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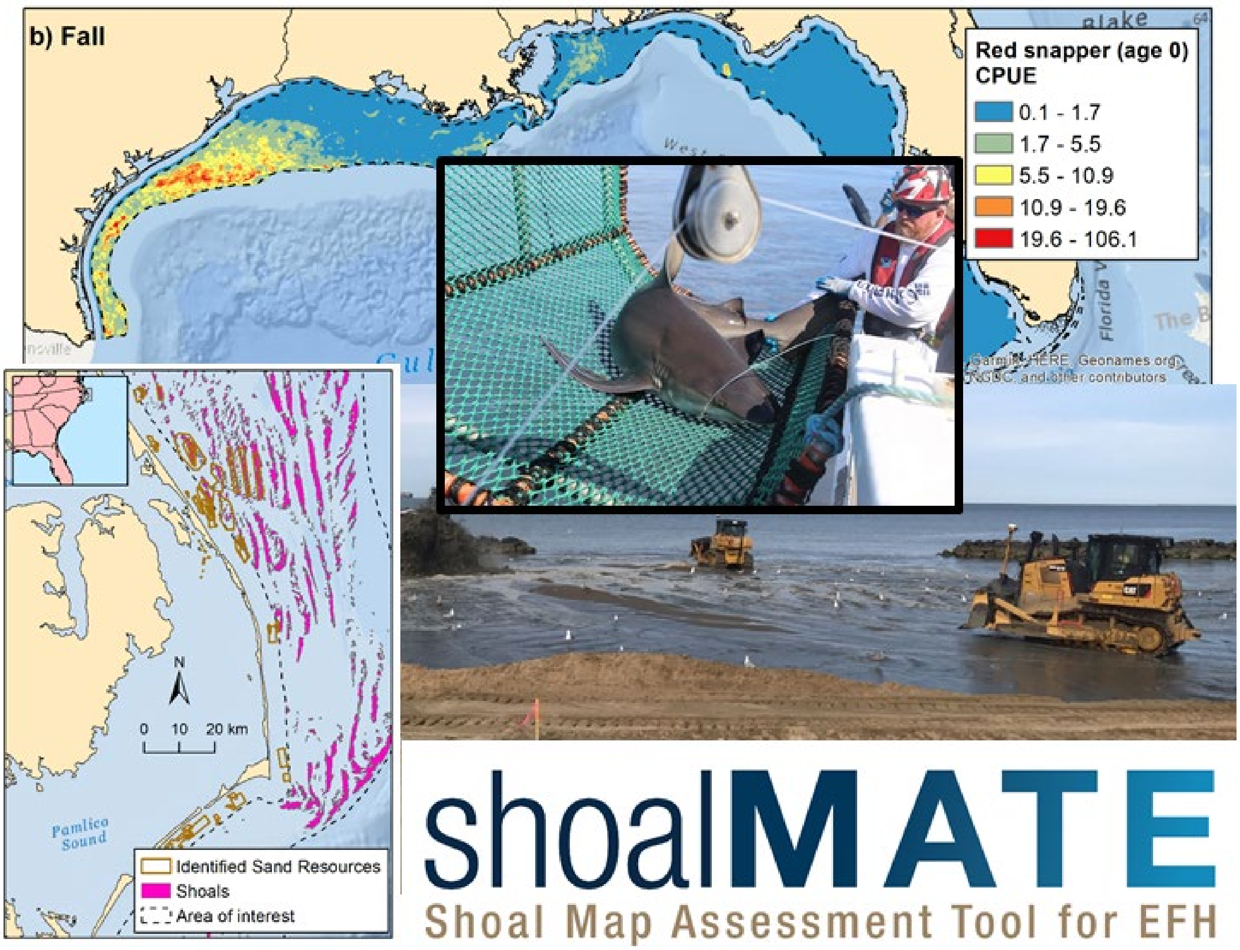
Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features



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Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features

January 2020

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DISCLAIMER

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List of Abbreviations and Acronyms

BOEM	Bureau of Ocean Energy Management
CMECS	Coastal Marine Ecological Classification Standard
CPUE	catch per unit effort
EFH	Essential Fish Habitat
Magnuson-Stevens Act	Magnuson-Stevens Fishery Conservation and Management Act
nm	nautical miles
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
OCS	Outer Continental Shelf
SDM	species distribution model
ShoalMATE	Shoal Map and Assessment Tool for EFH

1 Executive Summary

1.1 Background

The demand for marine sand resources is increasing in the United States (Drucker et al. 2004), as coastal and offshore sands are commonly used for beach renourishment and barrier island restoration. The dredging of the Outer Continental Shelf (OCS), and sand shoals in particular, is likely to increase in the near future because nearshore sand resources are being depleted while demand increases due to renourishment cycles for beaches, emergency repairs of beaches after storms, and the projected effects of sea-level rise (Nairn et al. 2004). The Bureau of Ocean Energy Management (BOEM), as part of the U.S. Department of the Interior, is responsible for the management and development of energy and mineral resources on the OCS, including marine minerals. Concurrently, 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, the National Oceanic and Atmospheric Administration (NOAA), National Marine Fisheries Service (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. Projects authorized or conducted by the Federal Government must consult with NMFS regarding EFH to ensure full consideration of the environmental effects and possible mitigation measures so that fish and their habitats are not adversely affected. This study follows an comprehensive literature review by Rutecki et al. (2014) on sand shoal geology, geography, and general biological value. Here, we concentrate on the attributes of sand dredging that relate directly to fish and EFH: the potential impacts of dredging on fish, a synthesis of known fish habitat associations, and the development of predictive models to identify the location and types of shoals and distribution of fish species. These new data were then incorporated into an interactive mapping tool "ShoalMATE" that generates semi-automated reports for the EFH consultation process.

1.2 Scope of the Report

The project covers the shallow waters of the OCS for the Atlantic and US Gulf of Mexico. More specifically, the landward boundary of the study area was defined by the Outer Continental Shelf Lands Act (1953), which distinguishes Federal and state jurisdictions (3 nm for all states except 9 nm for Gulf coasts of Florida and Texas). The oceanic boundary of the study area was defined by a 50-m contour line from NOAA's Coastal Relief Model (NOAA National Centers for Environmental Information 2010). Only waters ≤ 50 m deep were included in this study because logistics and costs of dredging in deeper depths and distance from shore generally prohibits dredging projects in deeper waters. 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 report presents the key findings in four volumes as described below.

1.3 Key Findings

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

This volume synthesizes the latest international scientific knowledge on the direct effects of dredging on fish and their habitats, as well as reviews the state-of-science on fish habitat associations. Sand and sediment dredging in the OCS 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. 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. We also synthesized the documented regional habitat associations 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

Volume 2: Shoal Identification and Classification of Sand Resources

This volume comprises two components. First, we develop a predictive model for identifying and delineating potential sand shoals using broadly available, unified digital elevation models for the seafloor along the Gulf of Mexico and Atlantic coastlines from 3 nm from shore to the 50-m depth contour. Seafloor complexity and relief metrics were derived from the Coastal Relief Model and used to predict areas of relative higher relief and to produce polygons showing geomorphological features consistent with sand shoals, ridges, and swales. Maps depict bedforms, shoal complexes, and yet-unclassified features along the Gulf and Atlantic coastlines. Recognition of these features in the context of EFH and sand resource demand will aide in improved planning and permitting for sand dredging activities. We then classify these features and sand shoals according to a new scheme developed by subject matter experts in the fields of geology, biology, and seafloor habitats during three facilitated workshops and webinars. This new classification scheme is proposed for adoption under the Coastal Marine Ecological Classification Standard (CMECS) as a new schema for classifying OCS sand features.

Volume 3: Predicting the Distribution of Select Fish Species of the Gulf of Mexico, South Atlantic, and Greater Atlantic

This volume presents the scientific analysis and results of predictive modeling for select federally managed fish. Species distribution models (SDMs) are a state-of-the-art statistical modeling approach that quantifies the relationships between species and spatially explicit environmental data. SDMs work by extending the identified species-habitat relationships to the entire distribution of species under consideration. These predictive modeling results are ideal to inform management decisions. In this volume, we used a variety of fisheries-independent data sources in the Gulf of Mexico and South Atlantic to produce SDMs for select marine fish and shrimp species. Environmental data on habitats included oceanographic conditions, geomorphology, geography, prey, and the nearby ecosystems of wetlands and estuaries. For the Greater Atlantic, we summarize SDMs developed by the Northeast Fisheries Science Center which combined trawl surveys with data on oceanographic conditions, substrate, and zooplankton. Together, these maps and quantified habitat relationships (or lack thereof) add to the information synthesized in Volume 1, "Fish Habitat Associations and the Potential Effects of Dredging on the Atlantic and Gulf of Mexico Outer Continental Shelf." The analyses evaluated the best habitat predictors of marine species and depicted the distribution of select marine fish and shrimp species. Species' relationships with geomorphology characteristics were limited and of minor importance compared to other habitat predictors. None of the Gulf of Mexico species examined were related to bottom currents, slope, or heterogeneity of depth. Of minor importance in the models, white shrimp had a higher catch per unit effort (CPUE) farther away from shoals, and pink shrimp were positively related to sand grain sizes. Red snapper age-0 had a higher CPUE in close proximity to shoals and where the bathymetric position index predominately showed a hill topography. In the South Atlantic, none of the five species examined were associated with geomorphology characteristics. Overall, species' distributions were primarily related to oceanographic conditions, nearby wetlands and estuaries, and prey species. When applicable, geomorphology predictors only had minor influence on species' distribution.

Volume 4: Development of ShoalMATE: Shoal Map Assessment Tool for Essential Fish Habitat

This volume outlines the process and framework used to develop the interactive mapping tool, user interface, and automated reporting of ShoalMATE (Shoal Map and Assessment Tool for EFH). ShoalMATE was conceived as a standardized reporting tool to facilitate better communication between BOEM and NOAA during EFH assessments required for dredging projects on the OCS. Development initiated by gathering requirements from BOEM’s Marine Minerals Program and NOAA’s Habitat Conservation Division. A database architecture and workflow were proposed to meet the needs of access and usability for stakeholders with varying levels of familiarity with Geographic Information Systems. We ran the data necessary to support the tool (e.g., habitat descriptors, species models, project boundaries) through a series of custom scripts that store information describing each identified shoal in a database that was specifically designed for expedited queries within the front-end application. The front-end application presents this queried information within a web browser and generates a template report as a Microsoft Word document, which can be edited by analysts to create a final, consistent product across a range of geographies and project extents.

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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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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

January 2020

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DISCLAIMER

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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).

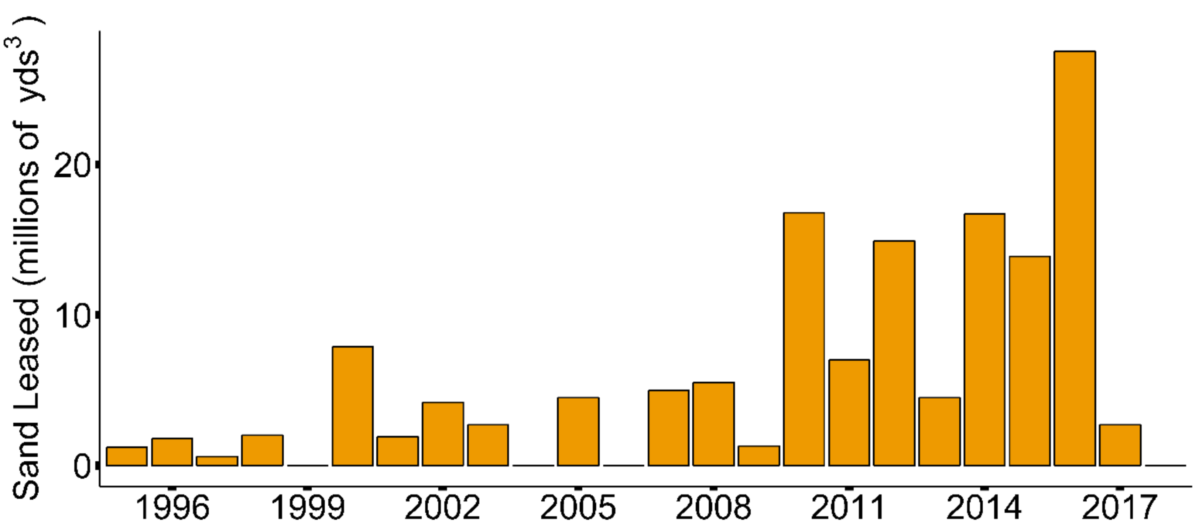

Figure 1-1. Trend in quantity of sand leased in Federal waters.
Source: BOEM Marine Minerals Information Systems sand lease data and https://www.boem.gov/MMP-in-Your-State/, accessed 2019 March.

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.

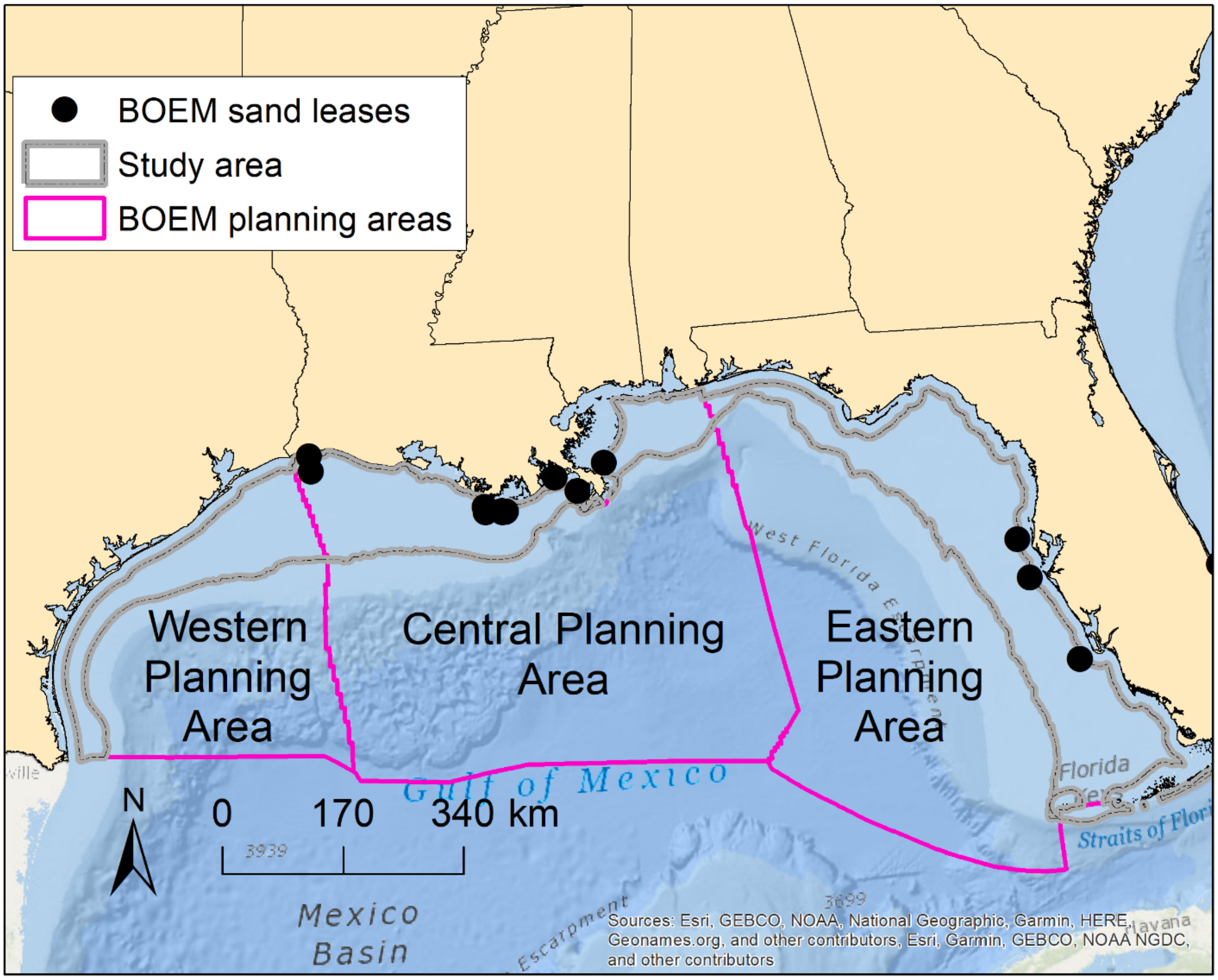

Figure 1-2. The GoM study area defined by the landward boundary of Federal waters and the seaward boundary of the 50-m depth contour line.
Sand lease areas, as of the 2016 fiscal year, are shown as centroids for reference (not to scale).

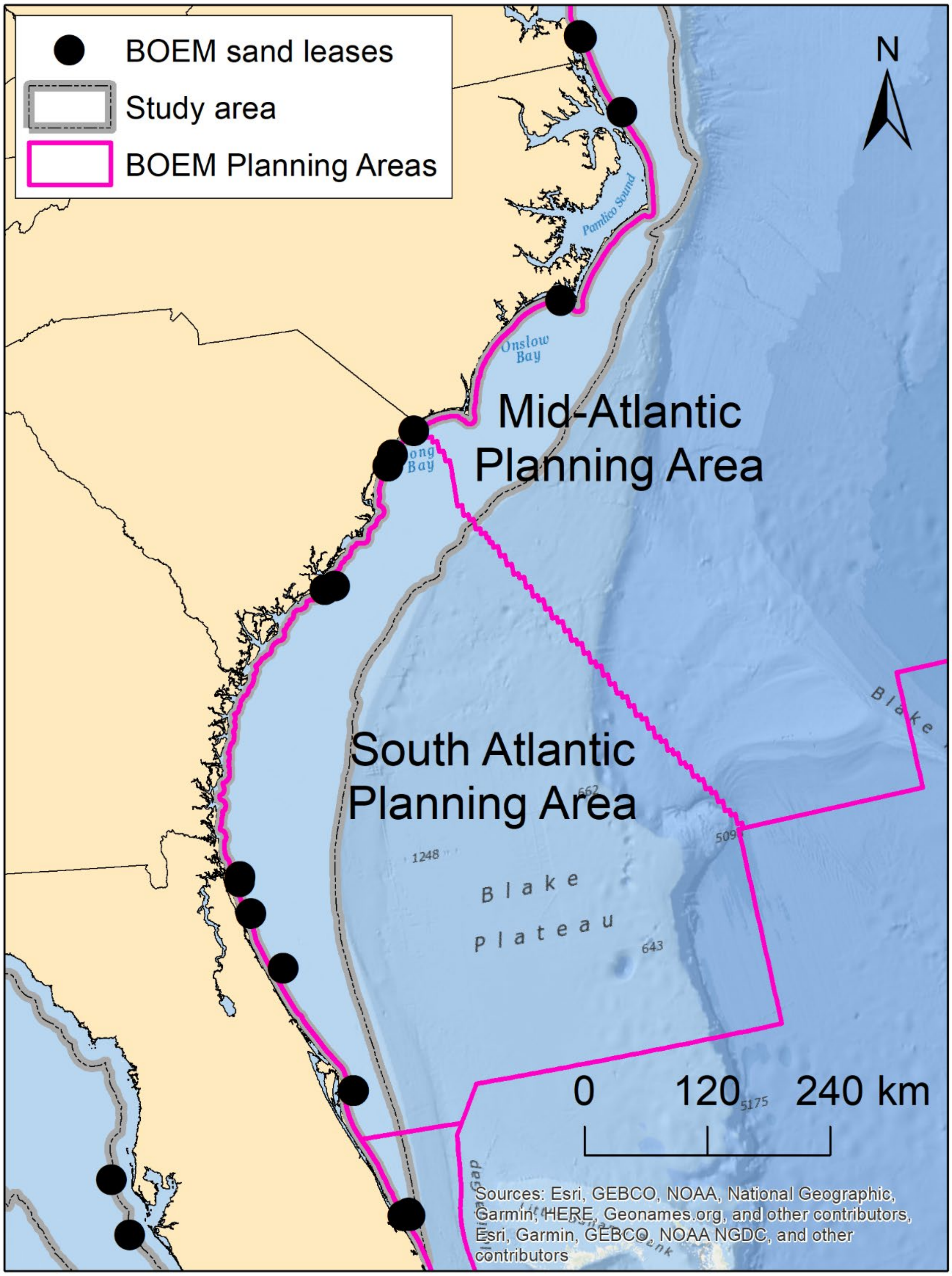

Figure 1-3. The South Atlantic study area defined by the landward boundary of Federal waters and the seaward boundary of the 50-m depth contour line.
Sand lease areas, as of the 2016 fiscal year, are shown as centroids for reference (not to scale).

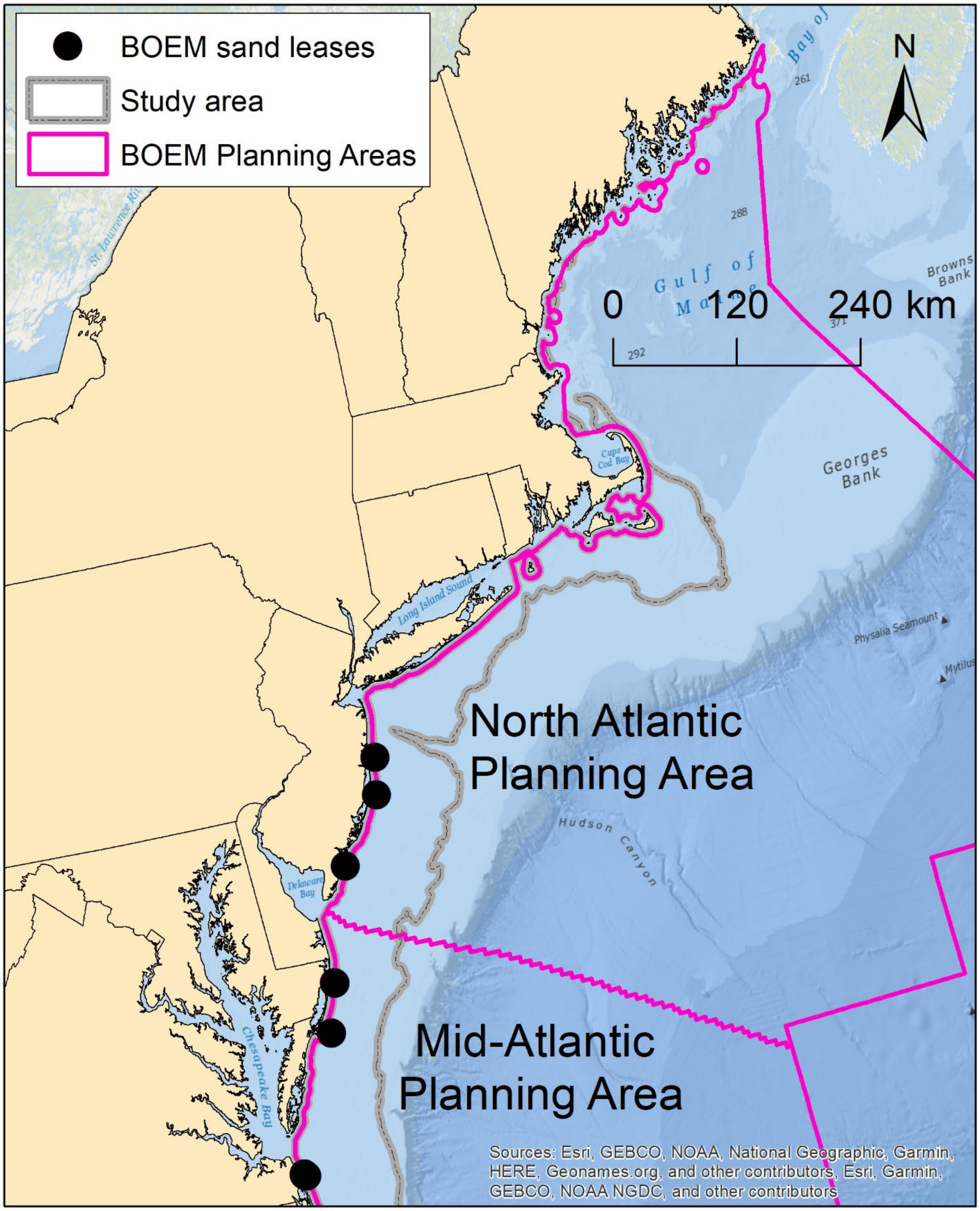

Figure 1-4. The northeast Atlantic study area defined by the landward boundary of Federal waters and the seaward boundary of the 50-m depth contour line.
Sand lease areas, as of the 2016 fiscal year, are shown as centroids for reference (not to scale).

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).

