-2)**. In regard to continents of study, North America (70 studies), Europe (38), and Australia (29) were most studied, followed by Asia (19), South America (6), Africa (3), and Antarctica (1). Taken together, the Atlantic Ocean coasts of North America and Europe, plus the Mediterranean Sea encompassed 46% of all predictive modeling published research on marine fish. Twenty-eight studies (12%) covered multiple oceans, including 11 that were worldwide in scope.

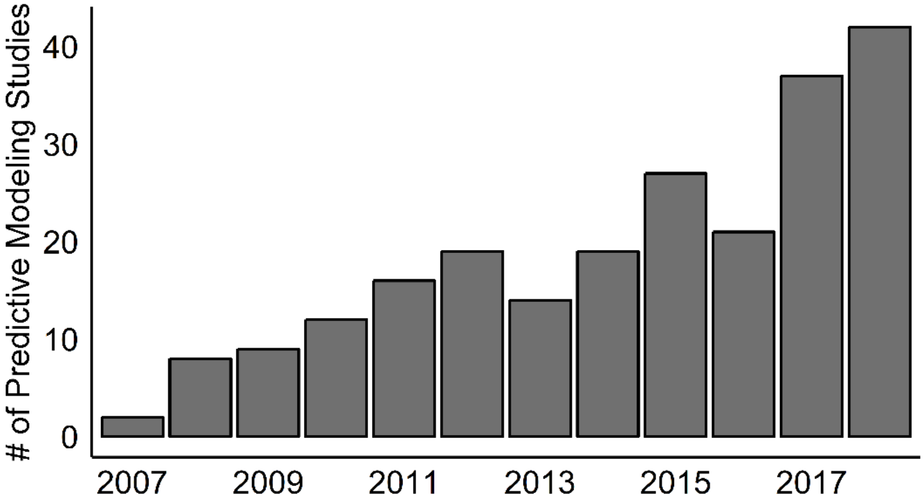


Figure 4-1. Number of peer-reviewed marine fish spatially explicit predictive model studies 2007–2018.