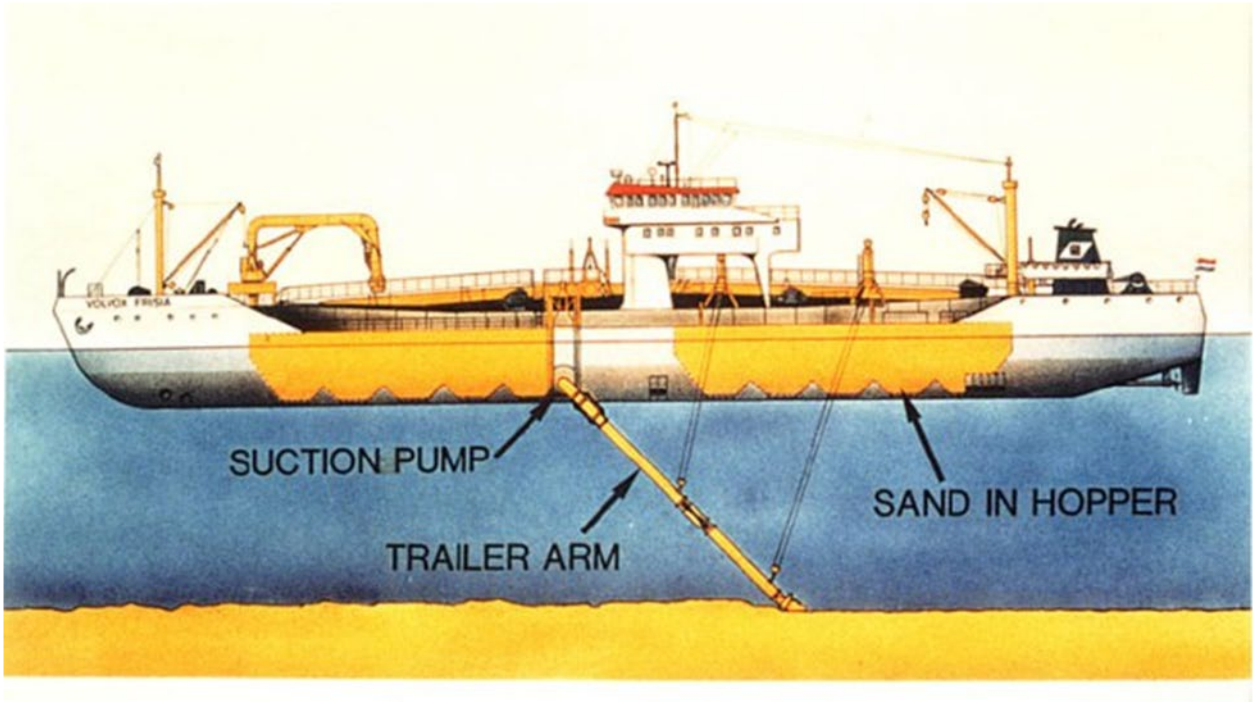

Figure 2-1. Main components of a TSHD. Additionally, the draghead is attached to the bottom of the trailer arm and is where the hydraulic suction is generated.
Source: Adapted from https://oceanandairtechnology.wordpress.com/2013/06/11/trailing-suction-hopper-dredger/

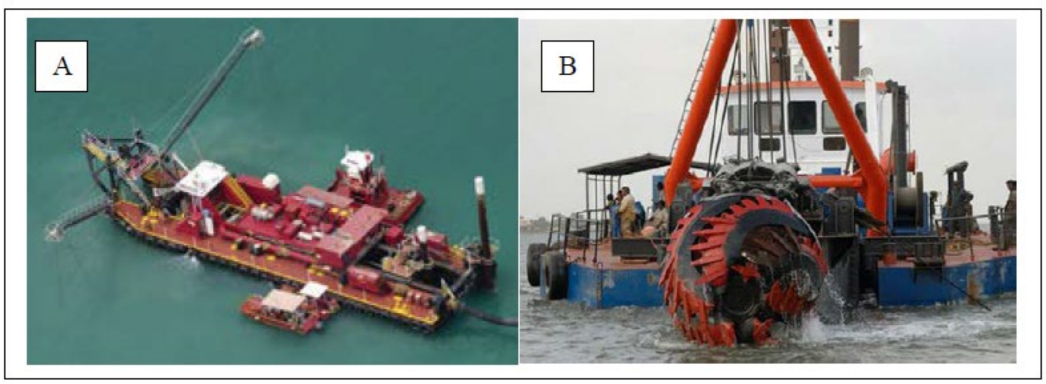

Figure 2-2. A) Cutterhead suction dredge and B) cutterhead.
Source: Adapted from Michel (2013) and www.dredgepoint.org.

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**).

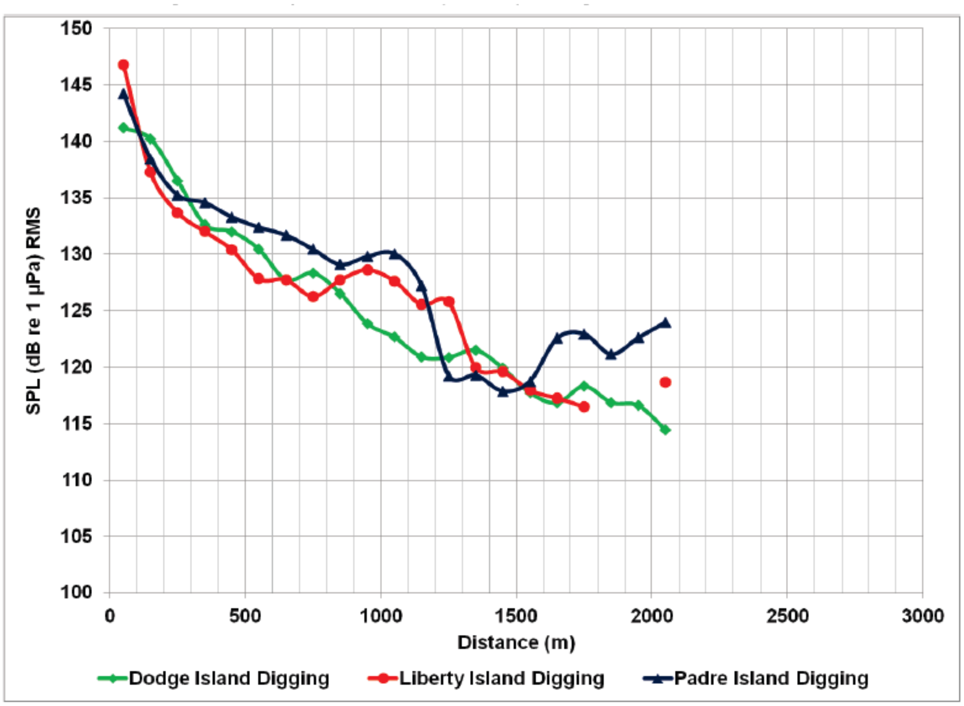

Figure 2-3. An example of underwater sound emitted, as measured in sound pressure levels (SPL) (y-axis), at incremental distances (x-axis) from three trailing hopper suction dredges operating in offshore marine waters of the Atlantic Ocean near Wallops Island, Virginia.
Source: Image from Reine et al.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

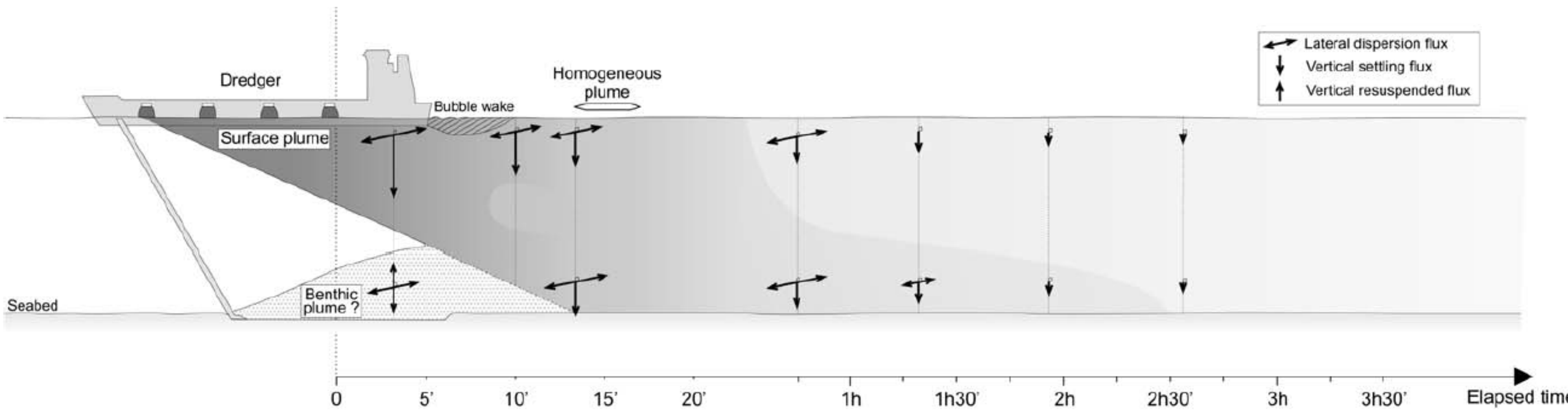

Figure 2-4. A conceptual model of suspended sediment movement from a dredging operation over time.
Source: Adapted from Duclos et al. (2013). Reproduced with permission from the Coastal Education and Research Foundation, Inc.

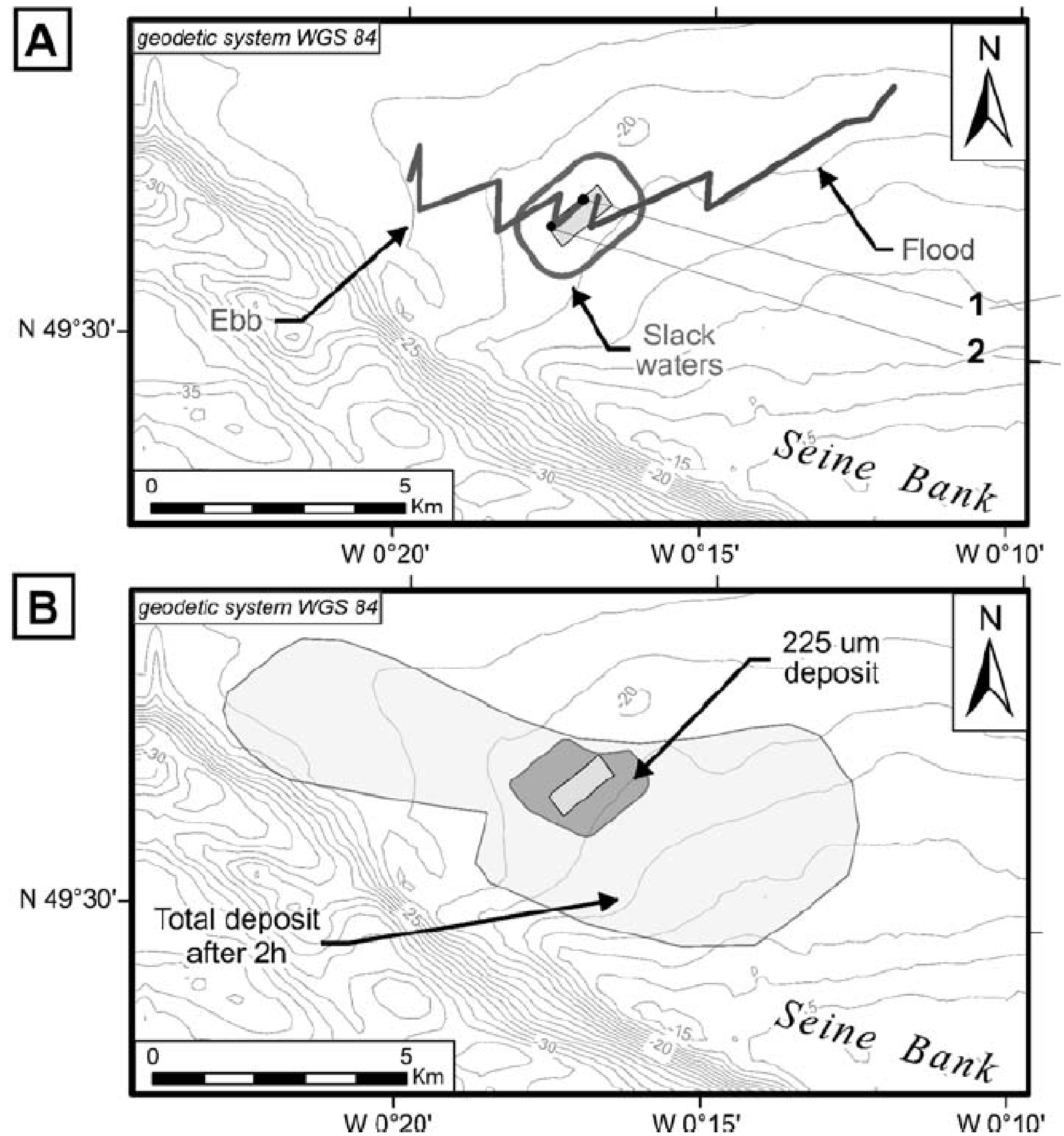

Figure 2-5. The extent of a subsurface sediment plume and sedimentation resulting from aggregate dredging in the English Channel, UK. A) sediment plume extent and direction during ebb, flood, and slack tidal phases of a mean tide; B) sediment deposition two hours after dredging.

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 ≥ 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.

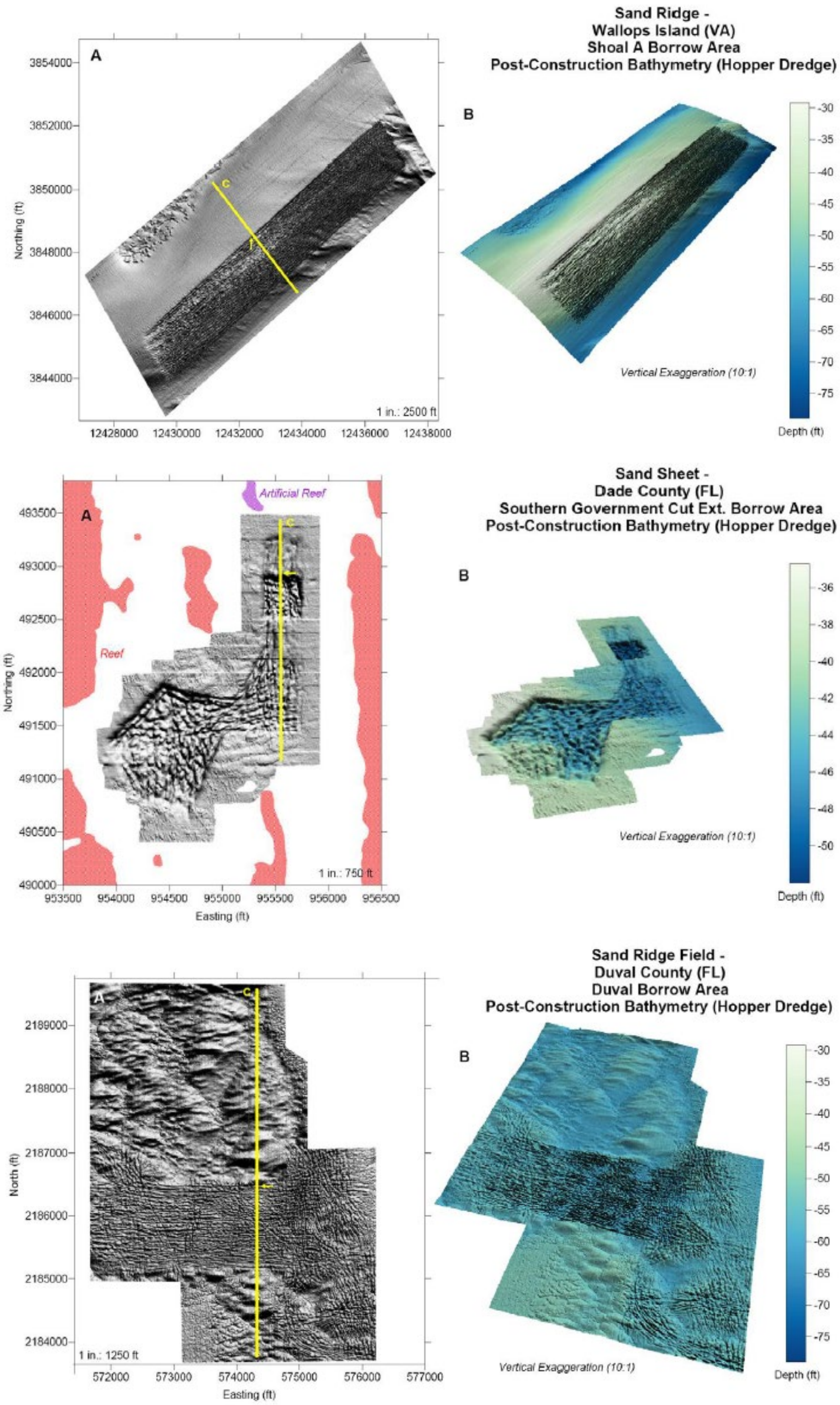

Figure 2-6. Digital elevation models of post-dredging bathymetry for representative OCS borrow areas.
Source: Figure modified from Michel et al. (2013). The pink color represents hard bottom habitat for Dade County, Florida, USA.

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).

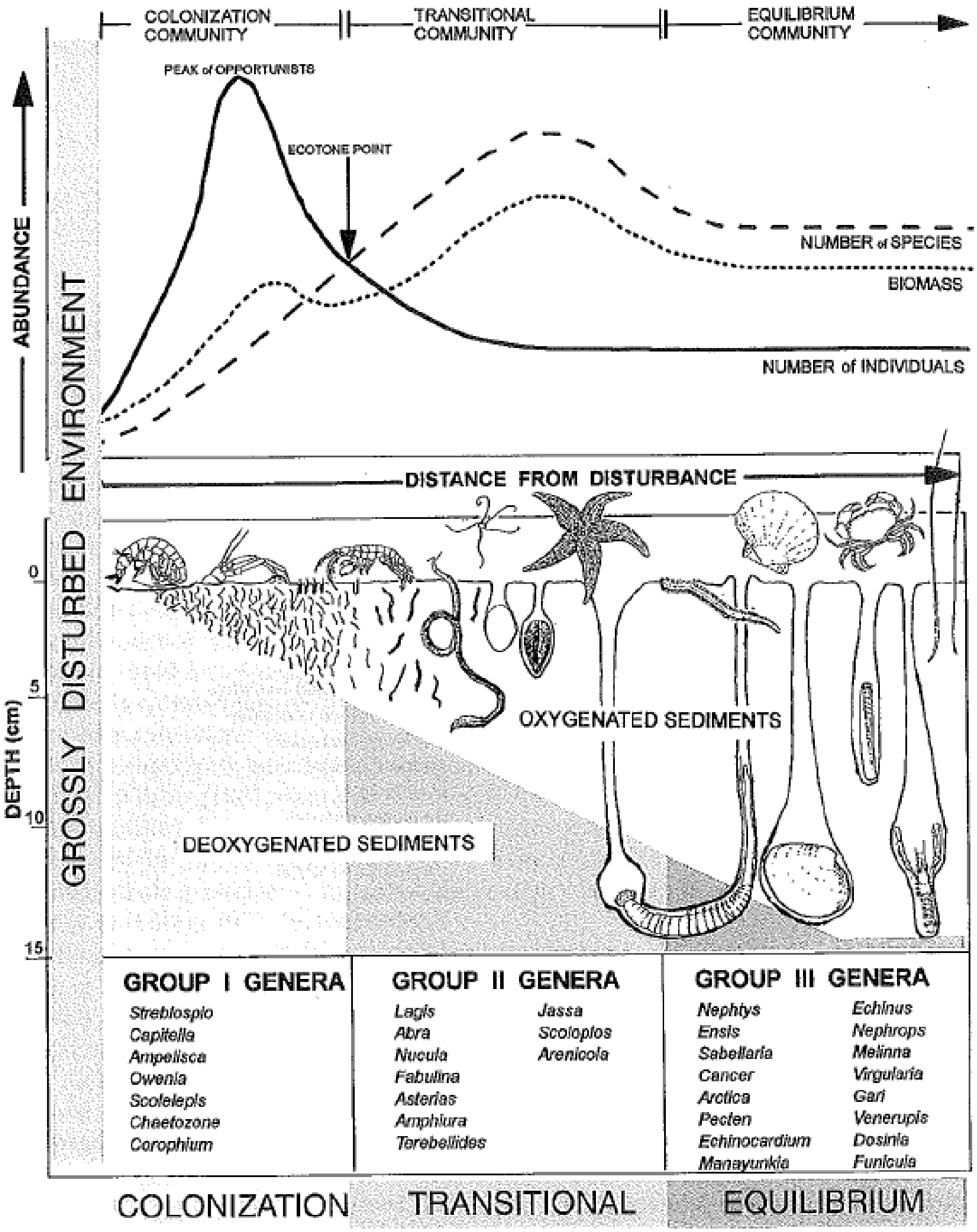

Figure 2-7. Conceptual diagram of ecological succession of benthic communities.
Source: Adapted from Newell et al. (1998) and based on Pearson & Rosenberg (1978) and Rhoads et al. (1978). Reproduced with permission from the Taylor and Francis.

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 $(\text{height/base depth})_{max}$ = (base depth-5)/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).

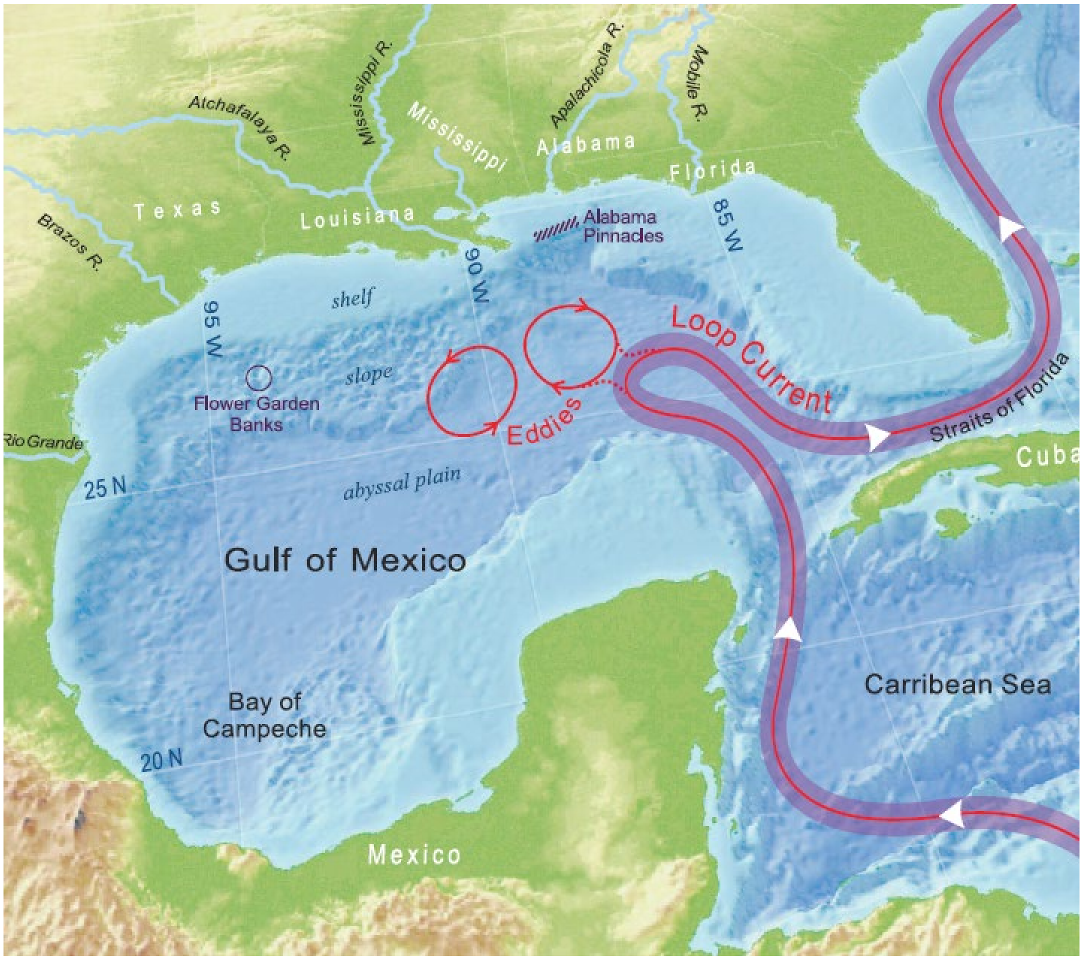

Figure 3-1. GoM's Loop Current, spin-off eddies, major rivers, and select geological features.
Source: Adapted from Spies et al. (2016). Reproduced with permission from the University of Southern Mississippi, Gulf and Caribbean Research.

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.