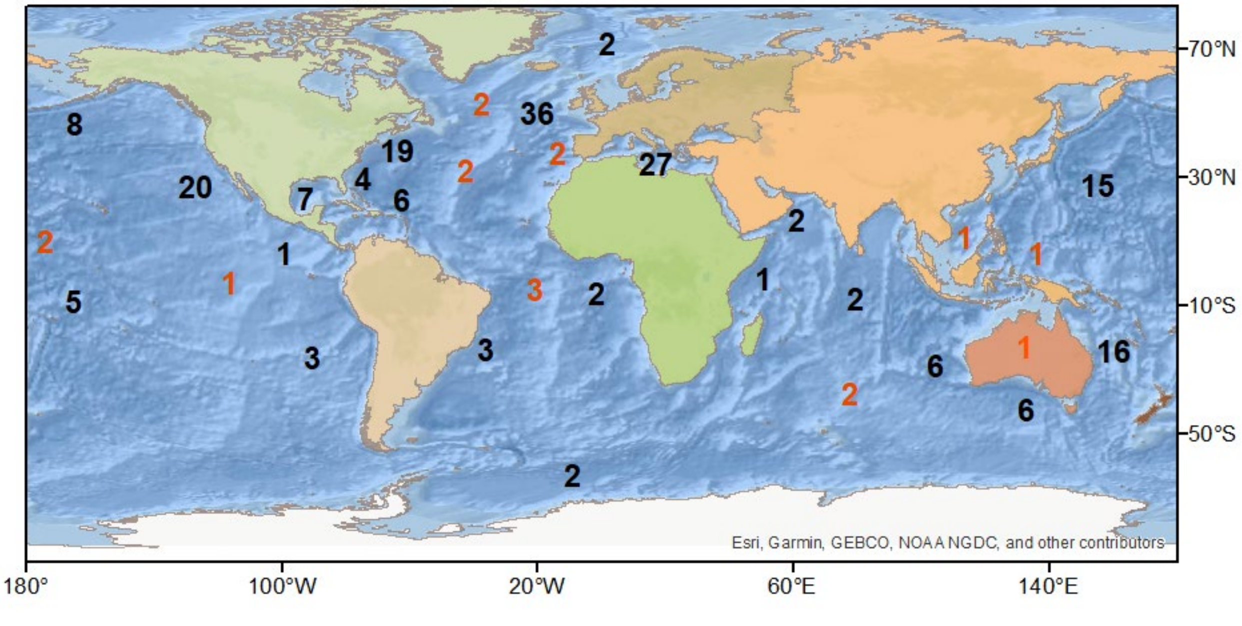


Figure 4-2. Geography of studies that predicted the distribution of marine fish (2007–2018) (n=226).

Studies that depicted multiple oceans, or covered extremely large geographies, are shown in orange and are located near the center of their extent. Single ocean studies are shown in black. Eleven studies were worldwide (not shown), and one study is counted in both the entire Atlantic and Indian/Southern Ocean.

Studies of demersal species were most common, followed by general studies and large pelagic fish (**Table 4-1**). Shark, forage fish, coral reef, hard bottom, and medium pelagic fish were moderately studied, while the distribution of diadromous and invasive species were rarely modeled. Fish data sources used for modeling were dominated by fishery-independent surveys (61%), followed by fishery-dependent surveys (24%), international databases (15%), fishery-independent and dependent surveys (7%), and previous research or museum specimens (5%). Of the 137 studies that utilized fishery-independent data sources, 51% were by trawl, 20% video, 12% field survey (e.g., scuba diving), 8% ichthyoplankton survey, 8% bottom/vertical longline, 7% acoustic survey, 4% satellite tag, 3% traps, 3% gill/seine net, and 2% citizen science.

GAMs were the most common modeling method (**Table 4-2**). The high frequency of GAMs and the emergence of machine-learning statistics emphasizes the importance of nonlinear habitat relationships. Together, these statistical techniques were present in 40% of all predictive marine fish studies. Of the machine-learning methods, boosted regression trees (6% of all studies), classification trees (5%), and random forests (4%) were most common, followed by mulivariate adaptive regression splines (3%) and artificial neural networks (3%). Seventeen percent of all studies used more than one statistical method. Individual-based models, quantile regression, occupancy models, and ordinary least squares were each used in < 2% of studies.

As the statistical reference, demersal studies used GAMs (34%), machine learning (10%), GLMs (29%), Maxent (12%), habitat suitability indices (5%), multivariate (5%), envelopes (3%), and geostatistics (15%). As expected from studies that typically analyzed tens to hundreds of species (e.g., species richness), envelope methods were more common in general species studies ($p < 0.005$, $\beta = 2.96 \pm 1.05$). Maxent models, based on presence-only modeling methods, were more common with invasive species ($p = 0.019$ $\beta = 2.73 \pm 1.16$). Habitat suitability indices were more common with invasive species ($p = 0.07$ $\beta = 2.30 \pm 1.28$) and pelagics ($p = 0.003$ $\beta = 2.32 \pm 0.77$). The use of geostatistic techniques were negatively associated with pelagics ($p = 0.08$, $\beta = -1.93 \pm 1.11$), reef fish ($p = 0.09$ $\beta = -1.88 \pm 1.11$), and geostatistics were not used with studies of invasive fish, diadromous, general species. For pelagics, GLMs ($p = 0.064$ $\beta = -1.37 \pm 0.74$) were less common and GAMs were more commonly used ($p = 0.04$ $\beta = 1.06 \pm 0.52$). Machine-learning methods were more common with reef fish ($p = 0.08$, $\beta = 1.17 \pm 0.67$). Multivariate methods were not related to fish guild.