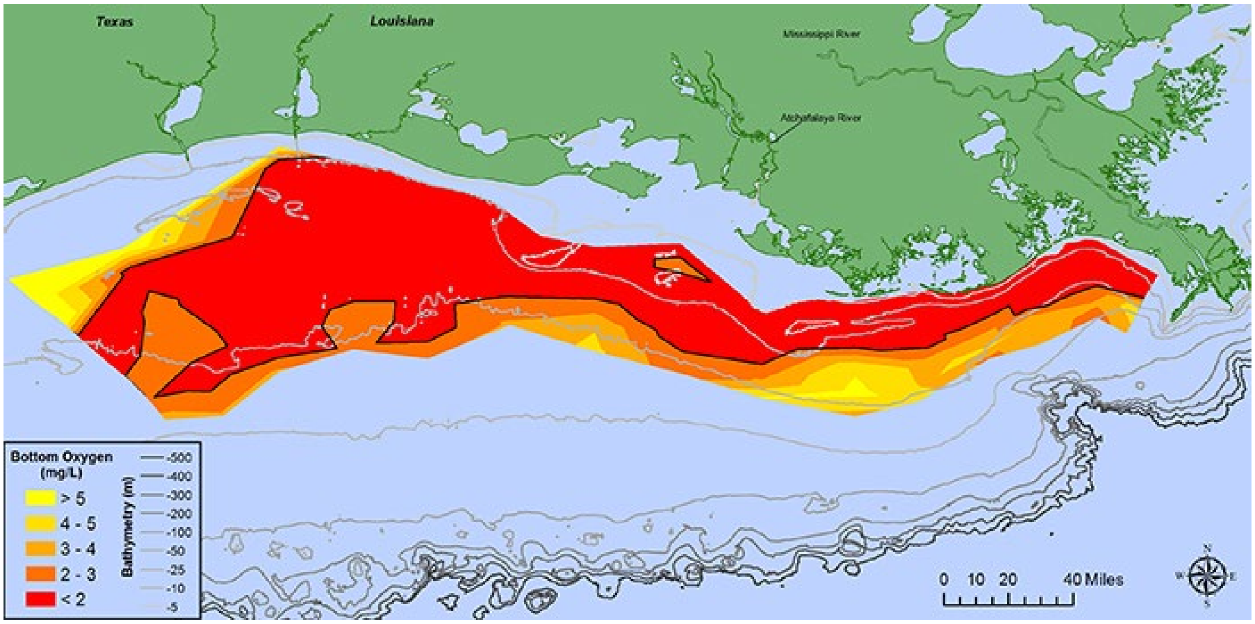

Figure 3-2. Map of GoM hypoxic zone in July 2017. The black line shows the area where DO is < 2 mg L-1, which is considered the threshold for hypoxia.
Source: NOAA, https://www.noaa.gov/media-release/gulf-of-mexico-dead-zone-is-largest-ever-measured (accessed 2019 Aug 8) and is courtesy of N. Rabalais,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

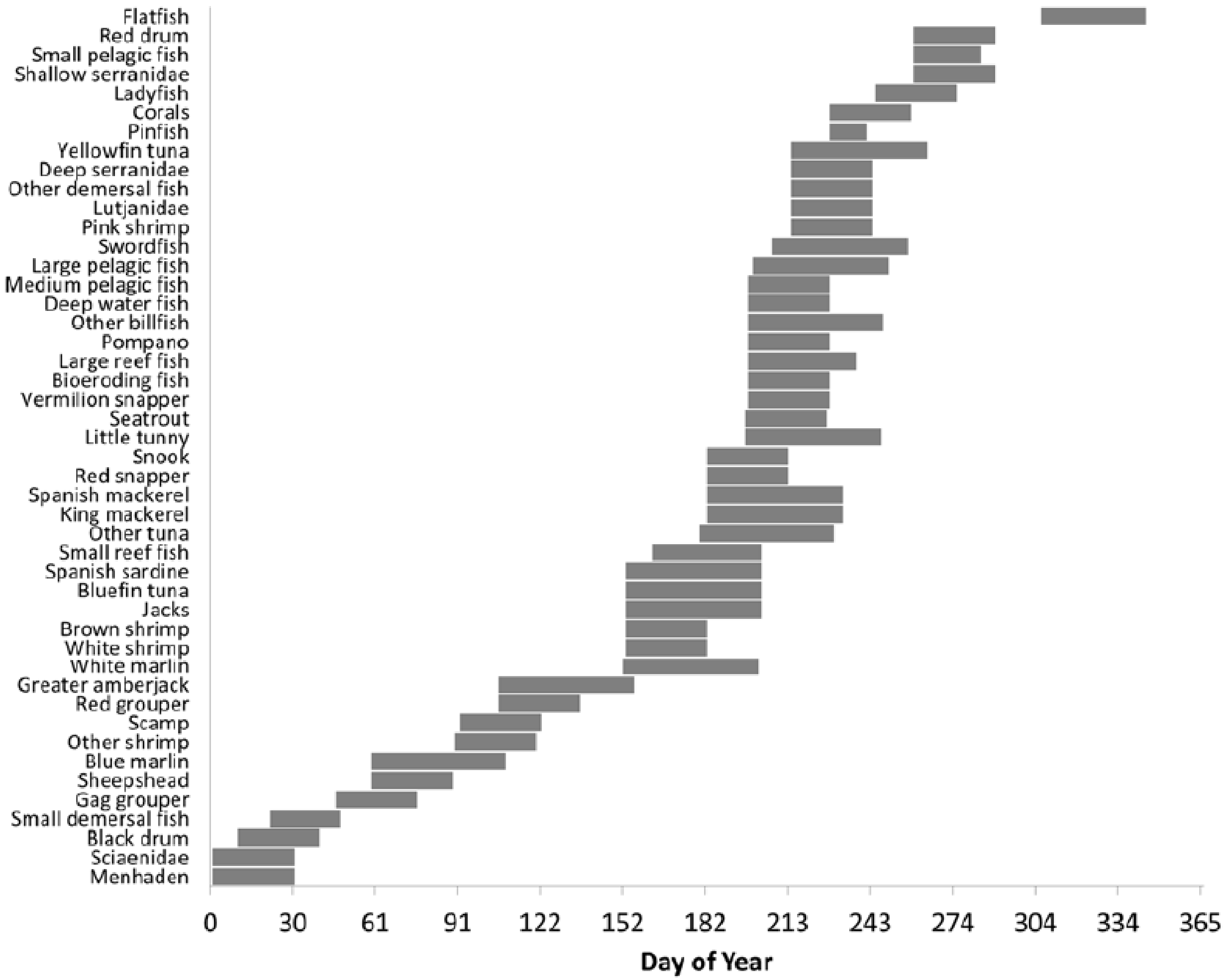

Figure 3-3. Seasonality of GoM spawning seasons.
Source: Adapted from Ainsworth et al. (2015). Reproduced with permission from the Creative Commons Attribution License.

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, 5^{th} and 95^{th} 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.

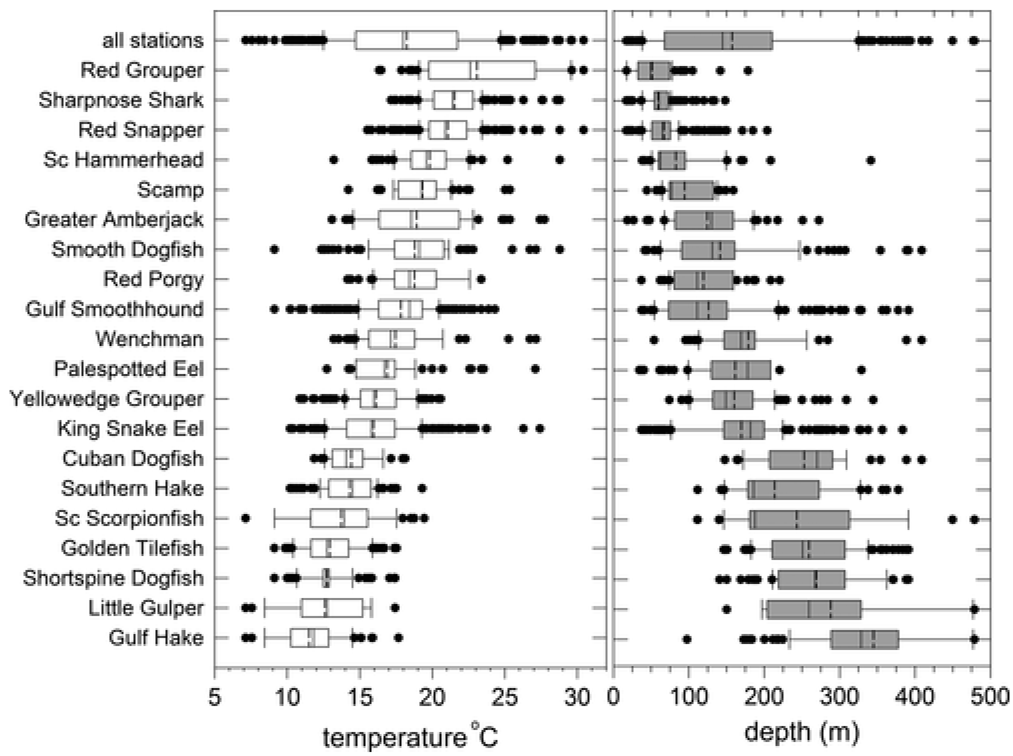

Figure 3-4. GoM fish habitat associations with water temperature and depth.
Source: Adapted from Murawski et al. (2018) with open access permissions.

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).

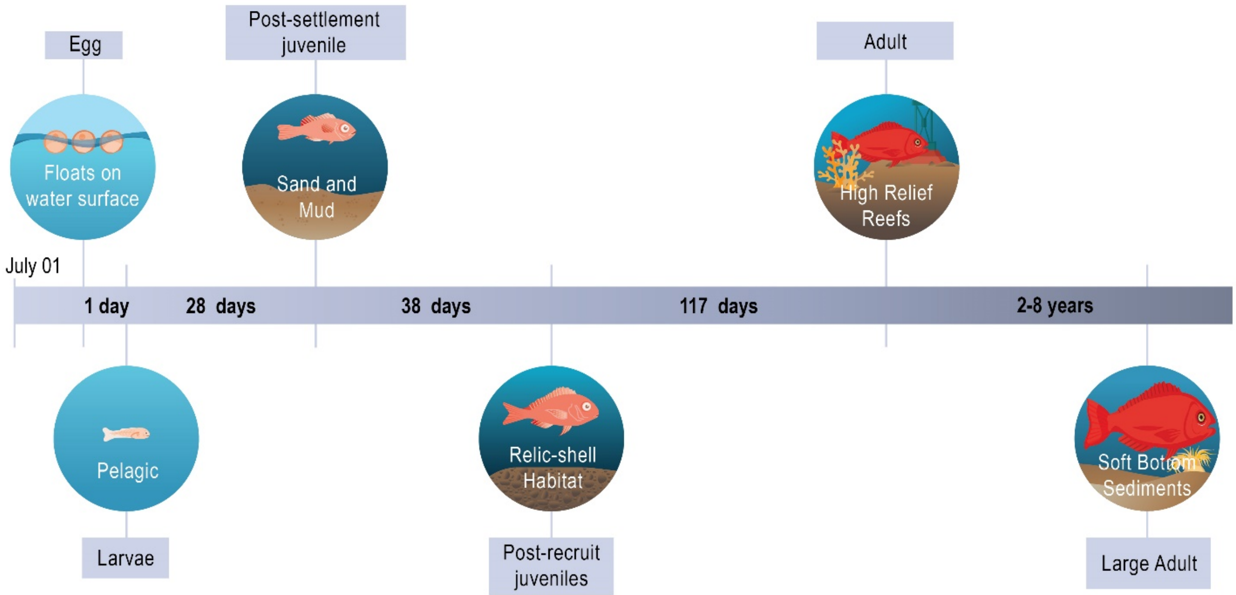

Figure 3-5. Summary of red snapper life stages and broad description of habitats used. Dates for larvae and juvenile stages depict an example with an egg hatching July 1st.

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 ≥ 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℃ (Froehlich et al. 2019). Emigration from reefs has also been reported following cold fronts elsewhere and fish returned when waters were > 20℃ (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**).

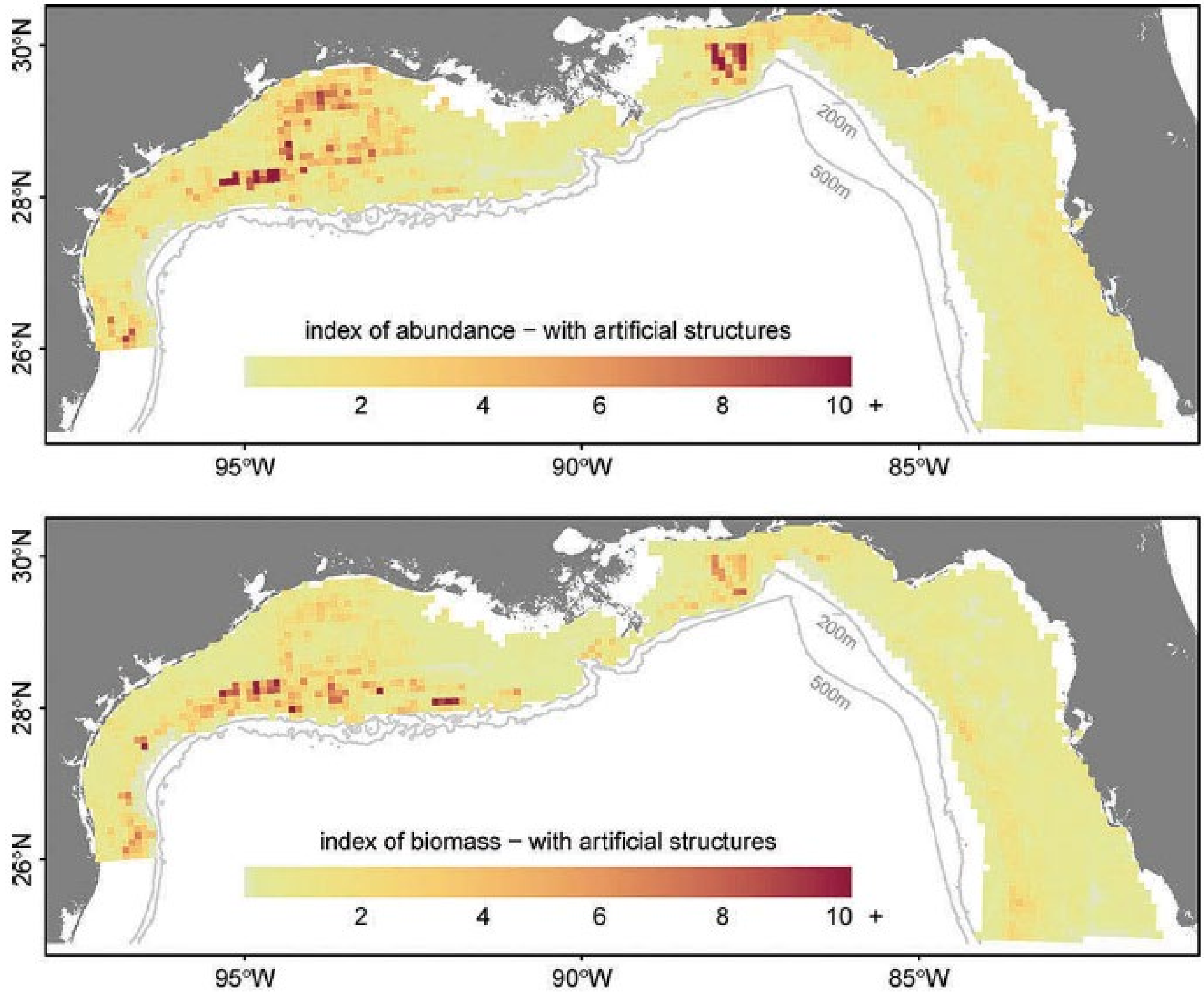

Figure 3-6. Predicted red snapper abundance and biomass for the GoM based on models of abundance accounting for artificial structures.
Source: Adapted from Karnauskas et al. (2017). Reproduced with permission from the Creative Commons Attribution License.

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.

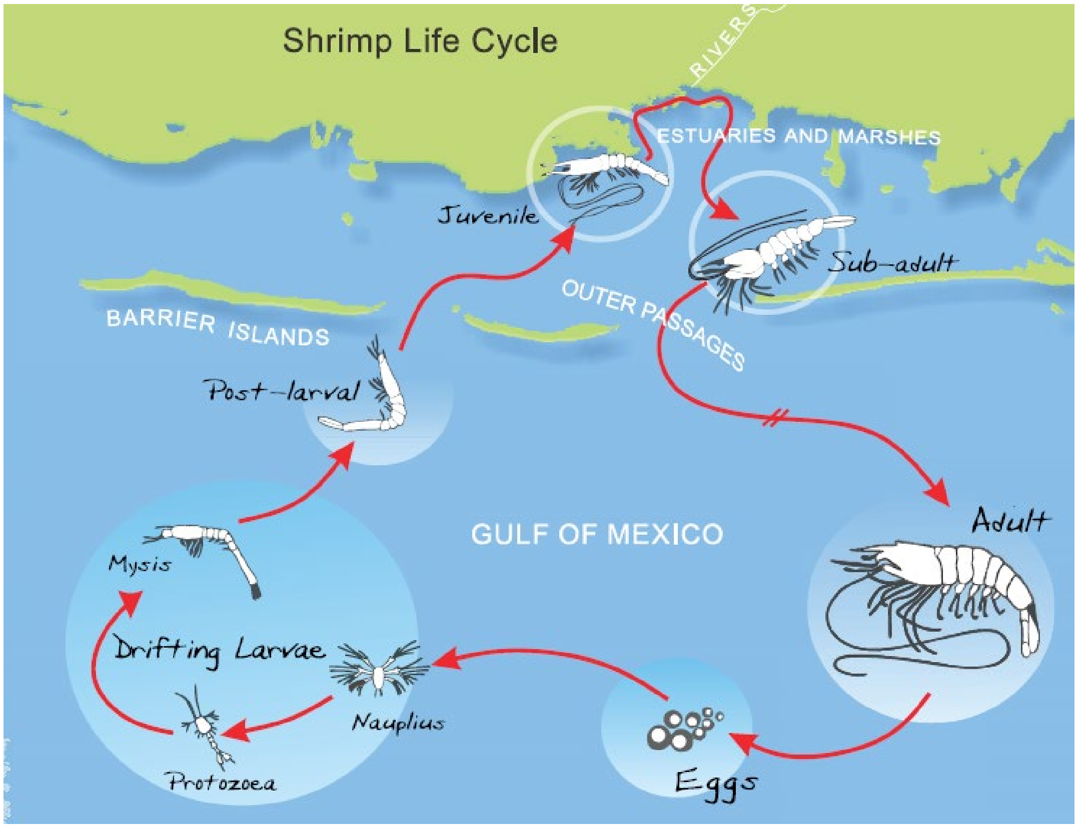

Figure 3-7. Life cycle of brown and white shrimp in the GoM.
Source**: A**dapted from Spies et al. (2016) and diagram by J.R. Allen. Reproduced with permission from the University of Southern Mississippi journal, *Gulf and Caribbean Research*.

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).

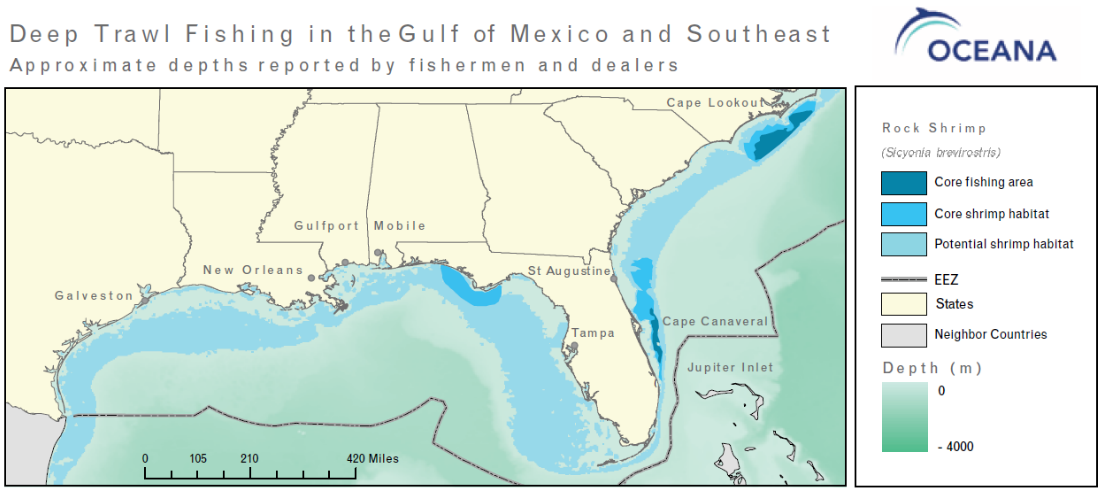

Figure 3-8. Potential distribution of rock shrimp based on interviews, FMC documents, and scientific literature where available.
Source: Adapted from Stiles et al. (2007) with open access permissions.

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.

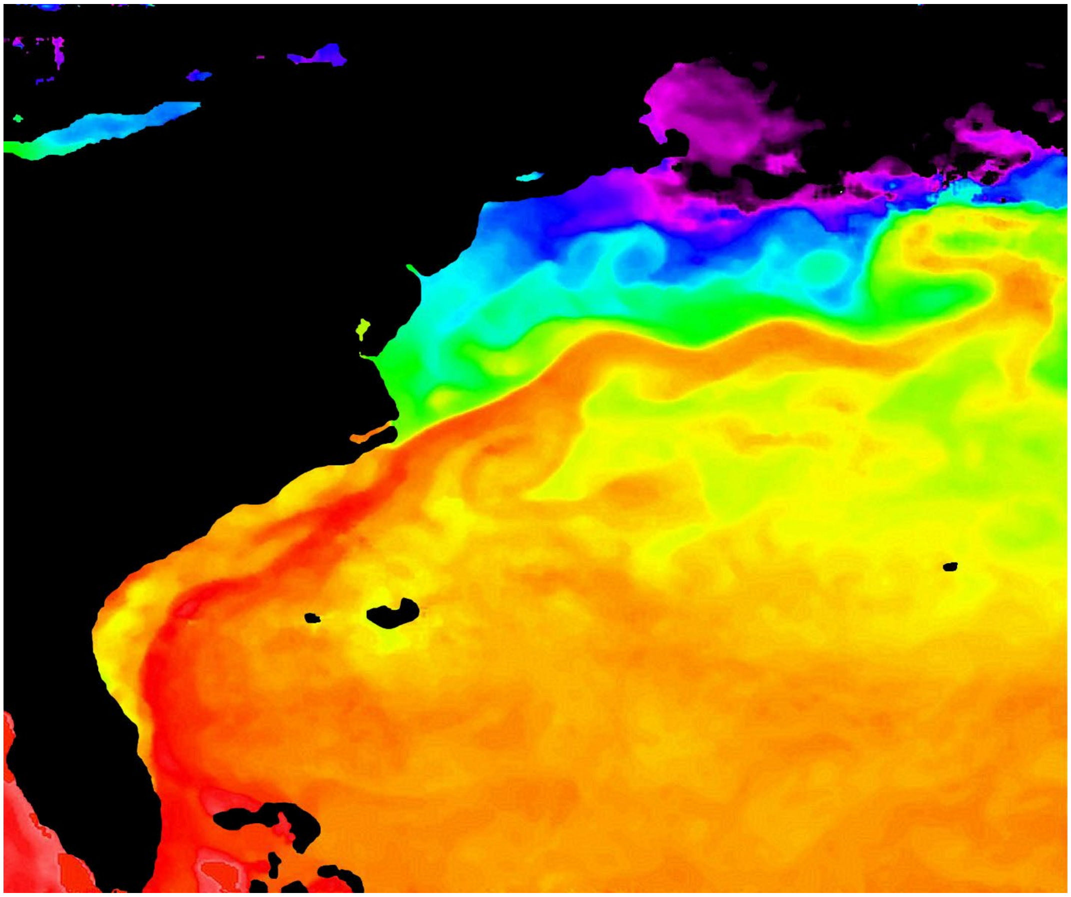

Figure 3-9. The Gulf Stream current depicted in red moving from southern Florida toward the northeast.
Source: Adapted from NASA Jet Propulsion Laboratory.

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.

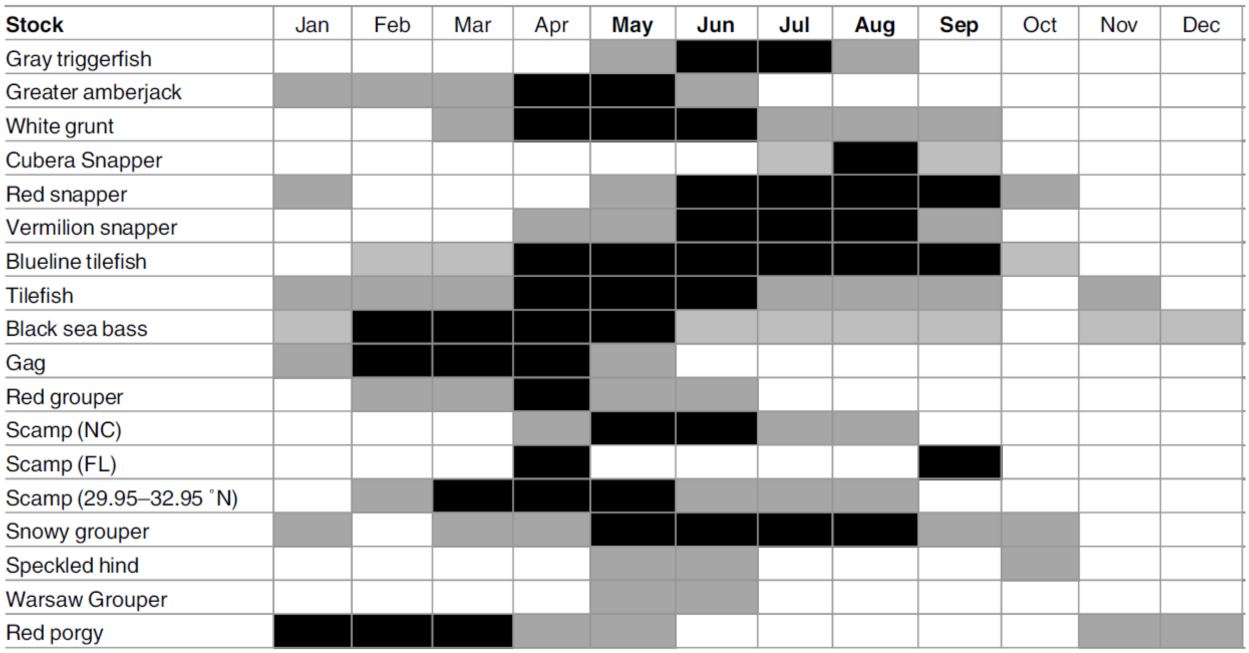

Figure 3-10. Timing of spawning (gray) and peak spawning (black) for selected species in the snapper-group EFH group of the southeastern Atlantic Ocean of the US.
Source: Adapted from Farmer et al. (2017). Reproduced with open access permission from PlosOne.

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 ≤ 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).

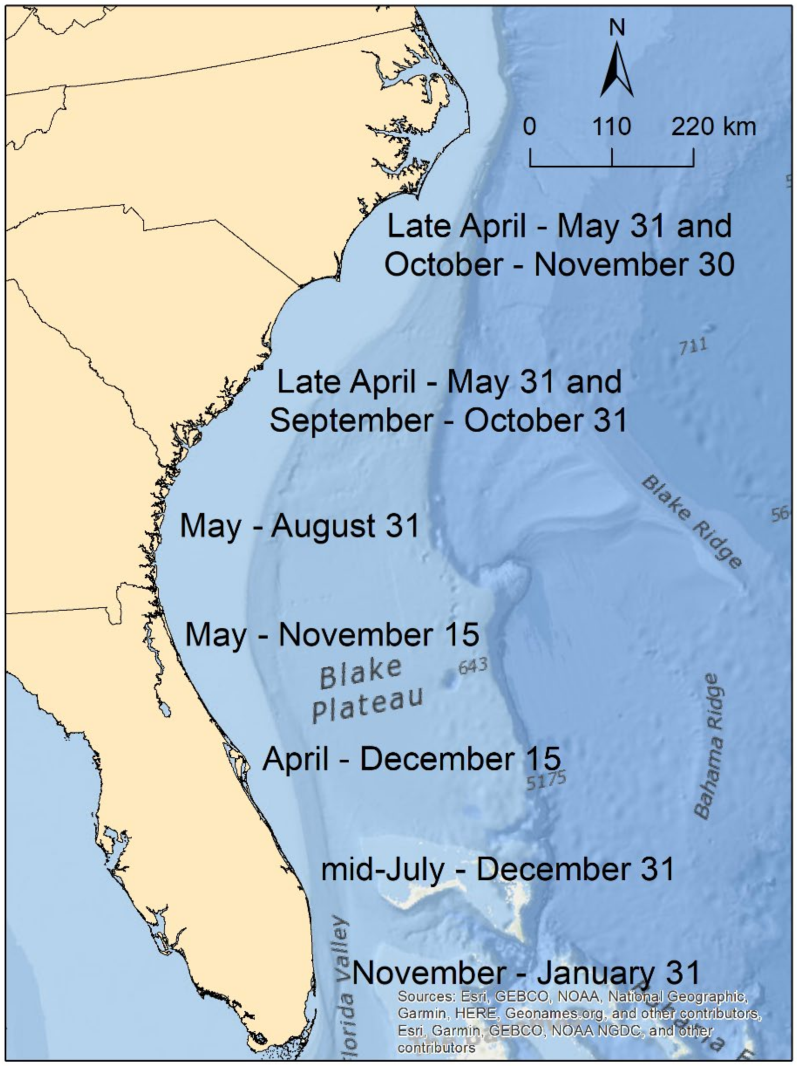

Figure 3-11. King mackerel seasonality with dates matching to approximate locations depicted by Trent et al. (1987).