Demersal-, forage-, coral reef-, hard bottom- and diadromous fish studies were dominated by fishery-independent data sources (**Table 4-1**). Compared to demersal species, pelagics ($p = 0.03$, $\beta = 1.42 \pm 0.65$) and sharks ($p = 0.09$, $\beta = 1.23 \pm 0.73$) had more research based on fishery-dependent data. International databases served as a major data source for invasive species and general species models, but they did not statistically differ from demersal species. Pelagic models less commonly used international databases ($p= 0.03$, $\beta = -2.37 \pm 1.08$), and no studies of invasive species used fishery-dependent data. With the exception of reef fish, all other fish guilds had fewer studies from fishery-independent data compared to demersal fish studies (all $p < 0.02$).

Table 4-2. Methods used to predict the distribution of marine fish in studies spanning 2007–2018 (n = 226).

Statistics for modeling	Percent of studies
General additive model	32
Generalized linear model	19
Maxent	16
Habitat suitability index	13
Machine learning	11
Multivariate statistics	9
Envelope models	8
Geostatistics/kriging	8
Bayesian	5

Note: Methods used in < 2% of papers are not shown; studies may have used multiple methods.

LDA results (**Figure 4-3**, **Table 4-3**) showed fish groups could be separated based on what habitat variables were tested. **Table 4-3** includes the discriminant coefficients from the three discriminants of the LDA model and indicator of the driving variables for separation of our most separable groups. The median values are evident from the scatterplots (**Figure 4-3**). The checkmarks indicate the characteristics that are most likely included in studies for the given fish group.

A 3D scatterplot and three pairwise scatterplots describe the three linear discriminants utilized for this separation as they together explain 77% of the trace (39%+20%+17%=77%). From these, we saw good separation for demersal, general, invasive, pelagic, and reef fish groups. Finally, the model fit the data fairly well with percent agreement of 67.6 and Cohen's kappa of 0.60 ($\kappa = 0$ is random agreement, $\kappa = 1$ is perfect agreement).

Overall, the following results were observed from LDA:

- Reef fish were most distinguished because studies often examined substrates attributes, such as rugosity, standard deviation of depth, aspect, sessile biota, slope of slope (a derivative of slope), and reefs.
- Demersal studies were represented by depth, sediment grain size, bottom temperatures, soft bottom proportion, or distance to soft bottom. Substrate variables related to substrate complexity were not a characteristic of this group.
- Pelagic species were represented by sea surface height anomaly, DO, pH, phosphate, and prey.
- Invasive species were represented by common, broad-scale variables of SST, chlorophyll, and salinity.
- General species (e.g., species richness) were only distinguished by ice and salinity measures.

Overall, these patterns show how researchers have perceived these marine fish functional groups and demonstrates knowledge gaps that will require further testing.