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.

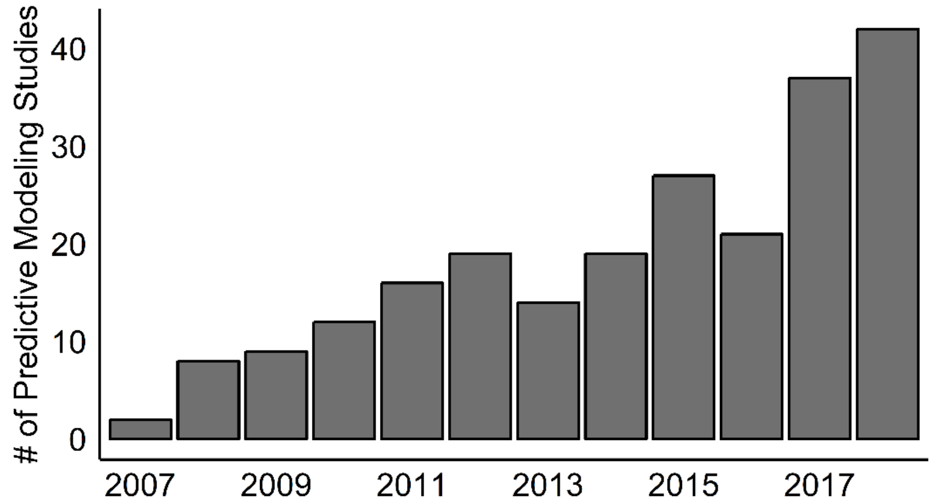

Figure 4-1. Number of peer-reviewed marine fish spatially explicit predictive model studies 2007–2018.

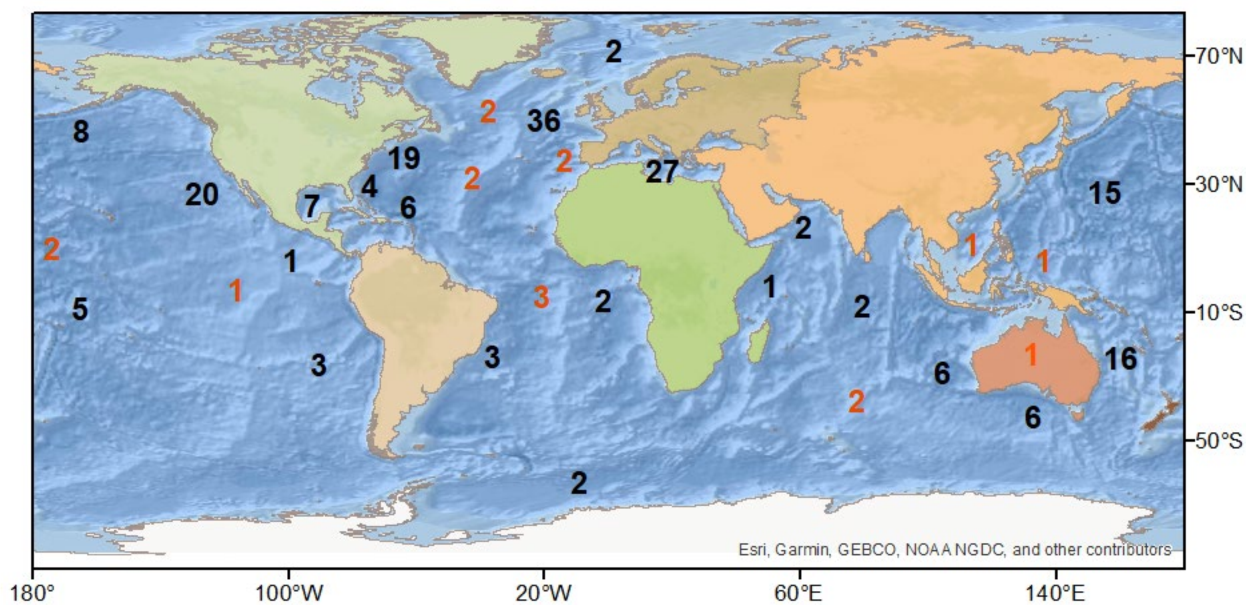

Figure 4-2. Geography of studies that predicted the distribution of marine fish (2007–2018) (n=226).

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.

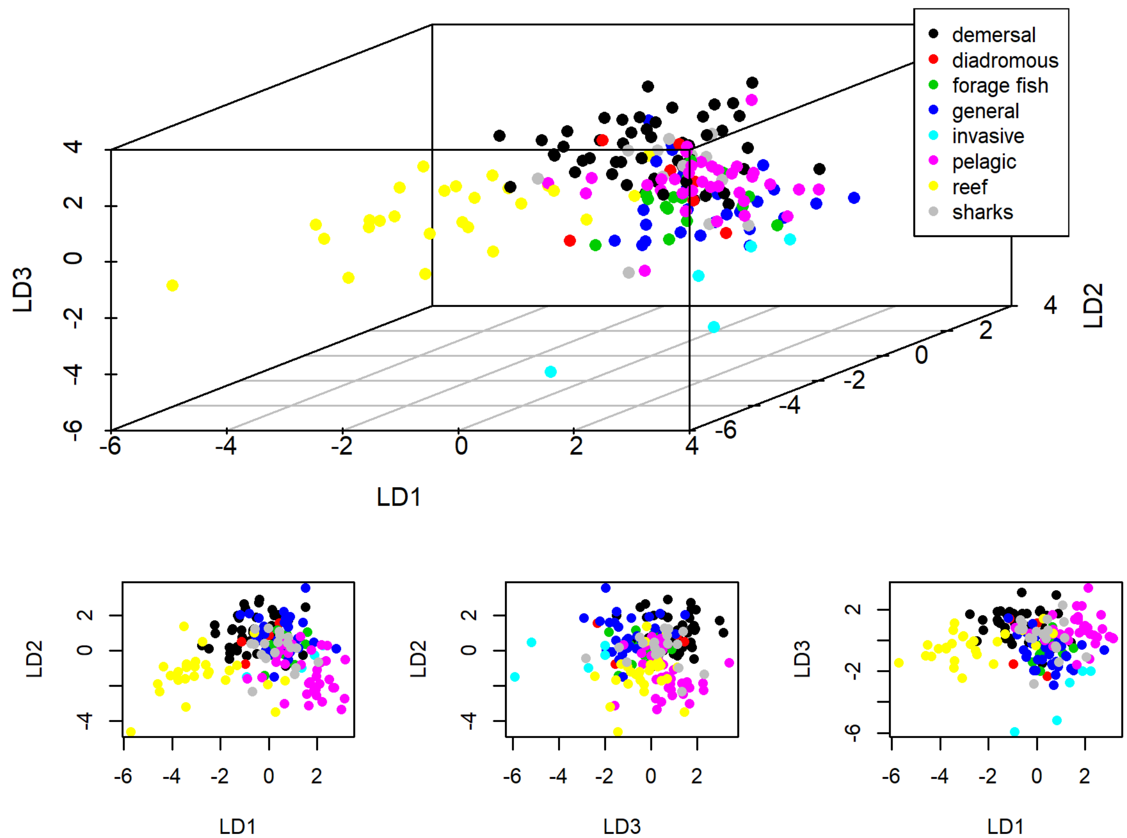

Figure 4-3. LDA results that show habitat variables tested in predictive modeling studies differ with fish functional groups.

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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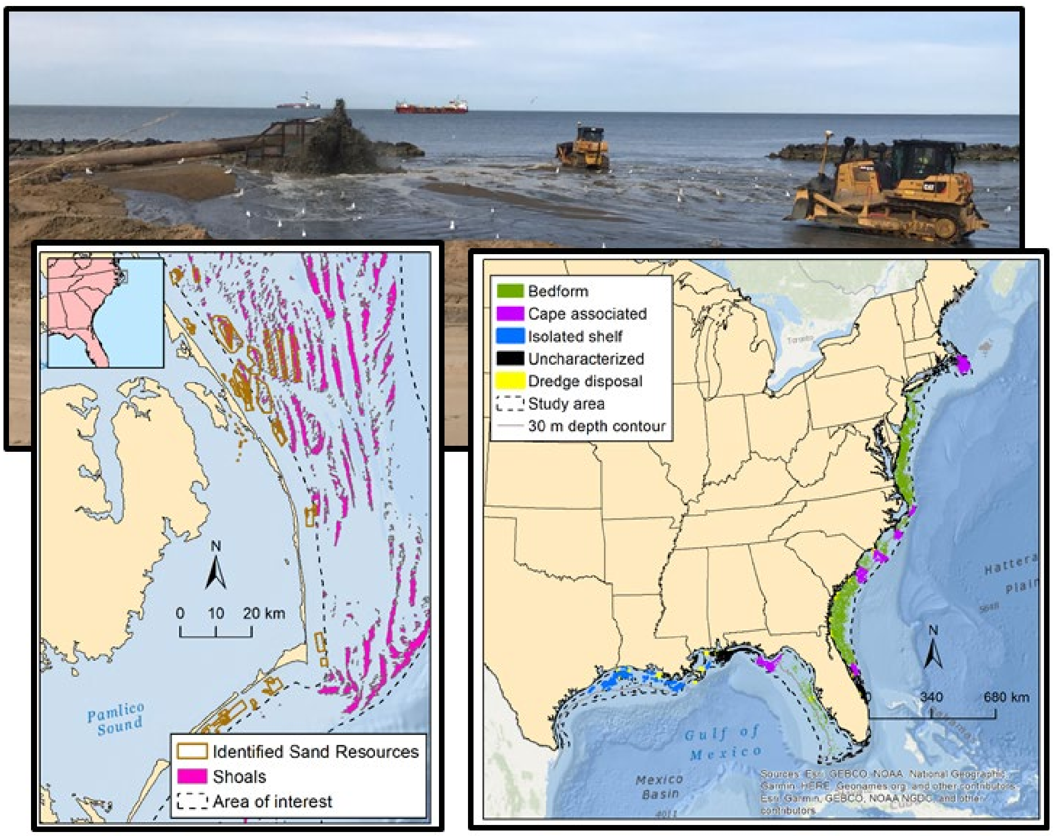
NOAA NCCOS 270

Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features

Volume 2: Shoal Identification and Classification of Sand Resources



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Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features

Volume 2: Shoal Identification and Classification of Sand Resources

January 2020

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DISCLAIMER

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To download a PDF file of this report, go to the US Department of the Interior, Bureau of Ocean Energy Management Data and Information Systems webpage (http://www.boem.gov/Environmental-Studies-EnvData/), click on the link for the Environmental Studies Program Information System (ESPIS), and search on 2019-2020. The report is also available at the National Technical Reports Library at https://ntrl.ntis.gov/NTRL/.

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List of Abbreviations and Acronyms

BC	Biotic Component
BGN	Board on Geographic Names
BOEM	Bureau of Ocean Energy Management
BPI	bathymetric position index
CATAMI	Collaborative and Automated Tools for Analysis of Marine Imagery
CMECS	Coastal Marine Ecological Classification Standard
CRM	Coastal Relief Model
cy	cubic yard(s)
DEM	digital elevation model
DOC	Department of Commerce
DOI	Department of the Interior
EFH	Essential Fish Habitat
GC	Geoform Component
GMFMC	Gulf of Mexico Marine Fisheries Council
GNS	GEOnet Names Server
ha	hectare
HAPC	Habitat Areas of Particular Concern
km	kilometer(s)
m	meter(s)
mm	millimeter(s)
MMIS	Marine Minerals Information System
NGA	National Geospatial-Intelligence Agency
NGDC	National Geophysical Data Center
NCCOS	National Centers for Coastal Ocean Science
nm	nautical mile(s)
NOAA	National Oceanic and Atmospheric Administration
OCS	Outer Continental Shelf
SC	Substrate Component
ShoalMATE	Shoal Mapping Assessment Tool for EFH
SME	subject matter expert
TNC	The Nature Conservancy
US	US Geological Survey
US EPA	US Environmental Protection Agency
USACE	US Army Corps of Engineers
USGS	US Geological Survey

Abstract

The demand is increasing in the United States for marine sand resources to mitigate risks of coastal erosion from modifications to the shoreline due to coastal development or impacts of waves and currents during major storms. As sources for sediment in the nearshore are being depleted, there is increased demand for sand resources from the Outer Continental Shelf (OCS). Extensive studies have helped elucidate the dynamics and geomorphological characteristics of some prominent shoals, but we lack a broader understanding of the extent and types of shoals that exist along the Gulf and Atlantic Coasts of the OCS. Furthermore, offshore sand features have typically been characterized in a geological framework, and we lack descriptors for sand features that contribute to Essential Fish Habitat (EFH) descriptions. This volume comprises two components. First, we developed a seabed classification model that identifies and delineates potential sand shoals using broadly available, unified digital elevation models for the seafloor along the Gulf of Mexico and Atlantic coastlines from 3 nautical miles (nm) from shore to the 50-m depth contour. Distance from shore as well as seafloor complexity and relief metrics, derived from the Coastal Relief Model, were used to classify offshore areas of relative higher relief and produce polygons showing geomorphological features consistent with sand shoals, ridges, and swales. Newly created maps depict bedforms, shoal complexes, and unclassified features along the Gulf and Atlantic OCS. Recognition of these features in the context of EFH and sand resource demand can aide in improved planning and conservation recommendations for sand dredging activities. Three workshops were hosted with biological and geological subject matter experts to define criteria for classifying shoals and other sand features. We then used this new classification scheme to attribute sand shoals according to characteristics such as geoforms and environmental criteria to develop a new scheme. This new classification scheme was formalized under the Coastal Marine Ecological Classification Standard (CMECS) and proposed for adoption as a new schema for classifying OCS shoals.

1 Introduction

The demand for marine sand resources is increasing in the United States (Drucker et al. 2004) and worldwide (Charlier and Charlier 1992; de Jong et al. 2014; Kim et al. 2008; La Porta et al. 2009). The Netherlands alone uses an estimated 24 million m^3 (31 million cy) of dredged sand annually, and the amount is expected to grow rapidly with rising sea level (de Jong et al. 2014). In the United States, coastal and offshore marine sands are commonly used for beach renourishment, barrier island restoration, and wetland restoration. As human populations and development expands along the coastline, erosion will continue to bring challenges to ensuring sustainability in the coastal zone. The restoration and maintenance of beaches, barrier islands, and other coastal infrastructure will require substantial sediment resources. Although prior sand mining activities have focused on nearshore sand sources (i.e., in state waters), the dredging of Outer Continental Shelf (OCS) sand resources is likely to increase in the near future as nearshore sand resources are depleted (Nairn et al. 2004). 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.

Sand resource mapping in offshore waters has been ongoing for decades, and cooperative agreements have been established between Federal and state governments to synthesize existing and new information, including the mapping of sand shoals appropriate for sand mining (Drucker et al. 2004). Marine sand dredging occurs in relatively shallow waters (≤ 50 m), often within ridge and swale complexes where large volumes of sand can be extracted over relatively small areas. As of 2019, all 17 Atlantic and Gulf of Mexico coastal states have cooperative agreements with BOEM to identify available sand resources. As of 2017, there were no BOEM sand and gravel leases in New England and New York, 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. The overall strong upward trend of OCS sand dredging necessitates a greater strategic vision for managing sand resources as a whole rather than the site-by-site approach that has been undertaken to this point. Current geophysical and geotechnical survey and other seabed mapping activities have only covered a small percentage of the OCS and vary in extent across geographic regions, leaving major gaps in knowledge of the location and extent of sand resources.

Sand shoals are also habitats for economically important fisheries. Along the United States (US) Atlantic and Gulf of Mexico Coasts, sand shoals are used by juvenile reef fish, shrimp, coastal pelagic fish species during seasonal spawning migrations, and as feeding grounds for demersal fishes and sharks (Rutecki et al. 2014). Particular shoals features have been designated as EFH, and conservation recommendations are provided for special considerations that minimize environmental impacts caused by commercial uses of the ocean and seafloor. The Bureau of Ocean Energy Management (BOEM), as part of the US Department of the Interior, is responsible for the management and development of sand resources on the OCS. All proposed sand dredging projects require prior analysis of impacts to marine resources using the best available science to understand and mitigate environmental impacts when possible. The mapping of shoals will help assess potential impacts of sand dredging, assist efforts to understand species' habitat relationships with shoals, and set the stage for a broader classification of marine unconsolidated sediment ecosystems. Lastly, better information on the extent of possible sand resources will inform strategic decisions about mineral resource use and the availability to meet demand.

Shoals are common geologic features on the continental shelf and are defined as sand, or other unconsolidated material, that result in shallower water depths than surrounding areas (Rutecki et al. 2014). Other terms noted for shoals include underwater “ridges,” “banks,” or “bars.” Characteristics of shoals include crests, troughs, areas with slope, and varying sediment substrates (Vasslides and Able

2008). In the simplest terms, shoals are distinguished as areas of topographic relief compared to flat, low-relief unconsolidated sediments. Bathymetric highs are often accompanied by troughs and swales. This report follows an intensive literature review by Rutecki et al. (2014), which provides extensive examples on the geology, geography, and general biological values of sand shoals. Although these definitions and case studies were sufficient to develop an understanding of the importance of shoals in coastal geology and as fish habitat, we still lack a method that delineates the distribution and extent of shoals on the OCS, as well as a unified classification scheme for characterizing sand features in terms of geomorphology and potential habitat value. The primary objective of this chapter was to develop spatially explicit models that classify, and whenever possible verify, the location and extent of potential shoals of the United States' OCS spanning the northwest Atlantic Ocean and the Gulf of Mexico. The second objective was to identify attributes that distinguish shoals from other marine geomorphic features and propose a classification scheme compatible with the Coastal and Marine Ecological Classification Standard (CMECS) (FGDC (Federal Geographic Data Committee) 2012). The classification scheme and shoal layers developed in this volume were incorporated into the interactive mapping and reporting tool ShoalMATE (Shoal Mapping Assessment Tool for EFH), which is described in more detail in Volume 4.

2 Identifying Shoals on the OCS

2.1 Study Area

The landward boundary of the study area was defined by the Outer Continental Shelf Lands Act, which distinguishes Federal and state managed waters. Federal waters are those ≥ 3 nm from the shoreline, with the exception of Texas and the Gulf Coast of Florida where Federal waters are defined as ≥ 9 nm from the shoreline. The oceanic boundary of the study area was defined by a 50 m contour line (**Figure 2-1)**, which is the deepest extent of anticipated dredging activities, although 30 m is currently the deepest extent of existing dredge activities. The contour line was derived from the Coastal Relief Model of the National Oceanic and Atmospheric Administration (NOAA) (NOAA National Centers for Environmental Information 2010). To account for potential regional differences in seafloor geomorphology, separate analyses were conducted for three regions: Greater Atlantic, South Atlantic, and Gulf of Mexico. Because of concurrent fish research, the boundaries of the three regions generally coincided with Federal fishery management council jurisdictions and NOAA's Essential Fish Habitat (EFH) designations. The Greater Atlantic region spanned from the northern extent of Maine to just south of Cape Hatteras, North Carolina. The South Atlantic region extended from Cape Hatteras, North Carolina, southward to the Florida Keys. The Gulf of Mexico region included Texas eastward through the Florida Keys.

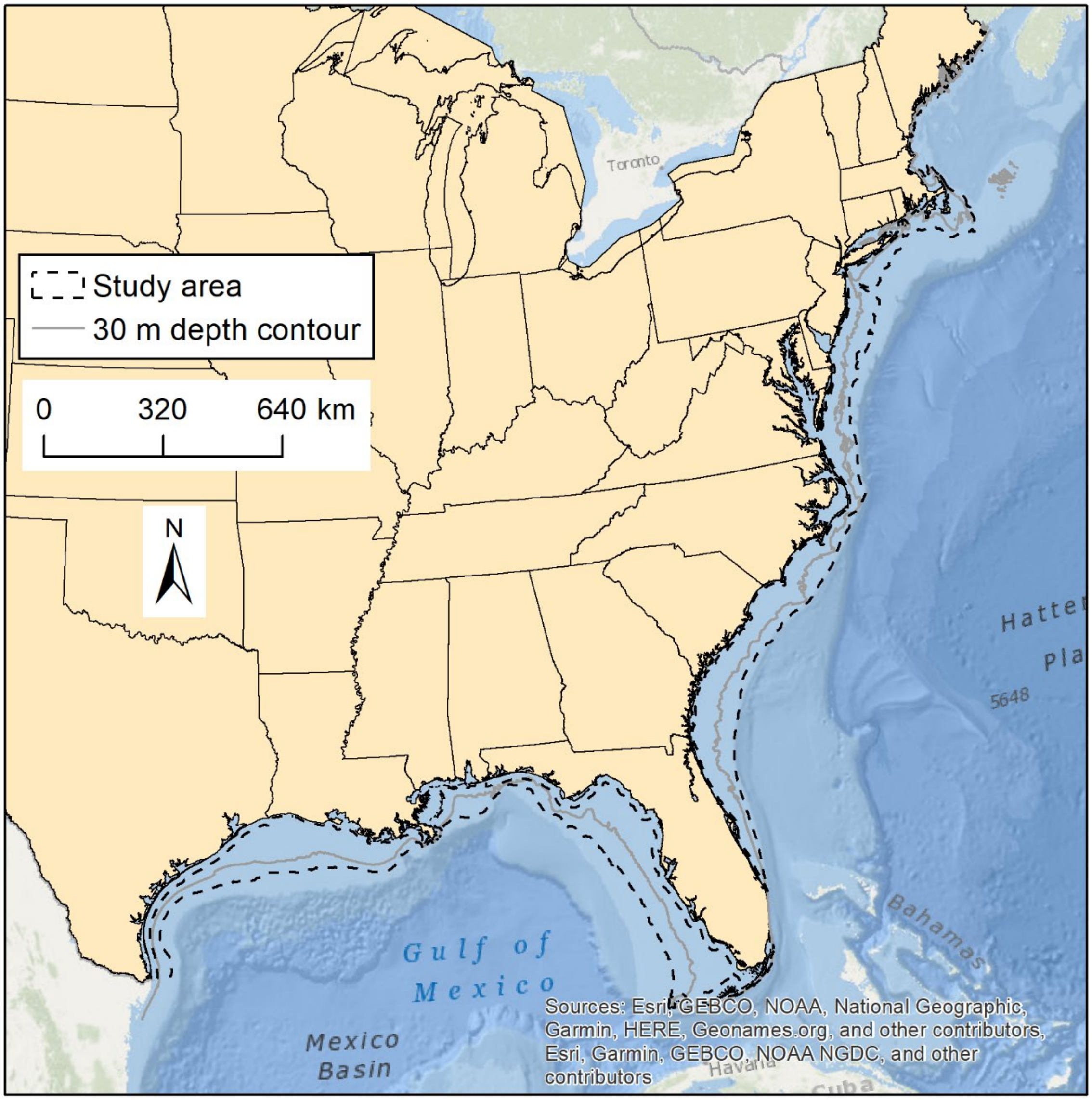

Figure 2-1. Geographic boundaries for the study area (dashed line) in the US Gulf of Mexico and Atlantic Coasts, extending from the offshore state/Federal boundary to 50-m depths.
Map also shows the 30-m depth contour, which is the deepest extent of current dredging activities.

2.2 Source Data and Spatial Data Variables

Existing bathymetric digital elevation models (DEMs) provided the primary source data for this study. Bathymetry for the central and western Gulf of Mexico was derived from NOAA's Coastal Relief Model (NOAA National Centers for Environmental Information), which characterized waters offshore of Texas, Louisiana, Mississippi, and Alabama. However, several large errors were prominent offshore of Florida. Therefore, we used sounding data summarized and developed into a 50-m raster grid by the US Geological Survey (Robbins et al. 2007). To be consistent with the other datasets, these data were resampled to a 90-m resolution with bilinear resampling. For the South Atlantic, we used a bathymetry dataset derived as part of The Nature Conservancy's South Atlantic Marine Bight Assessment (Conley et al. 2017). They synthesized 4.7 million depth sounding points obtained from the National Geophysical

Data Center's (NGDC's) Coastal Relief Model (CRM) (NOAA National Centers for Environmental Information 2010). The initial hydrographic surveys compiled by NOAA were completed between 1851 and 1965 and from survey data obtained digitally on National Ocean Service survey vessels since 1965 (NOS (National Ocean Service) 2008). The Nature Conservancy (TNC) used only data after the 1950s, and kriging was used to interpolate among sounding points (Conley et al. 2017). Despite these data being the best available for the South Atlantic, a few notable errors persisted. We updated a 1,600-km^2 and a 400-km^2 region offshore of northeastern North Carolina with sounding points acquired in 2016 (NOAA National Centers for Environmental Information 2016). To be consistent throughout the region, we used the same kriging methods conducted by TNC. Bathymetry data for the Greater Atlantic was simply obtained from the CRM (NOAA National Centers for Environmental Information 2010). Where the South Atlantic bathymetry overlapped with the Greater Atlantic study area (Virginia and northeast North Carolina), we used the South Atlantic bathymetry because it had already been improved with recent data. All spatial data were converted to the North America Alberts Equal Area Conic map projection. All data were maintained at 90-m resolution, and analyses were conducted with ArcGIS 10.6 (ESRI, Redlands, CA).

Where sand resources are found in topographic mounds such as shoals or ridges, localized spatial models have typically used relatively coarse (> 10-m horizontal cell size) DEMs to detect anomalies in the seafloor using predictors, such as slope or rugosity, to locate seabed forms consistent with shoals (Knorr 2017). In contrast, we used an unsupervised classification method for mapping shoals with data on depth, standard deviation of depth, slope, distance to shoreline, and the bathymetric position index (BPI) as key variables. In addition to the depth measures directly provided by bathymetry, we used the Benthic Terrain Modeler (Wright et al. 2012) to derive slope (3 x 3 cells) and the BPI (27 x 27 cells for Atlantic; 71 x 71 cells for Gulf of Mexico) (**Figure 2-2**). We selected the BPI analysis scale by initially exploring possibilities for each region. The relatively narrow shoals of the Atlantic became distinguished at a 27 x 27 cell analysis window, whereas shoals in the Gulf of Mexico were relatively wide and were distinguished by a 71 x 71 cell analysis window. The standard deviation of depth (SD depth) was computed with the Spatial Analyst, a focal statistics tool with a 9 x 9 cell analysis window. Distance to shoreline was calculated by back-transforming boundaries of the Submerged Lands Act (NOAA National Ocean Service, Office for Coastal Management 2016), which is set at a distance of 3–9 nm from the shoreline, depending on the state. We used the buffer tool to re-create the approximate shoreline boundaries, and then we calculated the Euclidean distance from the shoreline to the entirety of the study area.