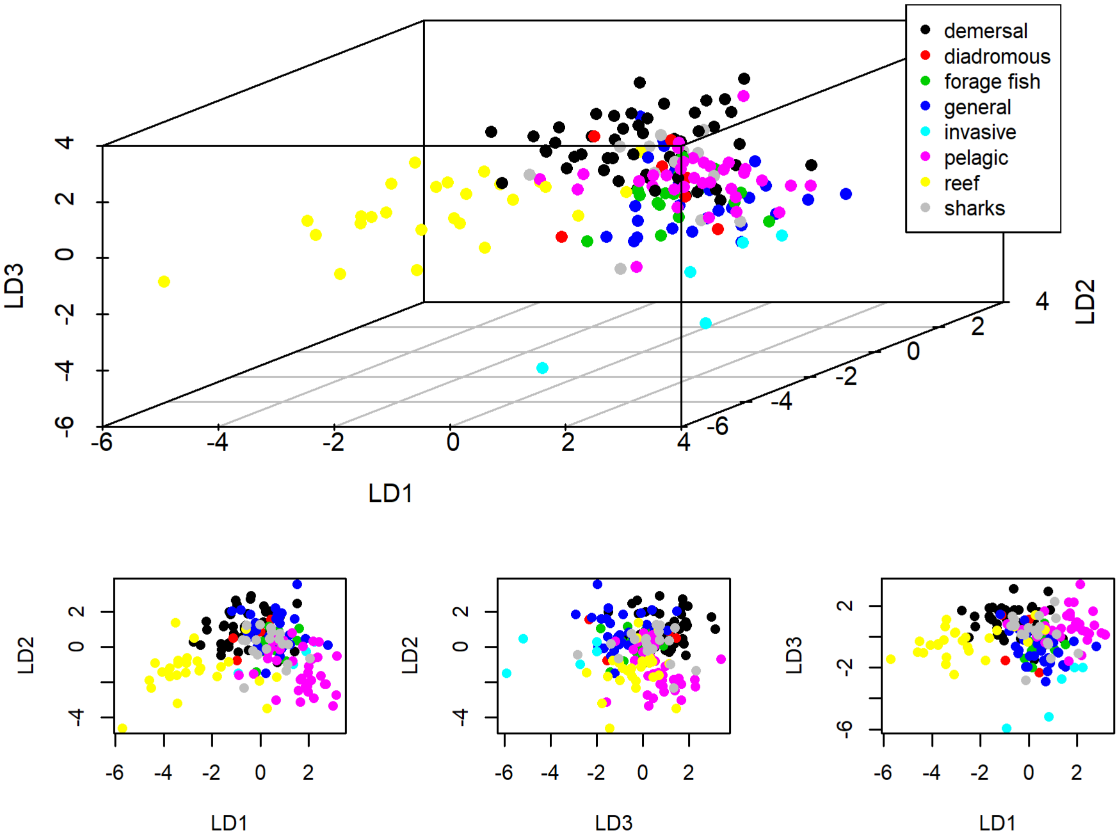


Figure 4-3. LDA results that show habitat variables tested in predictive modeling studies differ with fish functional groups.

Table 4-3. Discriminant coefficients from the three discriminants of the LDA and indicators of the driving variables for separation of the most separable groups.

Variable	LD1	LD2	LD3	Demersal (-0.56, 0.76, 0.73)	General (0.64, 0.71, -0.90)	Invasive (0.82, -0.14, -2.73)	Pelagic (1.81, -1.56, 0.44)	Reef (-3.08, -1.28, -0.30)
Depth	-0.43	0.61	0.54	✓	-	-	-	-
Ice	0.04	1.31	-1.75	-	✓	-	-	-
SST anomaly, climatic variable	-0.38	0.11	0.71	✓	-	-	-	-
Sea surface height anomaly	0.88	-1.73	1.75	-	-	-	✓	-
Bottom temperature	-0.06	0.16	0.59	✓	-	-	-	-
SST	0.98	-0.55	-0.53	-	-	✓	-	-
Chlorophyll	0.39	-0.04	-0.49	-	-	✓	-	-
Salinity at surface	0.42	0.28	-0.62	-	✓	✓	-	-
Salinity at bottom	0.46	0.19	-1.43	-	✓	✓	-	-
Dissolved oxygen	0.18	-0.40	0.93	-	-	-	✓	-
Phosphate	2.39	-5.18	2.50	-	-	-	✓	-
pH	1.93	-2.52	1.21	-	-	-	✓	-
POC	-0.57	1.65	0.76	✓	-	-	-	-
Prey	0.34	-0.25	0.02	-	-	-	✓	-
Anthropogenic stress	-0.96	-0.94	-1.17	-	-	-	-	✓
Distance to shoreline	-0.40	-0.51	-0.08	-	-	-	-	✓
Sediment grain size	-0.26	1.30	0.98	✓	-	-	-	-
Soft bottom (proportion or distance to)	-0.06	1.65	0.90	✓	-	-	-	-
Sessile biota	-1.41	-0.87	-0.15	-	-	-	-	✓
Reef or hard bottom	-0.58	-1.23	-0.30	-	-	-	-	✓
Slope of slope	-1.85	-2.40	-0.73	-	-	-	-	✓
Rugosity	-0.41	-0.61	-1.41	-	-	-	-	✓
SD of depth	-0.66	-1.03	-1.05	-	-	-	-	✓
Aspect	-0.43	-0.37	-1.17	-	-	-	-	✓
Seagrass, macroalgae, algae	0.37	-0.12	-1.21	-	-	✓	-	-
Habitat type or patch area	-1.20	0.51	0.87	✓	-	-	-	-

Notes: The values in parentheses are the median positions for these groups on the three linear discriminants.

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Appendix A. Supplemental Tables

Table A-1. Common and scientific names of species cited in the text.

Table modified from Rutecki et al. (2014).

Common Name	Scientific Name	Common Name	Scientific Name
Alewife	*Alosa pseudoharengus*	Flounder, winter	*Pseudopleuronectes americanus*
AmberJack	*Seriola* spp.	Haddock	*Melanogrammus aeglefinus*
AmberJack, greater	*Seriola dumerili*	Hake, white	*Urophycis tenuis*
AmberJack, lesser	*Seriola fasciata*	Halfbeak	*Hemiramphus brasiliensis*
Anchovy	Family: Engraulidae	Halibut, Atlantic	*Hippoglossus hippoglossus*
Atlantic bigeye	*Priacanthus arenatus*	Herring, Atlantic	*Clupea harengus*
Atlantic bumper	*Chloroscombrus chrysurus*	Herring, Atlantic thread	*Opisthonema oglinum*
Atlantic spadefish	*Chaetodipterus faber*	Herring, blueback	*Alosa aestivalis*
Atlantic tomcod	*Microgadus tomcod*	Herrings	Clupeidae
Anchovy, bay	*Anchoa mitchilli*	Herring, Pacific	*Clupea pallasii pallasii*
Atlantic, silverside	*Menidia menidia*	Herring, thread	*Opisthonema oglinum*
Bass, largemouth	*Micropterus salmoides*	Hind, red	*Epinephelus guttatus*
Bass, striped	*Morone saxatilis*	Hind, rock	*Epinephelus adscensionis*
Bermuda chub	*Kyphosus sectatrix*	Hind, speckled	*Epinephelus drummondhayi*
Bluefish	*Pomatomus saltatrix*	Hogfish	*Lachnolaimus maximus*
Bluegill	*Lepomis macrochirus*	Horse-eye jack	*Caranx latus*
Blue runner	*Caranx crysos*	Lobster, American	*Homarus* americanus
Butterfish	*Peprilus triacanthus*	Lobster, spiny	*Panulirus argus*
Clam, Atlantic surf	*Spisula solidissima*	Lobster, slipper	Scyllaridae
Clam, northern quahog	*Mercenaria mercenaria*	Longhorn sculpin	*Myoxocephalus octodecemspino*
Clam, ocean quahog	*Arctica islandica*	Mackerel, Atlantic	*Scomber scombrus*
Clam, quahog	*Mercenaria campechiensis*	Mackerel, chub	*Scomber colias*
Cobia	*Rachycentron canadum*	Mackerel, king	*Scomberomorus cavalla*
Cod, Atlantic	*Gadus morhua*	Mackerel, king and cero	*Scomberomorus* spp.
Coney	*Cephalopholis fulva*	Mackerel, Spanish	*Scomberomorus maculatus*