2.3 Delineation of Geomorphic Features

After examining bathymetry-derived variables and visually inspecting the large-scale geomorphological patterns along the OCS, we restricted the analyses to waters ≤ 40 m of depth because deeper waters were often influenced by the increasing slope at the continental shelf break, which differed dramatically from the geomorphology on the shelf. Depths from 40 to 100 m are typically characterized by emergent rocky outcrops, ledges, and areas of high relief and steep slope. We excluded this area to constrain the bathymetry-derived variables to those more typical on the shelf. We standardized all predictor variables by subtracting the mean and dividing by the standard deviation of the raster layer. The subsequent data distribution had a mean of approximately zero and a standard deviation of one. We used the standardized predictor variables as inputs into the ArcGIS ISO Cluster Unsupervised Classification, which identifies clusters of cells with similar attributes. In this method, the user defines the number of groups to be classified. We ran the tool iteratively with the number of output classifications ranging consecutively from 2 to 10. Given the lack of defined boundaries of shoals, we had the goal of creating the fewest classification classes possible to simplify the identification of shoals and other geomorphic features. The minimum cells per classification were set at 1,000, and half the cells were used to train the classification.

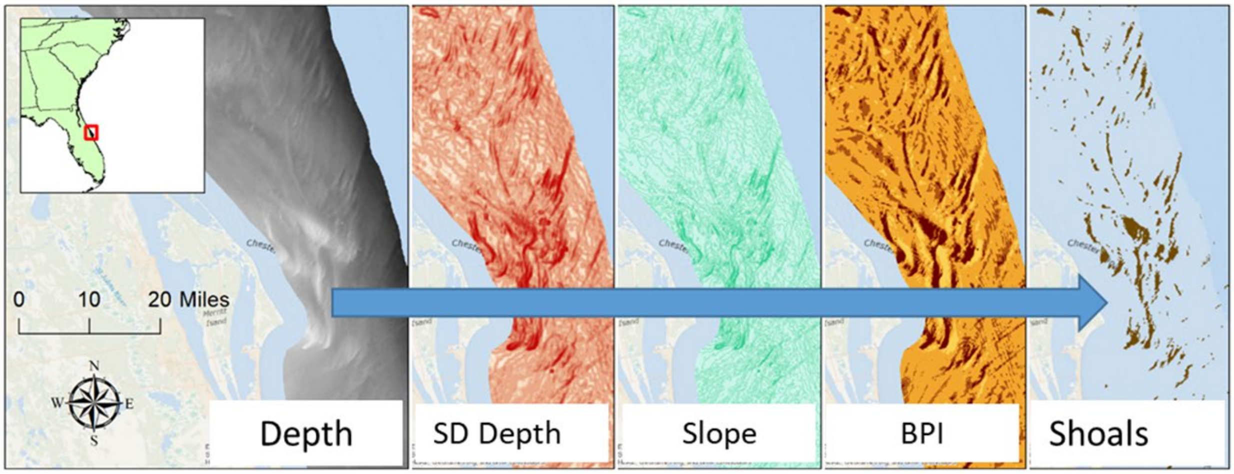

Figure 2-2. Examples of seafloor complexity metrics derived from bathymetry (depth) and used to delineate sand features and shoals.
SD depth = standard deviation of depth, BPI = bathymetric position index. The arrow represents the process of deriving three surfaces from Depth to classify features as Shoals. See text for descriptions for statistical assumptions for each derived surface.

After the initial identification, we used an eight-neighbor majority filter to remove small, isolated features. Based on the locations of a few known sand shoals (e.g., Ship Shoal in Gulf of Mexico, cape-associated shoals in Atlantic) and preliminary descriptive statistics, we classified features into "shoal," "swale," "high slope," and "non-shoal." Additionally, a minimum mapping unit of 5 hectare (ha) was used for shoal and swale classes in the Atlantic because 5 ha was the minimum size of shoal geomorphology identified in a recent review of sand shoals (Rutecki et al. 2014). For the Gulf of Mexico, a 20-ha minimum was used (see details below). All features below these size thresholds were removed, which further removed isolated cells.

For the Gulf of Mexico, the unsupervised classification resulted in nine classes, which included six labeled as "other seabed." We removed classified shoals, swales, or high slopes that were parallel and continuous with a slope contour, which were particularly evident offshore of Texas, Louisiana, and the Florida Keys. These features were often adjacent to the boundary of bathymetry data and where bathymetry surfaces appeared relatively coarse. To further address this issue, we removed classifications with a perimeter to area ratio of > 10; this resulted in the smallest classification of shoal or swale being 20 ha in size. Additionally, the two features identified as "high slope" were removed because they had such an extremely high slope that it was likely due to bathymetry errors rather than a true seabed feature. The two features removed included an area depicted as a large hole and a classified shoal/swale area > 120 km offshore of Florida (northwest of Tampa Bay) and an area in waters well known as a natural, rocky reef that had been repeatedly sampled as part of a reef fish surveys. The remaining features were aggregated into a single class called "shoals" or "swales". Statistics derived from the seafloor bathymetry are shallower in depth, higher slope, higher SD of depth and higher BPI than the surrounding seafloor (**Table 2-1**). Shoals were mapped throughout the Gulf of Mexico (**Figure 2-3**). Later in this volume, we further classify to shoal types using a new shoal classification scheme.

The South Atlantic shoals could be distinguished from regular bathymetric contours with a minimum of six classifications, and seven classifications distinguished both shoals and swales. With this methodology, an additional non-flat classification was created that depicted areas of extremely high slope and SD depth, as well as a positive BPI. Many of these features were sloping areas on the edge of shoals; however, we removed polygons parallel to the shelf break in Florida because these areas were part of the broader shelf geology. A few of these sloping features were in, or surrounding, holes; these were also removed from the shoal classification. We removed a circle-shaped classified shoal near Cape Hatteras, North Carolina, which was shown to be erroneous data in the raw CRM. We also removed erroneously classified shoal and swale features from the Florida Keys region because this geography is dominated by coral reefs and hard bottom as is characterized by the designated Habitat Areas of Particular Concern (HAPCs). The misclassification of shoals is likely because of similar characteristics, including relatively high depth heterogeneity, slope, and a positive BPI. Shoals in the South Atlantic, like the Gulf of Mexico, were distinct from the surrounding seafloor (**Table 2-1**, **Figure 2-3)**. For the Greater Atlantic, all shoals and swales were readily identifiable with four classes. Greater Atlantic shoals were similar distinct from the surrounding seafloor (**Table 2-1**, **Figure 2-3).**

Table 2-1. Variables used to delineate sand shoals via an unsupervised classification, descriptive statistics presented as mean ± (SD) for classified shoals, swales, and the entire dataset analyzed.
Analyses were conducted independently for each region and then combined.

		Greater Atlantic			South Atlantic			Gulf of Mexico		
Variable	**Scale of analysis (90-m cells)**	**Shoal**	**Swale**	**Entire seafloor**	**Shoal**	**Swale**	**Entire seafloor**	**Shoal**	**Swale**	**Entire seafloor**
Depth (m)	1	-20.87 (6.65)	-28.71 (8.26)	-28.11 (7.95)	-20.35 (6.65)	-26.45 (8.17)	-24.62 (8.42)	-19.93 (8.20)	-31.52 (7.17)	-24.21 (9.59)
SD of depth	9 x 9	1.32 (0.54)	2.46 (1.56)	0.67 (0.72)	0.99 (0.51)	1.72 (1.15)	0.49 (0.54)	0.53 (0.29)	1.33 (1.46)	0.23 (0.40)
Slope	3 x 3	0.39 (0.22)	0.79 (0.63)	0.15 (0.26)	0.29 (0.24)	0.54 (0.42)	0.11 (0.20)	0.14 (0.13)	0.41 (060)	0.04 (0.14)
Distance to shoreline (km)	NA	25.46 (13.78)	31.19 (16.60)	29.61 (17.34)	29.83 (17.31)	37.49 (20.92)	38.10 (22.44)	31.10 (15.02)	38.14 (21.84)	48.85 (27.49)
Bathymetric position index	Gulf of Mexico: 71 x 71; Atlantic: 27 x 27	2.49 (1.92)	-1.00 (3.88)	0.36 (1.62)	2.22 (1.27)	-0.56 (2.28)	0.32 (1.10)	2.41 (0.89)	-0.65 (2.92)	0.21 (1.15)

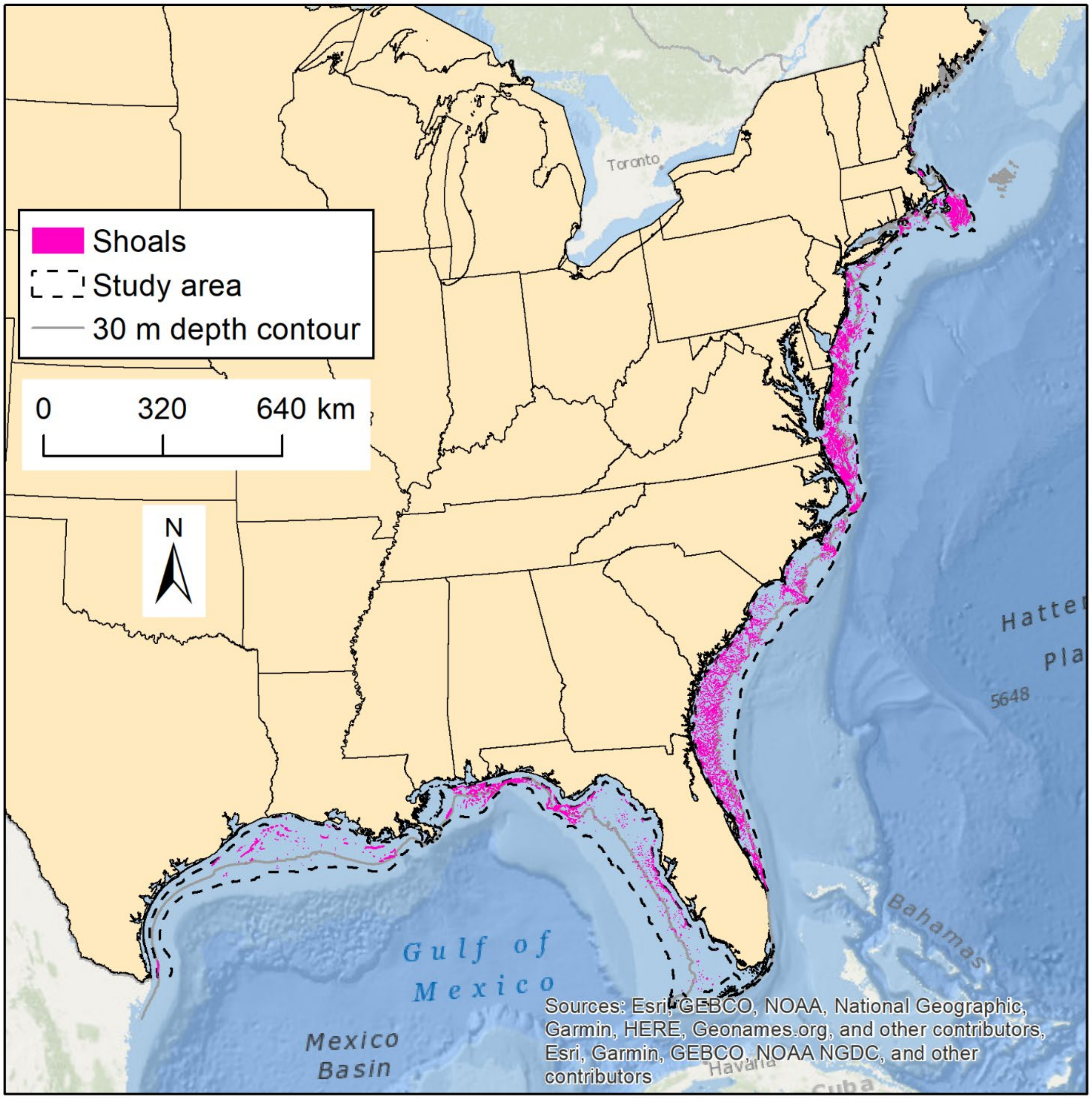

Figure 2-3. Delineated shoal features throughout the study area in the Gulf of Mexico and Atlantic OCS.

The dashed line indicates the study area of Federal waters to a maximum of 50-m depth. A 30-m depth contour also included as reference for deepest existing dredging project.

3 Classifying OCS Shoal and Sediment Resources

3.1 Goals and Design of the Shoal Classification Scheme

Having successfully identified shoals through semi-automated means, the next step was to group them into distinct classes that characterize their spatial location in the seascape, their geological origin and the environmental processes at work in their environment. This additional contextual information, conveyed through a classification system, is important to fully understanding the habitat value of these features. The classification system described in this section was based on physical and spatial variables relevant to managed fish species. This system applies to the OCS in Federal waters of the US Atlantic and Gulf of Mexico ≤ 50 m in depth.

The scheme is intended to meet the following criteria:

- It must be applicable over all US waters in the Atlantic and Gulf of Mexico.
- It must integrate with the CMECS. CMECS is a federally endorsed standard that also forms the framework for an associated BOEM product: the Marine Minerals Information System (MMIS). The Geoform and Substrate Components of CMECS are the most relevant ones in this case.
- It should be applicable at a variety of spatial scales.
- It should build on the best available science.
- Where possible, it should take into consideration the physical and chemical processes important to sand shoal formation, evolution, and temporal persistence.
- It should be conceptually open to updates as new science and data improve our knowledge.

A first step in the classification design process was a review of existing classification systems that might provide a framework or information relevant to this system. The following systems were reviewed:

- A Classification Scheme for Deep Seafloor Habitats (Greene et al. 1999)
- The CATAMI (Collaborative and Automated Tools for Analysis of Marine Imagery) Classification (Althaus et al. 2013)
- Seabed Geomorphology: A Two-Part Classification System (Dove et al. 2016)
- A Habitat Classification Scheme for the Long Island Sound Region (Auster et al. 2009)
- A New Classification Scheme of European Cold-Water Coral Habitats: Implications for Ecosystem-Based Management of the Deep Sea (Davies et al. 2017)
- A Geomorphic Classification of Estuaries and its Application to Coastal Resource Management – A New Zealand Example (Hume and Herdendorf 1988)
- INFOMAR Seabed Survey Seabed Habitat Classification (Thorsnes et al. 2018)
- A Habitat Classification Scheme for Seamount Landscapes: Assessing the Functional Role of Deep-Water Corals as Fish Habitat (Auster et al. 2005)
- Hierarchical Classifications of Sedimentary Architecture of Deep Marine Depositional Systems (Cullis et al. 2018)
- A Marine and Estuarine Habitat Classification System for Washington State (Dethier 1990)

Although some of these systems did reference geomorphological features, most included them as only as a descriptive part of a biological unit, or their focus was on a different geography than that of concern in this project. Three systems did address geomorphology and sand shoals directly (Auster et al. 2005; Dove et al. 2016; Greene et al. 1999), but only the following systems addressed the resources of interest in a systematic way, and they formed the source for most of the units:

- CMECS (FGDC (Federal Geographic Data Committee) 2012)
- 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)
- Seafloor Geomorphology as Benthic Habitat (Harris and Baker 2012)

Given the breadth of the CMECS system, we decided to use it as a basis for the shoal classification scheme, and the other schemes were used in support of the method. We drew upon international subject matter experts (SMEs) to develop a framework for a classification scheme through consensus. Meetings with SMEs were scheduled to coincide with regional and international conferences, as well as facilitated webinars. In all cases, the participants were given a short presentation on the scope and objectives of the entire project and the need for a new classification scheme for sand features. The first meeting invited experts in the Southeast US region following a workshop on Improving Coordination in Seafloor Mapping in the Southeast US OCS in April 2018. Though regional in nature, this meeting drew upon geological models and knowledge for sand features off the Carolinas, Georgia, and Florida in the South Atlantic region. A meeting with international experts was scheduled around the GeoHab conference held in May 2018 in Santa Barbara, CA. For this meeting, our team focused on international standards for classifying sand features. Lastly, we hosted two facilitated webinars to gather additional comments on a draft scheme in June 2018 and concluded with a presentation and review among BOEM and US Army Corps of Engineers (USACE) staff in July 2018. From the totality of these meetings and calls, a structure for the classification scheme was developed.

3.2 Classification Scheme Structure

The sand shoal classification scheme draws from several elements of the CMECS and incorporates some new individual units from Rutecki et al.'s (2014) review. The CMECS classification system was used as the starting point in selecting appropriate variables and domains. CMECS was originally created through a collaboration between NatureServe, the NOAA, the US Environmental Protection Agency (US EPA), US Geological Survey (USGS), and the University of Rhode Island. This classification standard provides a method for categorizing the physical, biological, and chemical components of coastal and marine ecosystems. **Figure 3-1** shows the hierarchical nature of CMECS with new levels and modifiers relevant to this study proposed for an update to CMECS. **Table 3-1** presents the settings and components, including those suggested for inclusion in classifying shoal and related sand features. The following tables provide further details of the classification scheme components **(Table 3-2 to Table 3-6**). In the Appendix, we provide more detailed definitions for each class and modifier. Furthermore, the shoal classification scheme was applied using available data in the process described in Volume 4 of this report.

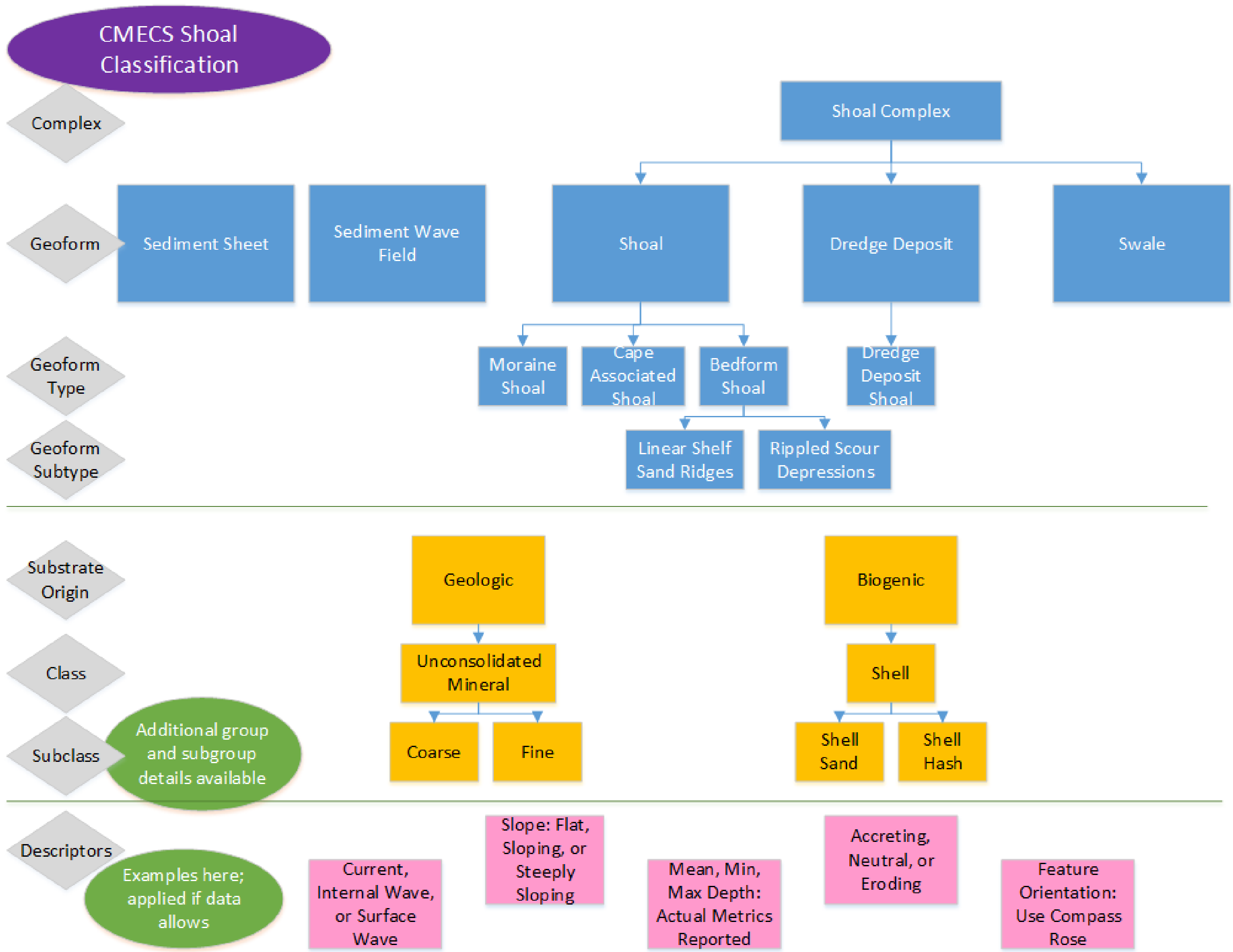

Figure 3-1. Hierarchical diagram of CMECS classification scheme proposed for sand features.

The following table illustrates the most basic structure of CMECS, which includes a series of settings, components, and modifiers. Indentation shows the hierarchical structure of items within each setting or component (**Table 3-1**). Items struck out in this classification scheme were not used in the ShoalMATE tool. Some variables in the ShoalMATE tool contain only a subset of original CMECS domains, which will be explained in further detail in subsequent tables. For the ShoalMATE tool, no distinction was made between Level 1 and Level 2 Geoforms, although due to the spatial resolution of input data layers, it is likely that most of these are Level 1. No information from the CMECS Aquatic Setting category was used in the ShoalMATE tool.

In **Table 3-2**, a subset of the available values from the CMECS ecoregion variable were used in the ShoalMATE tool. The Gulf of Mexico Fisheries Management Council (GMFMC) subdivided the Gulf of Mexico into a set of ecoregions that do not coincide with those from CMECS. These GMFMC-derived ecoregions were included in the ShoalMATE tool in a separate attribute for added spatial resolution and to match fish distribution descriptions in the GMFMC Fisheries Management Plan. For EcoregionFMC, entries may also be plural to include multiple ecoregions, such as “GMFMC Ecoregions 2-5.”

Table 3-2), the Salinity and Temperature Subcomponents are broken down into qualitative variables based on quantitative range descriptions. The ShoalMATE tool, however, includes quantitative range variables (min/max) as well as a variable for average temperature. This method ensures the highest level of precision for these values can be taken directly from source literature.

In **Table 3-3**, CMECS divides Geoform Components into Level 1 and Level 2. Level 1 Geoform components are generally larger than one square kilometer, while Level 2 are generally less than one square kilometer (see Federal Geographic Data Committee (FGDC) 2012, Section 6 for more details). However, the ShoalMATE tool does not make this distinction. A subset of the possible values were used for the CMECS variables Origin, Geoform, and GeoformType.

Table 3-4 shows the subset of CMECS Substrate Component used in the ShoalMATE tool in the native CMECS hierarchical structure.

In **Table 3-5**, the CMECS Biotic Setting and Biotic Class were selected for use in the ShoalMATE tool. Based on the level of detail in source documentation, it was determined that inclusion of the CMECS Biotic Subclass, Biotic Group, and Biotic Community were not necessary.

Lastly, in the variables are based on CMECS modifiers. In these cases, the CMECS modifiers were qualitative values based often on quantitative ranges, while the final variables selected for the ShoalMATE tool remained quantitative and captured varying ranges.

Table 3-1. CMECS settings and components.

Also see Table 2.1 in Federal Geographic Data Subcommittee (2012). Modifiers may be applied to one or more components. See Federal Geographic Data Subcommittee 2012 for further details.

Biogeographic Setting (BS)	Aquatic Setting (AS)	Water Column Component (WC)	Geoform Component (GC)	Substrate Component (SC)	Biotic Component (BC)
~~Realm~~ *~~Province~~* *Ecoregion*	*~~System~~* *~~Subsystem~~* *~~Tidal Zone~~*	**~~Layer Subcomponent~~** **Salinity Subcomponent** **Temperature Subcomponent** **~~Hydroform Subcomponent~~** *~~Hydroform Class~~* *~~Hydroform~~* *~~Hydroform Type~~* **~~Beiogeochemical Feature Subcomponent~~**	**~~Tectonic Setting Subcomponent~~** **~~Physiographic Setting Subcomponent~~** **Level 1 Geoform Subcomponent** *Geoform Origin* *Level 1 Geoform* *Level 1 Geoform Type* **Level 2 Geoform Subcomponent** *Geoform Origin* *Level 2 Geoform* *Level 2 Geoform Type*	*Substrate Origin* *Substrate Class* *Substrate Subclass* *Substrate Group* *Substrate Subgroup*	*Biotic Setting* *~~Biotic Class~~* *~~Biotic Subclass~~* *~~Biotic Group~~* *~~Biotic Community~~*

Table 3-2. Biogeographic Setting.

Ecoregion	EcoregionFMC*
Scotian Shelf	GMFMC Ecoregion 1
Gulf of Maine/Bay of Fundy	GMFMC Ecoregion 2
Virginian	GMFMC Ecoregion 3
Carolinian	GMFMC Ecoregion 4
Northern Gulf of Mexico	GMFMC Ecoregion 5
Floridian	

Table 3-2. Water Column Component.

ShoalMATE tool Water Column Component Variables:
TempMin
TempMax
TempAvg
SalinityMin
SalinityMax

Table 3-3. Geoform Component.

Geoform Component Origin (GC Origin)	Geoform Component Geoform (GC Geoform)	Geoform Component Type (GC Type)
Geologic	Shoal	Moraine Shoal
		Cape-Associated Shoal (CMECS provisional unit)
		Bedform Shoal (CMECS provisional unit)
		Isolated Shelf Shoals (CMECS provisional unit)
	Dredge Deposit	Dredge Deposit Shoal
	Swale	-
	Sediment Wave Field	-
	Sediment Sheet	-
	Flat	Tidal Flat

Table 3-4. Substrate Component.