Common Name	Scientific Name	Common Name	Scientific Name
Crab, horseshoe	*Limulus polyphemus*	Mako, shortfin	*Isurus oxyrinchus*
Crab, blue	*Callinectes sapidus*	Menhaden, Atlantic	*Brevoortia tyrannus*
Crab, Florida stone	*Menippe mercenaria*	Menhaden, Gulf	*Brevoortia patronus*
Crab, Gulf stone	*Menippe adina*	Mullet	*Mugil* spp.
Crab, Dungeness	*Metacarcinus magister*	Oyster	Ostreidae
Croaker, Atlantic	*Micropogonias undulatus*	Pollock	*Pollachius virens*
Dogfish, smooth	*Mustelus canis*	Porgies	*Calamus* spp.
Dogfish, spiny	*Squalus acanthias*	Porgy, red	*Pagrus pagrus*
Dolphinfish	*Coryphaena hippurus*	Pout, ocean	*Zoarces americanus*
Drum, red	*Sciaenops ocellatus*	Redfish, Acadian	*Sebastes fasciatus*
Dwarf sand perch	*Diplectrum bivittatum*	Red hind	*Epinephelus guttatus*
Flounder, southern	*Paralichthys lethostigma*	Sheepshead	*Archosargus probatocephalus*
Flounder, summer	*Paralichthys dentatus*	Sand perch	*Diplectrum formosum*
Flounder, windowpane	*Scophthalmus aquosus*	Sardines	Clupeidae
Flounder, winter	*Pseudopleuronectes*	Scads	Carangidae
Flounder, witch	*Glyptocephalus cynoglossus*	Shrimp, seabob	*Xiphopenaeus kroyeri*
Flounder, yellowtail	*Limanda ferruginea*	Shrimp, rock	*Sicyonia brevirostris*
Flounder, American plaice	*Hippoglossoides platessoides*	Shrimp, royal red	*Pleoticus robustus*
Gag	*Mycteroperca microlepis*	Shrimp, sand	*Crangon* spp.
Goosefish (monkfish)	*Lophius americanus*	Shrimp, white	*Litopenaeus setiferus*
Gobies	Suborder: Gobioidei	Skates	Rajidae
Graysby	*Cephalopholis cruentata*	Skate, barndoor	*Dipturus laevis*
Groupers	*Serranidae* spp.	Skate, little	*Leucoraja erinacea*
Grouper, goliath	*Epinephelus itajara*	Snapper, blackfin	*Lutjanus buccanella*
Grouper, warsaw	*Hyporthodus nigritus*	Snapper, cubera	*Lutjanus cyanopterus*
Grouper, orange-spotted	*Epinephelus coioides*	Snapper, gray	*Lutjanus griseus*
Grouper, red	*Epinephelus morio*	Snapper, lane	*Lutjanus synagris*
Grunts	*Haemulon* spp.	Snapper, mutton	*Lutjanus analis*