Substrate Component Origin (SC Origin)	Substrate Component Class (SC Class)	Substrate Component Subclass (SC Subclass)	Substrate Component Group (SC Group)	Substrate Component SubGroup (SC Subgroup)
Geologic Substrate	Unconsolidated Mineral Substrate	Coarse Unconsolidated Substrate	Gravel	Boulder
				Cobble
				Pebble
				Granule
			Gravel Mixes	Sandy Gravel
				Muddy Sandy Gravel
				Muddy Gravel
			Gravelly	Gravelly Sand
				Gravelly Muddy Sand
				Gravelly Mud
		Fine Unconsolidated Substrate	Slightly Gravelly	Slightly Gravelly Sand
				Slightly Gravelly Muddy Sand
				Slightly Gravelly Sandy Mud
				Slightly Gravelly Mud
			Sand	Very Coarse Sand
				Coarse Sand
				Medium Sand
				Fine Sand
				Very Fine Sand
			Muddy Sand	Silty Sand
				Silty-Clayey Sand
				Clayey Sand
			Sandy Mud	Sandy Silt
				Sandy Silt-Clay
				Sandy Clay
			Mud	Silt
				Silt-Clay
				Clay
Biogenic Substrate	Shell Substrate	Shell Hash	Clam Hash	*Coquina* Hash
			Crepidula Hash	-
			Mussel Hash	-
			Oyster Hash	-
		Shell Sand	Clam Sand	*Coquina* Sand

Table 3-5. Biotic Component.

Biotic Component Setting (BCSetting)	Biotic Component Class (BCClass)
Planktonic	Zooplankton
	Floating/Suspended Plants and Macroalgae
	Phytoplankton
	Floating/Suspended Microbes
Benthic/Attached	Reef Biota
	Faunal Bed
	Microbial Communities
	Aquatic Vegetation Bed
	Emergent Wetland
	Scrub-Shrub Wetland

Table 3-6. Modifiers.

ShoalMATE tool Variable Name	ShoalMATE tool Variable Description	Original CMECS Variable Name	Original CMECS Variable Description
Grain Size (Phi)	Sediment grain size in Phi (numerical)	Seafloor Rugosity	Qualitative descriptors based on ranges of grain size
DOmin	Minimum dissolved oxygen (mg/L) at which a fish species or life stage can survive, or minimum dissolved oxygen measured over a sand resource	Oxygen	Qualitative descriptors based on ranges of dissolved oxygen values
DOmax	Maximum dissolved oxygen (mg/L) at which a fish species or life stage can survive, or maximum dissolved oxygen measured over a sand resource		
DepthMin_m	Minimum depth (m) at which a fish species or life stage can be found, or the minimum measured depth of a sand resource (not calculated during storm events)	Benthic Depth Zones	Qualitative descriptors based on depth ranges (m)
DepthMax_m	Maximum depth (m) at which a fish species or life stage can be found, or the maximum measured depth of a sand resource (not calculated during storm events)		
ChlaMin	Minimum chlorophyll a (µg/L) at which a fish species or life stage can survive, or the minimum measured chlorophyll a (µg/L) measured over a sand resource	Phytoplankton Productivity	Qualitative descriptors based on a range of chlorophyll a (µg/L)
ChlaMax	Maximum chlorophyll a (µg/L) at which a fish species or life stage can survive, or the maximum measured chlorophyll a (µg/L) measured over a sand resource		
TurbMin	Minimum turbidity at which a fish species or life stage can survive, or the minimum turbidity measured on a sand resource	Turbidity	Qualitative descriptors based on a range of turbidity values measured in Secchi depths
TurbMax	Maximum turbidity at which a fish species or life stage can survive, or the maximum turbidity measured on a sand resource		
SubstrateDesc	A subset of the associated CMECS variable (see right). Allowed values: Carbonate, Compacted, Mobile, Non-Mobile, Patchy, Siliciclastic, Sulfidic, Volcaniclastic, Volcanic Ash, Well-Mixed, Heterogenous (CMECS provisional unit)	Substrate Descriptors	Qualitative descriptors of seafloor substrate

3.3 Applying the Classification Scheme

The process of classifying the delineated shoals into scheme units builds on the characteristics described in **Section 2.3**, which covers the identification of the shoal. Rutecki et al. (2014) includes discussion of the environmental drivers that influence sand distribution and movement on the OCS. They also convey a number of shoal geoform types that have been integrated into this classification system as provisional CMECS units. Beyond the identification of shoal geoform types and their definitions, Rutecki et al. (2014) also provides specific examples of each geoform type in the Atlantic and Gulf of Mexico. In this section, we focus on classifying the shoal features into geoforms defined above.

Individual shoals were often grouped in space, especially in proximity to offshore capes. Groups of shoals are refered to as “complexes,” which contain two or more shoals with intermingling troughs or swales. The shoal complex are connected by past or present sedimentary and hydrodynamic processes (Rutecki et al. 2014). To assign individual shoals and swales into “shoal complexes,” we applied Tobler’s first law of geography (Tobler 1970), “everything is related to everything else, but near things are more related than distant things.” Specifically, we grouped all shoals/swales that were within 2.5 km of each other. Initially, we tested the aggregation of shoals/swales with distances ranging from 0.5–5 km. We selected a distance of 2.5 km because shorter distances led to multiple shoal complexes identified within singular, cape-associated shoal areas with the same geological origins. Meanwhile, distances ≥ 3 km led to cape-associated shoals clumped together with sand shoals with different geological origins (e.g., cape-associated and bedform shoals).

The labelling of shoals and complexes by geoform type is inevitably an interpretive process relying on a mix of localized studies, geographic position, and context, as well as other quantitative and qualitative criteria. These criteria can be referred to by the term “classifiers,” which are characteristics of individual shoals that are necessary to assign them to specific categories. For example, dominant tree height would be a classifier that allows one to distinguish between forest and shrub land. The geoform type definitions and specific geographic examples of each shoal type from Rutecki et al. (2014) formed a basis for the qualitative classification of the mapped shoals into the various categories. Specific classifiers applied in this process include:

- Proximity to coastal landforms such as capes, inlets, deltas, and historical deltas
- Shape and orientation
- Proximity and spatial relationship to nearby similar shoals
- Proximity to shoals that were classified into geoform types in Rutecki et al. (2014) and associated studies
- Position on the OCS (coastal or offshore)
- Spatial continuity with terrestrial ridge-like features

Evaluation of these factors, consideration of the quantitative bathymetric parameters, and prior characterization by other researchers was used to arrive at the final classified shoal data layer employed in ShoalMATE and presented in the following maps (**Figure 3-2** to **Figure 3-5**). Cape-associated shoals are distributed as their name indicates with prominent landward capes along the Atlantic and to lesser extent Gulf of Mexico (**Figure 3-2**). Isolated shelf shoals are the most dominant in the Gulf of Mexico (**Figure 3-3**), whereas bedform shoal features dominate the features on the Atlantic Coast (**Figure 3-4** and **Figure 3-5**).

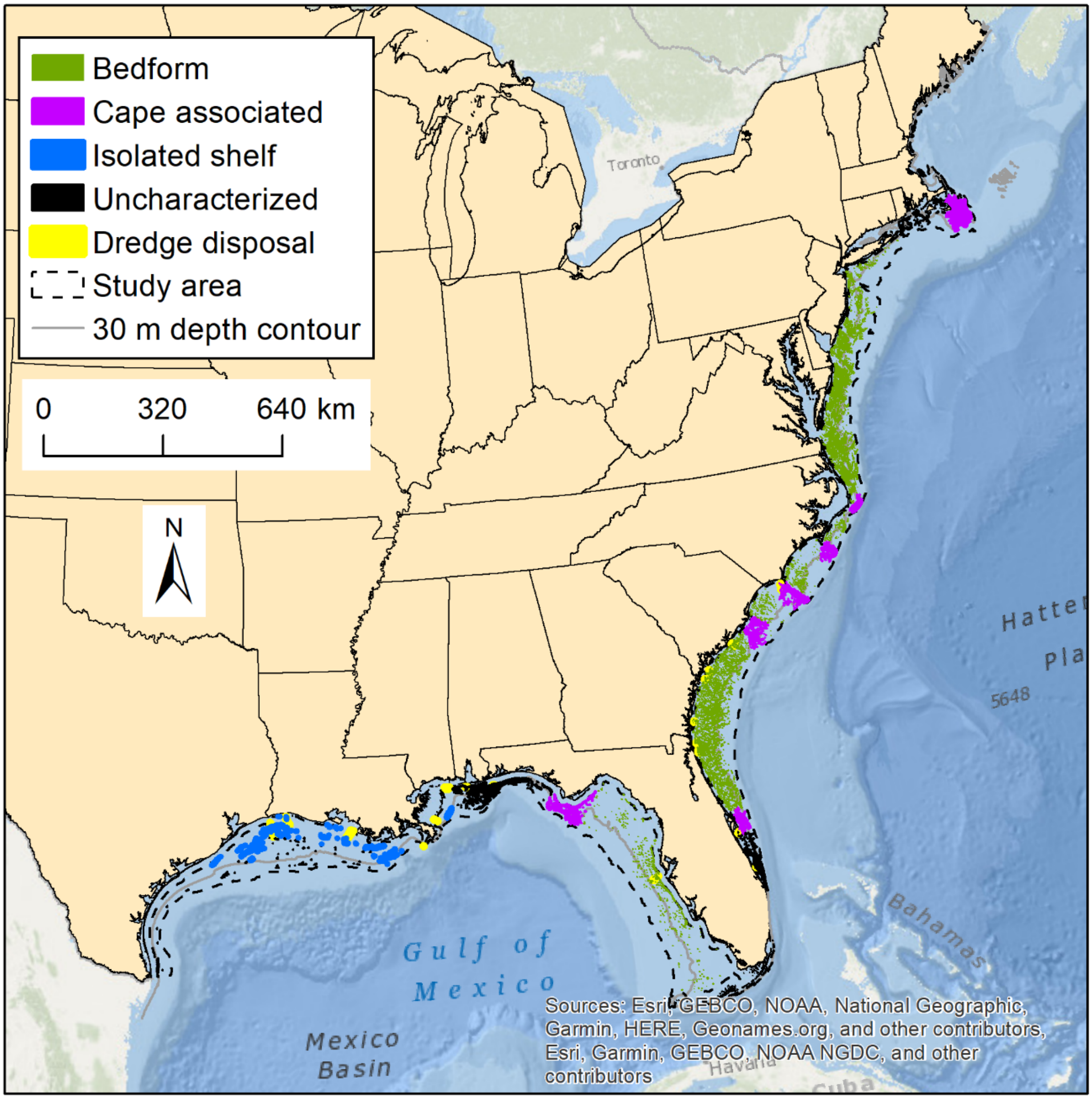

Figure 3-2. Classified sand features throughout the study area in the Gulf of Mexico and Atlantic OCS.

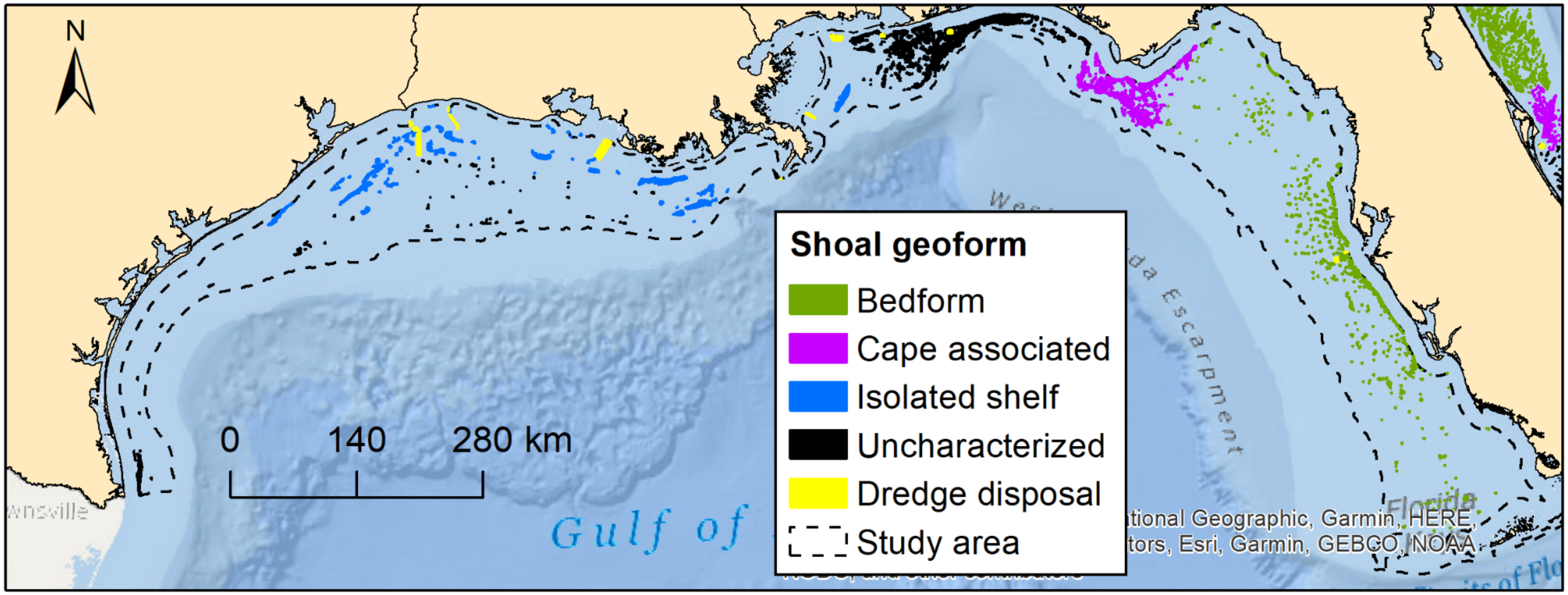

Figure 3-3. Overview map of classified shoal distributions in the northern Gulf of Mexico region.

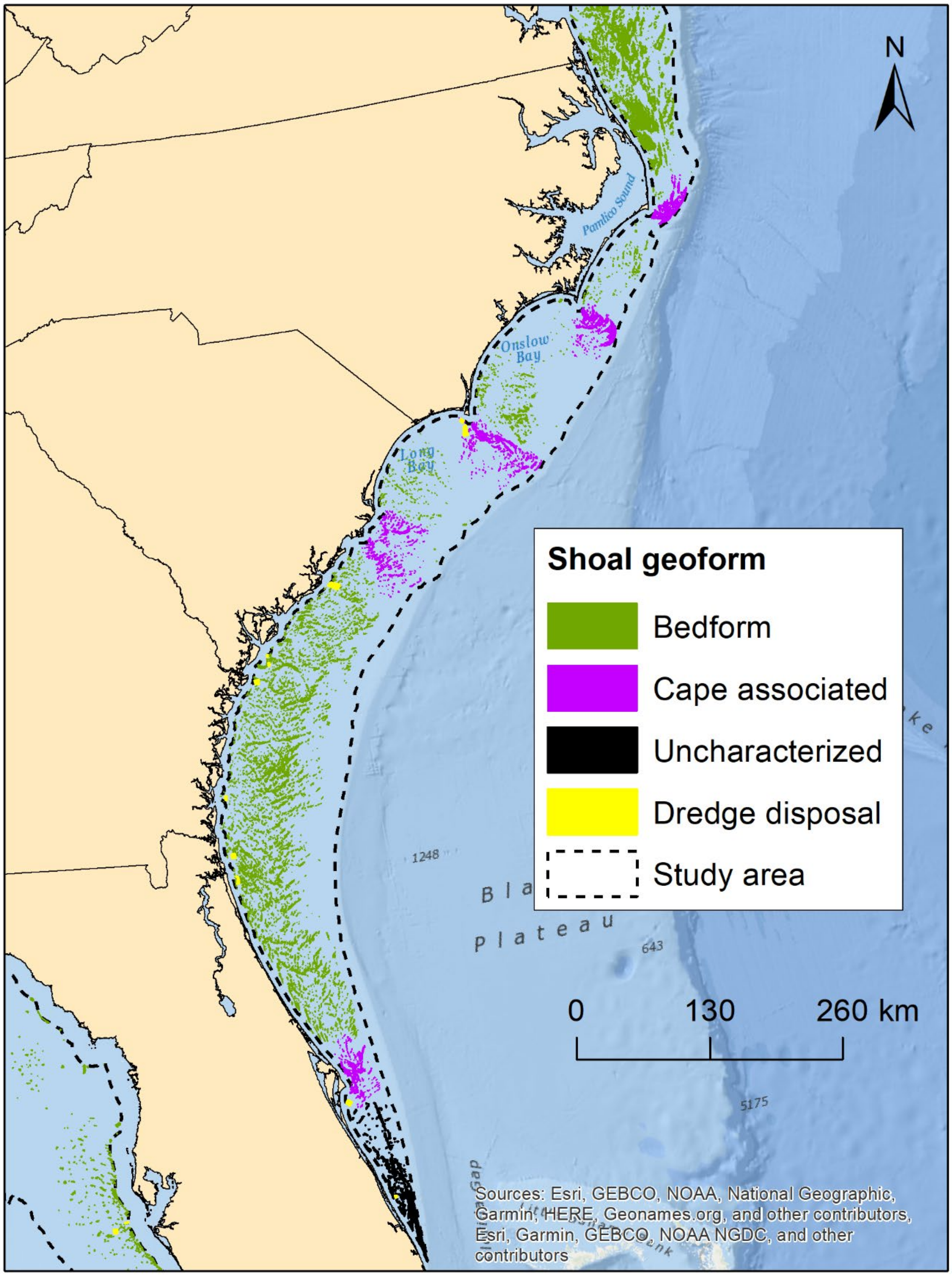

Figure 3-4. Overview map of classified shoals along the southeast coast of the US.

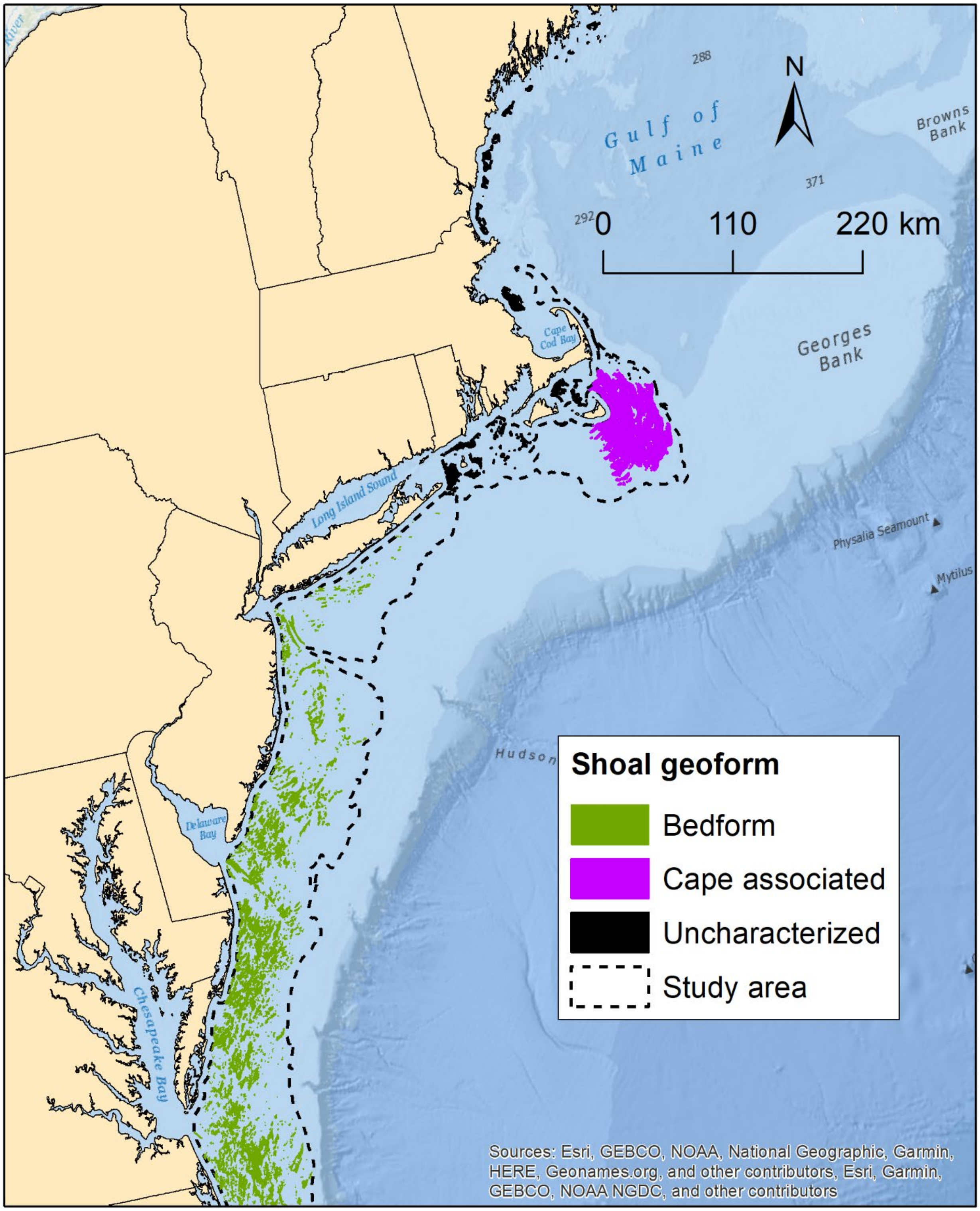

Figure 3-5. Overview map of classified shoal distributions in the US mid-Atlantic and New England region.

3.4 Verifying the Shoal and Shoal Complex Dataset

Before integrating the classified shoals dataset into the ShoalMATE tool, we assessed the accuracy of the results. We did not have the capability to collect new high-resolution geophysical or geotechnical data to quantitatively validate our modeled distribution of sand shoals. Instead, we conducted a qualitative assessment by comparing classified shoals to shoal features commonly known and mapped as “shoals” or similar features. More specifically, our classifications were compared to two reference place-name datasets to see how well the product characterized named shoals and shoal-like features.

1. *Marine Place Names* - 2016. These data are derived from features on the NOAA Electronic Navigational Charts and contain names for features in the US territorial waters and the OCS. Because different place names are displayed depending on the scale displayed, we used a 1:80,000 scale. The dataset had a total of 55 categories of features, and we used the following categories for spatial comparison with our results: a) bank, b) bar, c) ridge, d) shoal

2. *Undersea Feature Names* - 2017. The GEOnet Names Server (GNS) provides access to the National Geospatial Intelligence Agency's (NGA) and the US Board on Geographic Names' (BGN) database of geographic feature names. The database is the official repository of foreign place-name decisions approved by the BGN. Geographic coordinates are approximate and are intended for general location. Place-name information is based on the Geographic Names Database, which contains official standard names approved by the United States Board on Geographic Names and maintained by the National Geospatial-Intelligence Agency. A total of 55 types of features were present in this dataset. The following were extracted for the comparison: a) undersea bank and banks, b) undersea ridge and ridges, c) undersea shoal and shoals.

The source datasets have point topology and consist of the centroids of what become textual labels on Electronic Navigation Charts or other cartographic products. These labels are intended to loosely characterize the extent and orientation of the features on a map. To avoid false negatives where the textual label may have intersected the shoal dataset but the centroid did not, the centroid points were buffered to a 3-km radius. Both datasets include features in both state and Federal waters. Buffered features that fell outside that zone were removed because the area of interest pertains to Federal waters less than 50 m in depth and because we restricted our classification shoreward of the continental shelf break beginning at 40 m.

Overall the shoals dataset strongly aligns with named shoals and shoal-like features as compared with two nationally scoped datasets. Named shoals in these source data tended to be located in nearshore waters. This is probably a function of their role as a navigation hazard and ease of observation. Many of the named shoals in Federal waters were located near Nantucket, RI, so there is some geographical bias inherent in these results. Nonetheless, the results suggest that the derived shoal maps are accurately capturing these features and should be suitable for its purpose as a screening tool to identify other possible sand resources. Many of the largest sand features delineated corresponded well to verified sand resources. It is important to note that these classified features are predictions; those features not already named or verified as sand resources by BOEM and other agencies need further validation and high-resolution seafloor surveys.

The Marine Place Name database included the feature type “bars” that exclusively fell within state waters shoreward of the boundary for our study area, and ridges occurred at depths beyond 50 m or outside the Atlantic and Gulf of Mexico. The vast majority of named shoals also were identified in state waters, outside our area of study. The results showed a user’s accuracy (percent of named shoals present in the dataset) of 95% for shoal-like features (**Table 3-7**). An assessment of the producer's accuracy (how many of the mapped shoals have been mapped correctly) is not possible without a separate validation process.

The Undersea Feature Place Names database included a much smaller number of shoal-like features in the source data. This is likely a function of the offshore focus of this database. Fewer banks were present overall, but a strong percentage were located in the study area. Few ridges occurred in the database and in the study area as well. Those shoals that were present were accurately identified in the derived shoal maps (**Table 3-7**). The results showed a user's accuracy (percent of named shoals present in the dataset) of 88% for shoals and an aggregate accuracy of 57% for shoal-like features. As with the Marine Place Names data, a producer's accuracy is not possible at this time.

Table 3-7. Spatial alignment of detected shoal features with named places for "Marine Place Name" (Top) and "Undersea Feature Place Name" (Bottom) datasets.

Marine Place Name Feature Type	Number of Features in Source Data	Number of Features within Study Area	Number of Features Intersecting with Shoal Dataset
Bars	63	0	0
Ridges	30	0	0
Shoals	761	46	44
Undersea Feature Type	**Number of Features in Source Data**	**Number of Features within Study Area**	**Number of Features Intersecting with Shoal Dataset**
Banks	21	6	3
Ridges	7	1	1
Shoals	15	9	8

Our analysis of a large-scale coastal relief digital elevation model was limited by the quality and coverage in the dataset. The CRM is a compilation of soundings from current and modern hydrographic sonar survey, as well as historical and sometimes sparse soundings, smoothed to a consistent 90-m resolution through interpolation or modeling. The aggregation of data across time does ignore the temporal dynamics and movement of sediment along the Gulf and Atlantic Coasts and may only represent larger features or shoal complexes that are relatively persistent as features, even if their exact position may change over time with sediment transport (Pendleton et al. 2017). The classified collection of shoal complexes should be applied as an initial screening tool for planning modern surveys that would be required to locate the amount and shape of sand resources available. Our analysis restricted feature sizes to ≥ 5 ha in the Atlantic based upon prior reviews of significant sand resources (Rutecki et al. 2014), and we were further limited to identifying features of > 20 ha in the Gulf of Mexico. One of the major challenges in our analysis was detecting and delineating relatively low-relief and small bedform sand features, as are present off the coast of North Carolina, South Carolina, Georgia, and the western Gulf Coast of Florida. The relative importance of these features as sand resources and fish habitat need to be further explored.

A second step in evaluating the quality of the shoal dataset was a visual comparison to known sand resources. Classified shoals aligned well with verified sand resources identified by BOEM and its partners. In the Gulf of Mexico (**Figure 3-6**), Ship Shoal was identified as an isolated shelf shoal, though the shape of the polygon from the classification model deviates from the boundary of the sand resource. This is likely because sand resource mapping included both shoal and non-shoal sediment resources. Similarly, verified sand resources aligned well with the bedform shoals delineated off the northern Outer Banks of North Carolina (**Figure 3-7**), though the model classifies additional similarly shaped shore parallel sand features in the region. Differences in the shapes and extent between physically mapped sand resources and the classified layer could be due to the knowledge of the subsurface sand depth, resolution of the source data, or an artifact of temporal dynamics of the shoals. Verified sand resources have been surveyed using modern hydrographic techniques and higher resolution (meters to tens of meters) surfaces. The delineation of our classified layer was made from a composite CRM, which may include another temporal image of the shoal that may have shifted or changed in morphology due to sediment transport and coastal circulation dynamics. Similarly, bedform shoals were delineated and conform to identified sand resources off Virginia and Delaware, though once again, there are numerous sand features classified by the model, with many farther offshore that have not yet been verified as sand resources by BOEM and others (**Figure 3-8**).

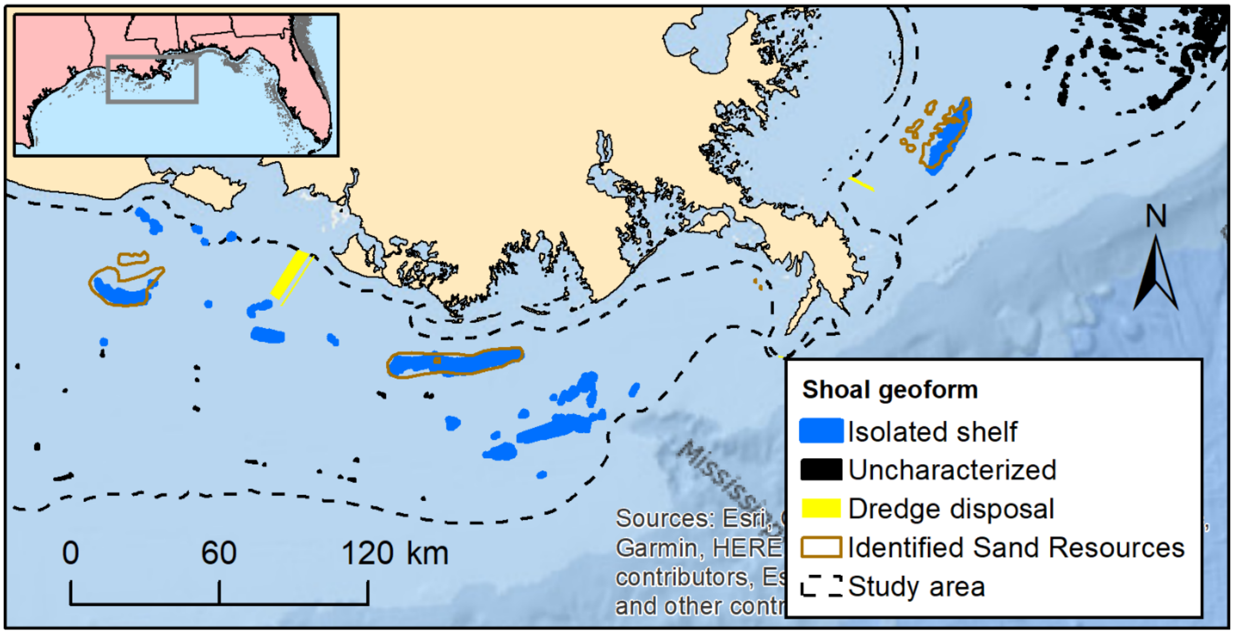

Figure 3-6. Concordance of classified shoals with known and identified sand resources (tan polygons) south of Louisiana in the Gulf of Mexico region.

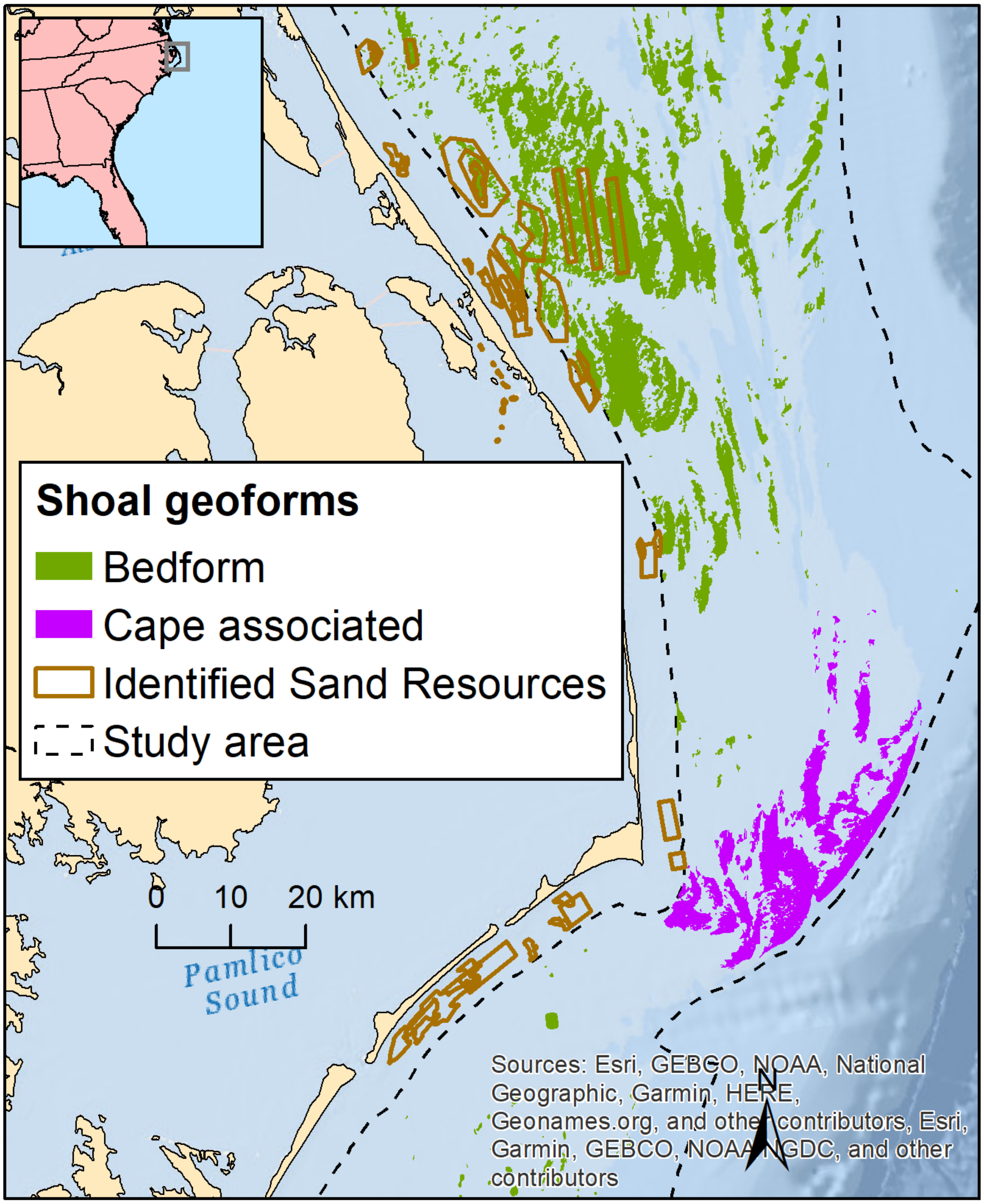

Figure 3-7. Concordance of classified shoals with identified sand resources (tan polygons) off Cape Hatteras in the South Atlantic region.
Note identified sand resource area also shown shoreward of the state boundary outside the study area.

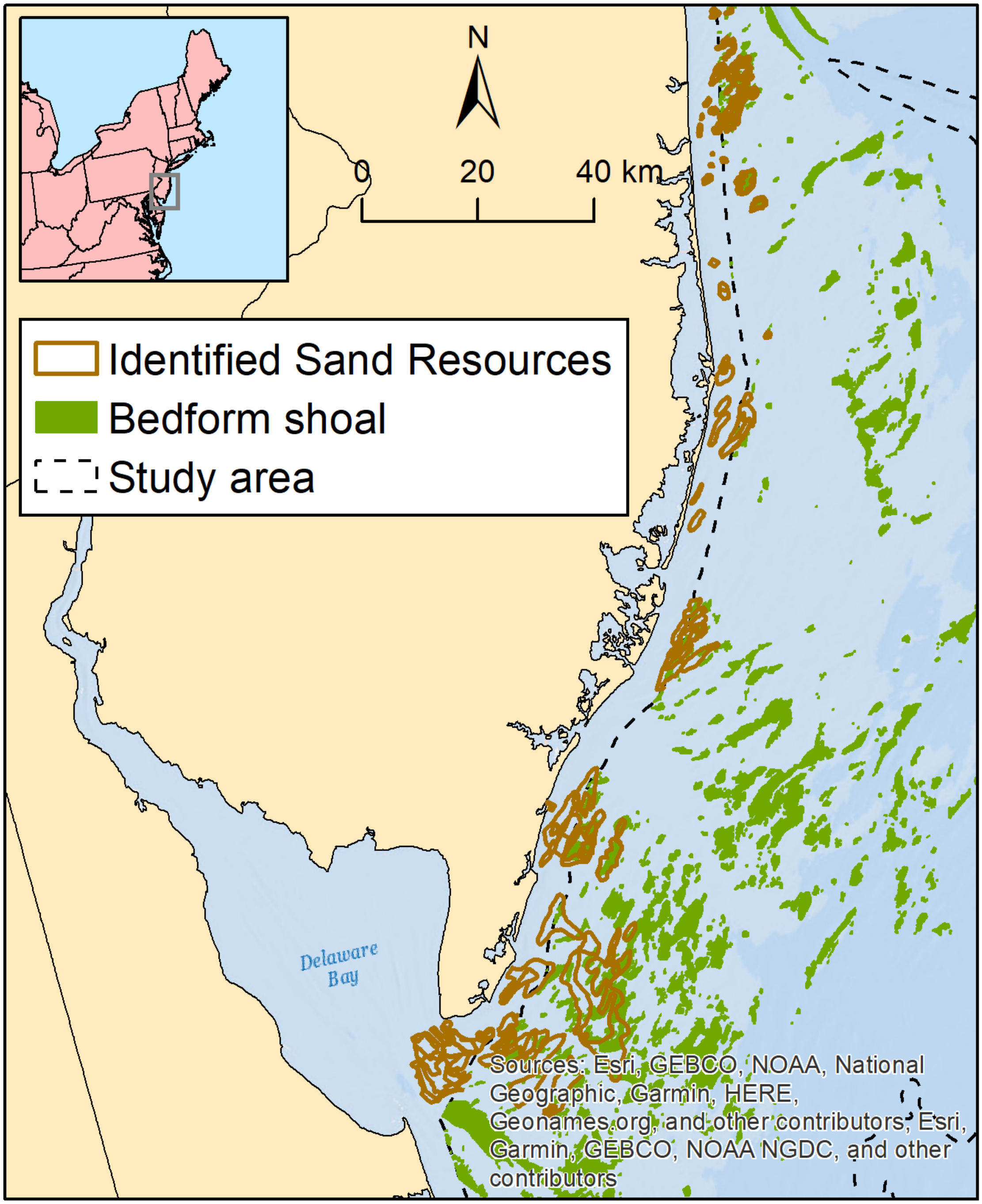

Figure 3-8. Concordance of classified shoals with verified sand resources (tan polygons) in the mid-Atlantic region.
Note identified sand resource area also shown shoreward of the state boundary outside the study area.

Improvements to this classification model and derived layers could come in several forms. First, our exercise focused exclusively on a study area bounded by Federal management jurisdictions and excluded waters under state management jurisdiction. Although the overall framework and modeling approach could be extended to include state waters, several important factors may complicate the modeling process, including increased variation and noise in nearshore BPI and slope values, as well as the reduced usefulness of factors like distance to shore and depth. A separate model derivation would be required to extend classifications to these areas. It is not likely that dredging efforts would extend beyond the 50-m depth contour, so extensions offshore may not be necessary. Secondly, we did not include sediment type into the shoal and sand feature classifications. Sediment and bedform maps exist for both the Gulf and Atlantic Coasts elsewhere, developed through separate initiatives (e.g., TNC's South Atlantic Bight Marine Assessment (Conley et al. 2017), Chris Jenkins, University of Colorado, unpublished data). These maps were modeled using various spatial interpolation techniques from compilation of historical to modern bottom samples. In all cases, these classifications should be taken as screening tools and will require modern geological surveys and validation.

4 Results and Discussion

This study builds upon previous syntheses of the dynamics and distribution of offshore sand features in the Gulf of Mexico and US Atlantic continental shelf (Rutecki et al. 2014). Prior shoal classification studies have focused on discrete areas, such as cape-associated shoals along coastal North Carolina (Thieler et al. 2014) or shoal complexes near southwest Florida in the Gulf of Mexico (Finkl et al. 2007). The classification conducted here extends the concept that seafloor geomorphology and complexity metrics derived from broad-scale coastal elevation models can be used as an initial survey of broad areas of the continental shelf to delineate shoals and shoal complexes (Knorr 2017). Previously, a classification model of the east coast of Florida used 10-m resolution seafloor imagery to detect shoals and similar sand features (Knorr 2017) and found that simple thresholds of complexity metrics like rugosity can be used to delineate sand features. In contrast to Knorr (2017), we used a broader extent and a coarser 90-m data resolution to analyze seafloor geomorphology and complexity metrics to delineate features and classify shoals. Our results showed seafloor complexity metrics were still distinct from the surrounding seafloor at this 90-m resolution. Seafloor metrics, such as slope and the BPI, were substantially higher than the surrounding seafloor across all three geographies analyzed here (**Table 2-1**). Our study also differed from Knorr (2017) in that we used the BPI and distance from shore as predictors. Large shoals and shoal complexes were readily visible in the bathymetry surfaces, and these were delineated using a range of spatial scales of the BPI, depending upon the geographic area. For the Gulf of Mexico in particular, the larger scale of BPI (71 x 71 cells) was helpful to delineate wide (> 3 km) shoals that were partially characterized by low slope (i.e., flat cells) within 90-m resolution cells. Distance to shoreline was helpful in the classification because shoals are distinguished as shallow areas that are farther offshore than other waters of similar depths. The unsupervised classification accounts for such predictor combinations, although such interactions are difficult to quantify in descriptive statistics.

In the Gulf of Mexico, large cape-associated and isolated shelf shoals were the most prominent sand features detected and classified (**Figure 3-3)**, specifically Ship Shoal, Trinity Shoal, Sabine Bank, and St. Bernard Shoals near Louisiana and the cape shoals near Apalachicola, FL. There were other classified shoals scattered along Texas, Louisiana, and Alabama that were left as uncharacterized sand feature classes. Similarly, the majority of the west Florida shelf was populated by smaller features that have been previously identified as valuable sand sources for beach nourishment in the region (Finkl et al. 2007). Cape-associated shoals are readily visible in the base bathymetry maps and easily delineated in our models in the US South Atlantic (**Figure 3-4**). These shoals are associated with dynamic seabed areas off Cape Canaveral, FL, and along the Carolinas and Virginia Coasts. However, the majority of areas of potential sand resources appear to be captured in the scattered and smaller bedform features off South Carolina and Georgia (**Figure 3-4**). The small size and relatively low relief of these features may result in uncertainty in size, number, and extent in this region. North of Cape Hatteras into the mid-Atlantic and

New England, smaller bedforms dominate except around Cape Cod where cape-associated shoals are present (**Figure 3-5**).

We labeled shoal complexes by their geoform whenever possible with a basis from Rutecki et al.'s (2014) classification based on geological origin. More specifically, we classified geoform types into: 1) cape-associated shoals, 2) bedform shoals, 3) isolated shelf shoals, 4) dredge disposal sites, and 5) uncharacterized shoals. To name shoals, we examined labels from the ESRI oceans basemap, literature sources, and examples from the Rutecki et al. (2014) review of sand shoals. Dredge disposal sites were identified from disposal locations that were categorized as dredged material disposal or spoil grounds (U.S. Environmental Protection Agency n.d.).

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Appendix A: Shoal Classification Scheme Dictionary

In the following text, superscript next to the unit name indicates the primary source of the definition.
[10] Indicates units or definitions drawn from CMECS text (FGDC 2012).
[24] Indicates definitions drawn from the Rutecki et al. (2015) report. In some cases, the definition is a hybrid of both.

A.1 Shoal Classification Scheme Component Definitions

Biogeographic Setting (EC)

EC units are included to add contextual information to the sediment resources data layer such as prevailing oceanographic conditions. The CMECS *Biogeographic Setting* is a fully hierarchical component. For the purposes of this project, we intend to apply *Ecoregion* units and aggregate upward in the hierarchy if necessary.

Ecoregions:[10] Areas of relatively homogeneous species composition, clearly distinct from adjacent systems. The species composition is likely to be determined by the predominance of a small number of ecosystems and/or a distinct suite of oceanographic or topographic features. The dominant biogeographic forcing agents defining the ecoregions vary from location to location but may include isolation, upwelling, nutrient inputs, freshwater influx, temperature regimes, ice regimes, exposure, sediments, currents, and bathymetric or coastal complexity."

CMECS ecoregions are drawn from the Marine Ecosystems of the World (Spalding et al. 2007) Units relevant to this report consist of the following:

- Scotian
- Gulf of Maine / Bay of Fundy
- Virginian
- Carolinian
- Northern Gulf of Mexico, and
- Floridian

Geoform (GC)

The geomorphology of the seafloor is one of the two primary characteristics of interest in this project. The *Geoform Component* is a semi-hierarchical framework. This project will focus on the physiographic setting, *Geoform* and *Goform Types*, which are hierarchical. It should be noted that like many geomorphic classifications, the definitions are somewhat subjective and there is conceptual overlap between units. Some banks can considered bars, some ridges can be considered shoals, and so on. Nevertheless, these terms are helpful and provide a mental picture of the feature.

Physiographic Setting:[10]

With the geographic scope of this project being the Federal waters of the Atlantic and Gulf of Mexico Exclusive Economic Zone, the only relevant physiographic setting needed was Continental Shelf. As the geography of the tool expands, it is likely that additional CMECS physiographic units may be come relevant and need to be included.

> **Continental/Island Shelf:**[10] That part of the continental margin that is between the shoreline and the continental slope (or a depth or 200 m when there is no noticeable continental slope); it is

characterized by its very gentle slope of 0.1°. Island shelves are analogous to the continental shelves, but surround islands.

For the purposes of this project, where only coarse resolution data may be available, this system proposes adding a provisional *Complex* level to the CMECS Geoform. The proposed units are as follows:

Geoform Complex:[10] This is a new provisional CMECS level**.** Geoform complexes consist of many small geoforms within an area or a repeatable assemblage of associated geoforms that function as a system. Examples include salt marshes that contain tidal creeks, marsh platforms, banks and pannes. The *Complex* level should be used when data resolutions are not high enough to distinguish the boundaries of individual geoforms or where the minimum mapping unit consists of multiple geoform units.

> **Shoal Complex:** [24] 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. An area consisting of several shoals too small to be distinguished individually due to data resolution or mapping constraints.

Geoforms[10] are physical, coastal and seafloor structures that are generally no larger than hundreds of square kilometers in size. This size determination may be an areal extent or a linear distance. Larger geoforms (Level 1) are generally larger than 1 km^2, and correspond to Megahabitats in the Greene et al. (2007) classification system. These features can be defined using geologic or geomorphic maps and bathymetric images of the seafloor at map scales of 1:250,000 or less. Smaller geoforms (Level 2) are generally less than 1 km^2 in size (or less than 1 km in distance); and correspond to Meso- and Macro-habitats in the Greene et al. (2007) system. Level 2 geoforms (such as individual coral reefs, tide pools, and sand wave fields) can be identified through *in situ* observational methods (such as underwater videography) or through low-altitude, high-resolution optical or acoustic remote sensing.

> **Shoal:**[10,24] A natural, underwater ridge, bank, or bar consisting of, or covered by, sand or other unconsolidated material, resulting in shallower water depths ($\geq$ 1 m) than surrounding areas. Morphologically and spatially dynamic, they are primarily shaped by waves and currents and can be driven across the seafloor during tropical storms and hurricanes as well as less intense (but more frequent northern meteorological fronts and other lower intensity events. In some cases, shoals may be exposed during low tides.
>
> > **Moraine Shoal:**[10] The submerged portion of a glacial moraine that reaches close to the surface. These often occur where sea-level rise has drowned former terrestrial glacial features.
> >
> > **Cape-Associated Shoals**:[24] Active sedimentary systems that extend from cuspate foreland promontories formed by two barrier islands (Rutecki et al. 2014 figures 2-3 and 2-5) or mainland beach ridges joined at approximately right angles (McNinch and Luettich Jr 2000). Cape-associated shoals form due to the convergence of two longshore drift cells, and as a result of self-organization of the coast in response to a high-angle-wave instability in shoreline shape. Cape-associated shoals can also be influenced by the preexisting geological framework (Rutecki et al. 2014 figure 2-4; Thieler and Ashton 2011)
> >
> > **Bedform Shoals**:[10,24] A continuum of morpho-sedimentary bedforms exists along the inner- and mid-continental shelf of siliciclastic passive continental margins. The continuum ranges from sorted bedforms occupying the sediment-starved end of the continuum and linear shoals and shore-attached ridges on the sediment abundant end of the continuum.

Isolated Shelf Shoals:[24] Shoals formed by relict coastal sedimentary processes exposed by ravinement. These are discrete features associated with a single landform or shoreline position.

Geoforms Subtypes are further refined types of geoforms and are fully nested within the Geoform level. Geoform Subtype is a new proposed level for CMECS based on the source data and needs of this project. The two units below are currently the only Geoform Subtypes proposed.

Linear Shelf Sand Ridges:[24] Along the mid-Atlantic coast, linear shore-normal shelf sand shoal complexes are most prominent along the Delaware-Maryland-Virginia inner shelf, where they are the dominant features (Hayes and Nairn 2004; Swift and Field 1981, Figures 2-8 and 2-9). According to Swift and Field (1981), there are three basic types of linear shore-normal shelf sand shoal (called ridge and swale by the authors) morphologies found within the Delaware-Maryland system; they include ***shore-attached ridges*** (0- to 3-m isobaths), ***nearshore ridges*** (6- to 18-m isobaths and within 10 km off shore), and ***offshore ridges*** (greater than 10 km offshore). Each ridge is roughly 3–4 km long and 0.5–1 km wide with ridges spaced 1–4 km apart.

Rippled Scour Depressions:[24] Are bathymetrically subtle, large-scale bed features that are characterized by alternating bands of coarse- and fine-grained sediment with wavelengths of hundreds of meters (Van Oyen et al. 2011), and negative relief of ~ 1 m that trend obliquely to the coast (Guitierrez et al. 2005, Figure 2-6). Where there is a dominant direction of suspended sediment transport, these features tend to be asymmetrical, with coarser flanks facing updrift, into the direction of dominant sediment transport. Where there is no dominant current direction, they tend to be symmetric (e.g., Goff et al. 2005; Diesing et al. 2006). The coarse material is in the troughs (or swales), and the ridges are finer grained. Self-organizing features due to the interaction of frictional sediment transport, bottom composition, and turbulence, with bottom roughness over the troughs causing turbulence that inhibits the settling of fines within the troughs.

Sediment Wave Field:[10] An area of wave-like bedforms in sand or other unconsolidated material, which are formed by the action of tides, currents, or waves. These bedforms range from centimeters to meters in size and may be superimposed on larger features. Sand waves lack the deep scour associated with dunes or megaripples (Bates and Jackson 1984). For this project, these features can be distinguished from other shoals due to their lack of physical relief relative to the surrounding sea floor. They are distinguished from Sediment Sheets by the presence of bedforms (ripples) indicating higher energy and potentially coarser substrates.

Sediment Sheet:[10] A thin, widespread, sedimentary deposit formed by a transgressive sea advancing for a considerable distance over a stable shelf area; may also be called a blanket deposit (Bates and Jackson 1984). For this project, the term will be used to describe unconsolidated substrates lacking bedforms or rippling and without physical relief relative to the surrounding seafloor.

Dredge Deposit:[10] An accumulation on the seafloor (or land surface) where spoil materials from a dredging operation are placed. They often exhibit some topographical expression and can support biological communities that are different than the surrounding area. These deposits are often unconsolidated in character, but they can also be relatively stable.

Dredge Deposit Shoal:[10] A subaqueous area that is substantially shallower than the surrounding area, which resulted from the deposition of materials from dredging and dumping.

Substrate Component (SC)

Classification units for describing the surficial substrate will draw directly from the CMECS SC. This is a fully hierarchical framework organized by substrate origin (geologic, biogenic, and anthropogenic). Most units to be applied in this project are from the Unconsolidated Sediments Class, which has sub-units based on Wentworth grain size fractions. The Biogenic Shell Substrate class is included because source literature indicates that the presence and amount of shell hash is important to fish habitat value.

It is unlikely that actual grain size information necessary to classify substrate to the group or subgroup level will be available throughout the project geography; therefore, the definitions below are at the CMECS subclass level. Definitions for *Substrate Group* and *Substrate Subgroup* levels can be found at https://iocm.noaa.gov/cmecs.

Substrate Subclasses:[10] are determined by the composition and particle size of the dominant substrate origin in the surface sediments. Class and subclass definitions represent a merging of approaches from Wentworth (1922), Folk (1954), and the FGDC-STD-004.