Common Name	**Scientific Name**	**Common Name**	**Scientific Name**
Grouper, yellowedge	*Hyporthodus flavolimbatus*	Snapper, pink	*Pagrus auratus*
Grouper, yellowmouth	*Mycteroperca interstitialis*	Snapper, red	*Lutjanus campechanus*
Grouper, snowy	*Hyporthodus niveatus*	Snapper, gray	*Lutjanus griseus*
Haddock	*Melanogrammus aeglefinus*	Snapper, silk	*Lutjanus vivanus*
Hagfish	*Myxine glutinosa*	Snapper, vermilion	*Rhomboplites aurorubens*
Hake, Atlantic, red/white	*Urophycis* spp.	Shark, bull	*Carcharhinus leucas*
Hake, offshore silver	*Merluccius albidus*	Shark, common thresher	*Alopias vulpinus*
Hake, red	*Urophycis chuss*	Shark, dusky	*Carcharhinus obscurus*
Hake, silver	*Merluccius bilinearis*	Shark, finetooth	*Carcharhinus isodon*
Rays	superorder: Batoidea	Shark, great	*Sphyrna mokarran*
Salmon	Salmonidae	Shark, lemon	*Negaprion brevirostris*
Salmon, Atlantic	*Salmo salar*	Shark, makos	*Isurus* spp.
Sand lance	*Ammodytes* spp.	Shark, porbeagle	*Lamna nasus*
Sand seatrout	*Cynoscion arenarius*	Shark, sand tiger	*Odontaspis taurus*
Scallop, bay	*Argopecten irradians*	Shark, sandbar	*Carcharhinus plumbeus*
Scallop, sea	*Placopecten magellanicus*	Shark, scalloped	*Sphyrna lewini*
Scamp	*Mycteroperca phenax*	Shark, silky	*Carcharhinus falciformis*
Scup	*Stenotomus chrysops*	Shark, smooth	*Sphyrna zygaena*
Scups or porgies	Sparidae spp.	Shark, spinner	*Carcharhinus brevipinna*
Sea bass, bank	*Centropristis ocyurus*	Shark, tiger	*Galeocerdo cuvier*
Sea bass, black	*Centropristis striata*	Shrimp, brown	*Farfantepenaeus aztecus*
Sea bass, rock	*Centropristis philadelphica*	Shrimp, pink	*Farfantepenaeus duorarum*
Seatrout, sand	*Cynoscion arenarius*	Shrimp, rock	*Sicyorzia brevirostris*
Shad, American	*Alosa sapidissima*	Short bigeye	*Pristigenys alta*
Seatrout, spotted	*Cynoscion nebulosus*	Slippery dick	*Halichoeres bivittatus*
Shark, Atlantic sharpnose	*Rhizoprionodon terraenovae*	Snappers	*Lutjaninae* spp.
Shark, blacknose	*Carcharhinus acronotus*	Snapper, gray	*Lutjanus griseus*
Shark, blacktip	*Carcharhinus limbatus*	Snapper, mutton	*Lutjanus analis*

Common Name	Scientific Name	Common Name	Scientific Name
Shark, blue	*Prionace glauca*	Snapper, silk	*Lutjanus vivanus*
Shark, bonnethead	*Sphyrna tiburo*	Snapper, vermilion	*Rhomboplites aurorubens*
Snapper, yellowtail	*Ocyurus chrysurus*	Snapper, blackfin	*Lutjanus buccanella*
Spot	*Leiostomus xanthurus*	Tuna, albacore	*Thunnus alalunga*
Squid, longfin	*Loligo pealei*	Tuna, bigeye	*Thunnus obesus*
Squid, northern shortfin	*Ilex illecebrosus*	Tuna, blackfin	*Thunnus atlanticus*
Squids	Squid spp.	Tuna, bluefin	*Thunnus thynnus*
Speckled hind	*Epinephelus drummonhayi*	Tuna, skipjack	*Katsuwonus pelamis*
Sturgeon, Gulf	*Acipenser oxyrinchus*	Tuna, yellowfin	*Thunnus albacares*
Sturgeon, shortnose	*Acipenser brevirostrum*	Tunas	*Thunnus* spp.
Swordfish	*Xiphias gladius*	Tunny, little	*Euthynnus alletteratus*
Tautog	*Tautoga onitis*	Walleye	*Sander vitreus*
Tilefish, blueline	*Caulolatilus microps*	Weakfish	*Cynoscion regalis*
Tilefish, golden	*Lopholatilus chamaeleonticeps*	Wenchman	*Pristipomoides aquilonaris*
Tilefish, sand	*Malacanthus plumieri*	Wolffish, Atlantic	*Anarhichas lupus*
Tilefishes	*Malacanthidae* spp.	Weakfish	*Cynoscion regalis*
Triggerfish, gray	*Balistes capriscus*	Wahoo	*Acanthocybium solandri*
Tomtate	*Haemulon aurolineatum*		
Trout	Subfamily: Salmoninae		

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The Department of the Interior protects and manages the Nation's natural resources and cultural heritage; provides scientific and other information about those resources; and honors the Nation's trust responsibilities or special commitments to American Indians, Alaska Natives, and affiliated island communities.

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The mission of the Bureau of Ocean Energy Management is to manage development of U.S. Outer Continental Shelf energy and mineral resources in an environmentally and economically responsible way.

BOEM Environmental Studies Program

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