> **Coarse Unconsolidated Substrate:**[10] Geologic Substrate surface layer contains > 5% Gravel (particles 2 mm to < 4,096 mm). These sediments are classified using the upper three rows of the Folk (1954) Gravel-Sand-Mud diagram.
>
> **Fine Unconsolidated Substrate:**[10] Geologic Substrate surface layer contains less than 5% gravel (particles 2 mm to < 4,096 mm in diameter). These sediments are classified using the bottom two rows of the Folk (1954) Gravel-Sand-Mud diagram and the entire Folk (1954) Sand-Silt-Clay diagram.
>
> **Shell Hash:**[10] Surface substrate layers are dominated by loose shell accumulations with a median particle size of 2 mm to < 64 mm (Granules and Pebbles). Shells may be broken or whole. The presence of Shell Hash is noted in this subclass (and in the following groups).
>
> **Shell Sand:**[10] Biogenic Substrate layers that are dominated by Sand that is primarily composed of shell particles with a median particle size of 0.0625 mm to < 2 mm (Sand). Shells or remains are generally broken and difficult to identify. For this reason, only substrate-forming taxa that produce distinctive Sand types are listed as substrate groups. When the composition and origin of Sand is unclear, it is assumed to be mineral Sand and is classified as a Geologic Origin substrate.

Biotic Component (BC)

The CMECS Biotic Component focuses on living organisms of the water column and seabed at a variety of scales. The BC is organized into a branched hierarchy of five nested levels: Biotic Setting, Biotic Class, Biotic Subclass, Biotic Group, and Biotic Community. The biotic setting indicates whether the biota are attached or closely associated with the benthos or are suspended or floating in the water column. Biotic classes and biotic subclasses describe major biological characteristics at a fairly coarse level. Unless otherwise noted, biotic classification units are defined by the dominance of life forms, taxa, or other classifiers in an observation. For collected observations (such as grab samples or cores), dominance is measured in terms of biomass or numbers of individuals, as specified by the user. In the case of images and visual estimates, dominance is assigned to the taxa with the greatest spatial percent cover in the observational footprint (image footprint or field of view).

Based on the source data available for the project, it is expected that only the Biotic Setting and Biotic Class units would be useful. Of these only a subset would be expected to occur in the project study area. These are defined below.

Planktonic Biota[10]: This setting includes biota that drift, float, or remain suspended in the water column in aggregations that are big enough to be (a) detected by the human eye (or with mild magnification) or (b) sampled with a fine-plankton net. Planktonic biota are not regularly associated with the seafloor.

Zooplankton:[10] Zooplankton are heterotrophic biota of the water column; zooplankton drift with the currents, but may (or may not) be able to move through the water under their own power. CMECS classifies zooplankton that may range in size from gigantic salp chains (strings of gelatinous filter feeding tunicates that attain a length of 30 m or more), to radiolarians (minute, shelled amoebas). CMECS was not designed to be used for the smallest planktonic forms (nanoplankton or picoplankton). CMECS Class Zooplankton includes both Holoplankton (that live out their entire life histories in the plankton) and Meroplankton (that are transient in the plankton).

Floating/Suspended Plants and Macroalgae:[10] This class includes areas dominated by vascular plants, detached plant parts, or macroalgae that are floating on the surface or are suspended in the water column—that is, plants and macroalgae that are not rooted or attached to the bottom.

Phytoplankton:[10] This class includes areas of floating or suspended microscopic algae that are capable of photosynthesis. Although some species are motile, they are generally passively transported by water movements. Under certain conditions, they can form aggregations, large blooms, or colonies.

Floating/Suspended Microbes:[10] Aggregations of microbes that are floating or suspended in the water column and not attached to the bottom or to any benthic substrate.

Benthic Biota[10]: This biotic setting describes areas where biota lives on, in, or in close association with the seafloor or other substrates (e.g., pilings, buoys), extending down to include the layers of sediment that contain multi-cellular life.

Reef Biota:[10] Areas dominated by reef-building fauna, including living corals, mollusks, polychaetes, or glass sponges. In order to be classified as Reef Biota, colonizing organisms must be judged to be sufficiently abundant to construct identifiable biogenic substrates. When not present in densities sufficient to construct reef substrate, the biota is classified in the Aquatic Vegetation Bed or Faunal Bed classes.

The Reef Biota Class refers to only the living component of reef structures. If referring to the reef structure, users should use the reef units in the Geoform Component. If referring to the composition of the reef substrate independent of the living cover, users should employ the Coral Substrate, Shell Substrate, or Worm Substrate Classes in the SC.

Faunal Bed:[10] In this class the seabed is dominated or characterized by a cover of animals that are closely associated with the bottom, including attached, clinging, sessile, infaunal, burrowing, laying, interstitial, and slow moving animals, but not animals that have created substrate (Reef Biota). Unlike Reef Biota, Faunal Bed biota cannot (or are not sufficiently abundant to) construct identifiable substrate. "Slow moving" animals included in the Faunal Bed class are defined as being incapable of moving outside the boundaries of the classification unit within one day. Faunal Bed organisms are aquatic, but they may be able to withstand periods of exposure to air.
Faunal Bed food webs may receive basic trophic inputs from benthic photosynthesis or chemosynthesis, plankton, allochthonous detritus and debris, or other sources. In nature, Faunal Bed habitats are often composed of complex mixes and associations of animals of different phyla,

sizes, feeding strategies, and habits, and these areas can be difficult to classify. Faunal Bed classifications are determined in CMECS by greatest percent cover of fauna or faunal structures, or (particularly for infauna) by estimates of greatest biomass.

Microbial Communities:[10] These are areas dominated by colonies of microscopic or single-celled organisms that form a hard structure, visible film, layer, or mat on or near the surface of the substrate. Colonies may be composed of benthic microalgae (e.g., diatoms), photosynthetic bacteria (e.g., cyanobacteria), archaea, saprotrophic bacteria (e.g., decomposers or decay organisms), chemoautotrophic bacteria, or other microbial groups. These features may exist on or near the surface of the sediment either subtidally or subaerially, or they may exist as extensive areas of decay associated with dead organisms that have fallen to the seafloor.

The additional remaining CMECS Classes of **Aquatic Vegetation Bed**, **Emergent Wetland**, and **Scrub-Shrub Wetland** are not expected to occur in the project study area and are not included in this data dictionary.

<u>Modifiers (M)</u>

Modifiers further describe classification units and can be applied as needed and where source data supports their use. In some cases, modifiers may themselves be mapping units (e.g., rugosity grids). Modifers for use in this project are grouped as follows:

Anthropogenic Modifiers

Dredged:[10] Landscape that is mechanically altered by the removal of sediments or other materials (e.g., shell) in order to deepen or widen channels (e.g., for navigation or alteration to hydrology).

Filled Deposition Site:[10] Areas where materials (such as sand or shell) have been placed on (or in) an area of coast or a water body.

Physical Modifiers

Energy Direction:[10] Energy can be classified according to its principal direction of travel or influence. In the case of tidal energy, this is generally an oscillation between onshore and offshore motions. In the case of currents and waves, the energy is usually directional.

Baroclinic	Motion along lines of equal pressure within the water column
Circular	Motion in a closed, circular form
Downward	Descending and perpendicular to the sea surface or bottom
Horizontal	Parallel to the sea surface or bottom
Mixed	Combination of more than one of above directions
Seaward	On land, water currents following a topographic gradient toward the sea
Upward	Ascending and perpendicular to the sea surface or bottom

Energy Intensity:[10] Energy Intensity is classified into four categories as shown. Additional terms may be applied in this system as necessary to better reflect conditions at the sediment/water interface.

Very Low Current Energy	Area experiences little current motion under most conditions
Low Current Energy	Area typically experiences very weak currents (0–1 knots)
Moderate Current Energy	Area regularly experiences moderate tidal currents (> 1–3 knots)
High Current Energy	Area regularly experiences strong currents (> 3 knots)

Energy Type:[10] The Energy Type Modifier is adapted from Dethier (1990) and Zacharias et al. (1998) with type categories as follows:

Current	Coherent directional motion of the water
Internal Wave	Vertical and transverse oscillating water motion, below the surface, due to seismic energy or a pressure differential
Surface Wave	Vertical and transverse oscillating surface water motion due to wind or seismic energy
Tide	Periodic, horizontally oscillating water motion
Wind	Coherent directional motion of the atmosphere

Seafloor Rugosity:[10] Seafloor rugosity, a measure of surface "roughness," is applicable at several scales using different measures (e.g., bathymetric x-y-z data, measured transect data, video data). Rugosity is derived as the ratio of surface area to planar (flat) area for a grid cell, or as the ratio of surface area to linear area along transects, and is calculated as follows:

$$fr = Ar / Ag$$

where Ar is the real (true, actual) surface area and Ag is the geometric surface area (IUPAC 1997).

Values for Seafloor Rugosity are taken from Greene et al. 2007. The five rugosity types and their associated numeric values are shown below

Very Low	1.0 to < 1.25
Low	1.25 to < 1.50
Moderate	1.50 to < 1.75
High	1.75 to < 2.00
Very High	≥ 2.00

Rugosity Value: Recognizing that small differences in rugosity may be important to the habitat value of certain biota, this field will be populated by the actual rugosity value and not assigned to one of the more general CMECS categories.

Slope:[10] The Slope modifier refers to the angle of the substrate at a scale appropriate for the feature being described; Greene et al.'s (2007) geological classification is followed here to characterize slope.

Substrate Descriptors:[10] Although the CMECS SC describes substrate size and composition, following Wentworth (1922) and Folk (1954) to describe particle sizes and mixes, it generally does not consider geologic composition or several other important attributes. The following substrate descriptors provide consistent terminology to meet the needs of this project.

Carbonate	Geologic Origin particles or substrates composed mainly of carbonate minerals (e.g., limestone, dolostone).
Compacted	Unconsolidated sediments with very little water content and a hard, packed form that resists penetration and resuspension. This is one of several terms that are used in CMECS to describe the fluid consistency of substrates.
Mobile	Bedded sediments which regularly re-suspend and/or move with local hydrodynamics due to the density, grain size, shape, and/or high water content of the sediment, or due to the higher hydrodynamic energy experienced in the local area. This term and the corresponding term Non-Mobile are used in CMECS to describe or predict behavior of substrates.
Non-mobile	Bedded sediments that do not regularly re-suspend and/or move with local hydrodynamics due to the density, grain size, shape, and/or compaction (low water content) of the sediment particles, or due to the lower hydrodynamic energy experienced in the local area.
Patchy	Different elements within a sample, observational unit, or reporting unit are grouped into clusters or patches at the scale of the sample or unit. "Patchy" implies that clusters of elements or particles are arranged in a haphazard manner, as clusters of pebbles scattered on sand. This is one of several terms used in CMECS to describe unit variability.
Siliciclastic	Geologic Substrate Origin particles or substrates composed primarily of silicate minerals, e.g., quartz, sandstone, siltstone.
Sulfidic	Substrate in which bacterial sulfate reduction is an important biogeochemical process; this generally occurs in anaerobic environments, is often identifiable by a very low reflectance black or blue color, and is a characteristic "rotten egg" odor when sediments are examined in air.
Volcaniclastic	Geologic Origin particles or substrates composed primarily of volcanic rock, crystals, glassy pumice, ash, or other volcanic products.
Volcanic Ash	A substrate or substrate layer composed primarily of volcanic dust and volcanic ash, often with various aeolian or marine-generated particles mixed in. In areas of the deep sea, where terrigenous input and bioturbation are limited, Volcanic Ash may be present in distinct layers at depth in the substrate matrix (see the "Layers" modifier).

Well-mixed Different elements within a sample, observational unit, or reporting unit are well-mixed or poorly sorted at the scale of the sample or unit. Well-mixed implies that elements or particles are completely and relatively evenly intermingled, e.g., Granule/Sand/Mud particles in an area with high bioturbation. This is one of several terms used in CMECS to describe unit variability. Note that CMECS does not use the equivalent geological term "Poorly Sorted," because the descriptor may be used to describe distributions of non-geological features (such as biological communities or Geoform Component structures).

Well-sorted Different elements within a sample, observational unit, or reporting unit are separated into different areas at the scale of the sample or unit. Well-sorted implies that elements or particles are (or have been) separated and arranged in a non-haphazard manner, as an area of Coarse Sand adjacent to an area of Clay. This is one of several terms used in CMECS to describe unit variability.

Heterogenous A diverse complex substrate pattern that contains multiple pattern types or may be applied in situations where there is evidence of several patterns, but resolution of the data does not allow discrimination of the individual types.

Surface Pattern:[10] These roughness patterns may have physical origins (e.g., caused by wave or current action) or biological origins (due to activities of life forms, e.g., mounds or tunnels).

Biological Roughness appears due to bioturbation, fecal mounds, tunneling, feeding or locomotory activities of megafauna, or other faunal activities. Further characterization of biological features is described in the Biotic Component.

Irregular Sediment surface has a perceptible roughness or texture that is non-regular in either frequency, direction, or amplitude.

Physical Roughness appears due to water motion, but the nature of the roughness is other than Rippled.

Rippled Closely spaced, regular, repeating, vertical variations in the height of a sandy or muddy bottom, with a very short wavelength on the order of centimeters. A rippled substrate is generally caused by the physical processes of water motion.

Scarred Roughness appears due to localized sediment disturbance resulting either from natural causes (e.g., slumps) or anthropogenic causes (e.g., anchor scars, propeller scars, trawl scars, or other fishing gear scars), but not as an artifact of camera or sampling gear deployment.

Smooth There is no perceptible roughness or texture to sediment surface at scales of less than 1 m.

Heterogenous The surface has a complex mix of patterns or the pattern appears mixed but cannot be further described due to insufficient resolution of the data.

Physiochemical Modifiers

Oxygen Regime:[10] For the purposes of this project actual dissolved oxygen minimum values will be reported rather than the range categories in CMECS. These values will be reported in mg/liter.

Spatial Modifiers

Benthic Depth Zone:[10] These are generally based on the zones in which surf or ocean swell influences bottom communities, lower limits of vegetation (such as kelp), overall photic availability, and temperature. The zones within this category are drawn or adapted from Greene et al. (2007) and Connor (1997). The following definitions are intended as guidance for adaptation of depth ranges to regional environmental conditions:

Shallow Infralittoral	0 to < 5 m deep
Deep Infralittoral	5 to < 30 m deep
Circalittoral	30 to < 200 m deep

Temporal Modifiers

Temporal Persistence:[10] The Temporal Persistence Modifier describes the permanency or variability of a hydromorphic, geomorphic, or biological feature. Though qualitative and relative, it is useful is distinguishing between features that are similar in morphology—but are temporally diverse in terms of stability. For this project this will refer to the physical integrity of a feature (shoal, ridge, etc.) and over what time period it maintains its shape. The following CMECS temporal persistence units are included in this project:

Months

Years

Inter-Annual

Decades

Centuries

<u>Additional Modifiers (M)</u>

A series of additional modifiers or descriptive units have been considered potentially valuable to assessing the character of sand resources/shoals. These are intended to characterize the spatial relationship of individual features to the surrounding landscape and understand their longevity and integrity, which may be important to their long-term function as fish habitat. It is understood that further research, spatial analysis, and time-series information will be needed to apply these modifiers with confidence. As data becomes available to conduct these types of analysis, it is expected that they will become part of the shoals descriptive database.

Benthic Depth Zone:[10] These are generally based on the zones in which surf or ocean swell influences bottom communities, lower limits of vegetation (such as kelp), overall photic availability, and temperature. The zones within this category are drawn or adapted from Greene et al. (2007) and Connor (1997). The following definitions are intended as guidance for adaptation of depth ranges to regional environmental conditions:

Shallow Infralittoral	0 to < 5 m deep
Deep Infralittoral	5 to < 30 m deep
Circalittoral	30 to < 200 m deep

Feature Orientation: Feature orientation is a reflection of the cardinal direction of the longest axis of a shoal, ridge, or other bathymetric feature. This will be expressed as a numeric value in degrees based on a geographic compass rose.

Positional Stability: This expresses the likelihood of an ephemeral feature changing geographic location while still generally maintaining its structure and size. Positional stability is most relevant in capturing the movement across the seascape of sand waves, dunes, etc. Units for this modifier are still in development and will be informed by time-series data.

Accretion Status: This is a measure of whether a feature (shoal, bank, ridge, dune, or bar) is growing in volume and extent due to sediment accretion or shrinking due to erosion. In cases where neither process is underway or where both processes are cancelling each other out, then the status would be neutral. Accretion status can only be assessed in the context of a timeframe. A window of at least 3 years is needed for BOEM borrow areas.

Shelf Position: This is a relative description of the spatial location of a feature on the continental shelf. Units for this modifier are still in development and will be determined by the source data.

Bathymetric Position Index (BPI): Output slope values (raster grids) are derived for each cell as the maximum rate of change from the cell to its neighbor. BPI is a second-order derivative of bathymetry modified from topographic position index as defined in Weiss (2001) and Iampietro and Kvitek (2002).

Standard Deviation of Depth: This metric will be applied to characterize the frequency (distance between ripple crests) and amplitude (height of crest above trough) of ripples and systematically repeating bedforms.

Disturbance Regime: This is measure of how often the surface of the substrate is disturbed or re-worked by storm events or strong ocean currents. Disturbance regimes of months to years are appropriate for sediment deposits in Federal waters less than 50 m deep.

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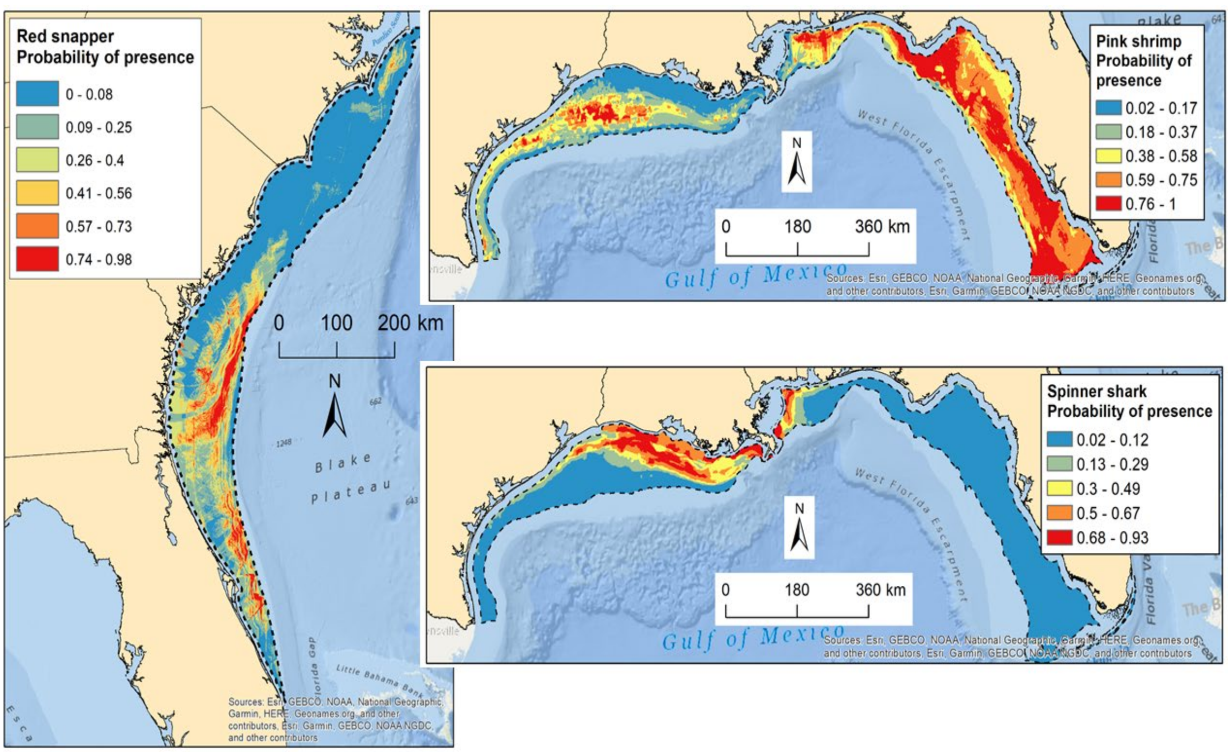
Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features

Volume 3: Predicting the Distribution of Select Fish Species of the Gulf of Mexico, South Atlantic, and Greater Atlantic



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Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features

Volume 3: Predicting the Distribution of Select Fish Species of the Gulf of Mexico, South Atlantic, and Greater Atlantic

January 2020

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List of Abbreviations and Acronyms

AUC	area under the curve statistic
BL	bottom longline
BOEM	Bureau of Ocean Energy Management
BPI	bathymetric position index
BRT	boosted regression trees
CPUE	catch per unit effort
CRM	Coastal Relief Model
CSSP	Congressional Supplemental Sampling Program
CCMA	Center for Coastal Monitoring and Assessment
CV	coefficient of variation
DOI	Department of the Interior
EFH	Essential Fish Habitat
FL	fork length
ft	foot/feet
GAM	generalized additive model
GHRSST	Group for High Resolution Sea Surface Temperature
GIS	Geographic Information System
IE	interaction effect
IUCN	International Union for Conservation of Nature
kg	kilogram(s)
km	kilometer(s)
km^2	kilometers squared
L	liter(s)
m	meter(s)
min	minute(s)
MLD	mixed layer depth
mg	milligram(s)
mm	millimeter(s)
MODIS	Moderate Resolution Imaging Spectroradiometer
MUR	Multiscale Ultrahigh Resolution
MSLABS	Southeast Fisheries Science Center, NMFS, Mississippi Laboratory
n	sample size
nat TL	natural total length
NCCOS	National Centers for Coastal Ocean Science
NCODA	Navy Coupled Ocean Data Assimilation
NEFSC	Northeast Fisheries Science Center
nGoM	northern Gulf of Mexico
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NOS	National Ocean Service
nm	nautical mile(s)
NWI	National Wetlands Inventory
OCS	Outer Continental Shelf
OCSLA	Outer Continental Shelf Lands Act
ppt	part(s) per thousand
psu	practical salinity unit(s)
R_s	Spearman rank correlation
s	second(s)

SDM	species distribution model
SEAMAP	Southeast Area Monitoring and Assessment Program
SERFS	Southeast Reef Fish Survey
SL	standard length
SLA	Submerged Lands Act
SST	sea surface temperature
TL	total length
°C	degree(s) celsius

Abstract

Species distribution models (SDMs) are a state-of-the-art statistical modeling approach that quantifies the relationships between species and spatially explicit environmental data. SDMs work by extending the identified species-habitat relationships to the entire distribution of species under consideration. These predictive modeling results are ideal to inform management decisions. In this volume, we used a variety of fisheries-independent data sources in the Gulf of Mexico and South Atlantic to produce SDMs for select marine fish and shrimp species. Environmental data on habitats included oceanographic conditions, geomorphology, geography, prey, and the nearby ecosystems of wetlands and estuaries. For the Greater Atlantic, we summarize SDMs developed by the Northeast Fisheries Science Center that combined trawl surveys with data on oceanographic conditions, substrate, and zooplankton. Together, these maps and quantified habitat relationships (or lack thereof) add to the information synthesized in *Volume 1: Fish Habitat Associations and the Potential Effects of Dredging on the Atlantic and Gulf of Mexico Outer Continental Shelf*. The analyses evaluated the best habitat predictors of marine species and depicted the distribution of select marine fish and shrimp species. Species' relationships with geomorphology characteristics were limited and of minor importance compared to other habitat predictors. None of the Gulf of Mexico species examined were related to bottom currents, slope, or the heterogeneity of depth. Of minor importance in the models, white shrimp had a higher catch per unit effort (CPUE) farther away from shoals, and pink shrimp were positively related to sand grain sizes. Red snapper age-0 had a higher CPUE in close proximity to shoals and where the bathymetric position index predominately showed a hill topography. In the South Atlantic, none of the five species examined were associated with geomorphology characteristics. Overall, species' distributions were primarily related to oceanographic conditions, nearby wetlands and estuaries, and prey species. When applicable, geomorphology predictors only had minor influence on species distribution.

1 Predicting the Marine Distribution of Three Penaeid Shrimp Species in the Northern Gulf of Mexico

1.1 Introduction

In the northern Gulf of Mexico (nGoM), USA, the Penaeid species of brown shrimp (*Penaeus aztecus*), pink shrimp (*Farfantepenaeus duorarum*), and white shrimp (*Litopenaeus setiferus*) have high economic value. From 2006 to 2015, annual commercial catches of shrimp in the region had an economic value ranging from \$327.6 to \$585.8 million and catches have ranged from 178.9 to 289.0 million pounds (National Marine Fisheries Service 2017). As a specific example of the Penaeid species, 2016 landings included brown shrimp (\$157 million), white shrimp (\$206 million), and pink shrimp (\$24.4 million) (NOAA NMFS Office of Science and Technology 2019). In addition to their economic importance, shrimp are key components of the ecosystem as a common prey for both benthic and pelagic fish (Tarnecki et al. 2016). In particular, brown shrimp have been documented as an integral part of the nGoM food web, as they are described as prey of small pelagic fish, small demersal fish, flatfish, king mackerel, Spanish mackerel, benthic feeding sharks, several snapper and grouper species, black drum, red drum, and others (Tarnecki et al. 2016).

Brown, pink, and white shrimp are estuarine-dependent in their early life stages, which is when they are relatively well studied. Within estuaries, juvenile white and brown shrimp are most abundant at open water-marsh edges, and they respond positively to the amount of time the marsh is flooded (Minello and Rozas 2002; Rozas and Minello 2015; Rozas et al. 2007). As adults, these three Penaeids are demersal and utilize the marine environment. Of the three Penaeids, brown shrimp are the best studied in the nGoM. Of particular importance, research has concentrated on the effects of hypoxia on brown shrimp (Craig 2012; Craig and Crowder 2005; Craig et al. 2005). These studies show that brown shrimp aggregate at the edge of hypoxic zones and may move farther inshore or offshore to avoid hypoxic waters. Since 2007, the development of species distribution models (SDMs) have begun to proliferate for marine species (Melo-Merino et al. 2020). SDMs use statistical species-habitat relationships to predict the spatial distribution of species (Guisan and Zimmermann 2000). Montero et al. (2016) developed an SDM for brown shrimp and found brown shrimp relative abundance was positively related to mud, shallow depths (< 100 m), lower salinities (< 20 ppt), higher bottom temperatures (> 20°C), latitude/longitude, and were uncommon in hypoxic conditions. For pink shrimp, Drexler and Ainsworth (2013) developed an SDM and tested five predictor variables. Their results were initially modeled at a 10-km scale and were aggregated into broad polygons that improved model accuracy (Drexler and Ainsworth 2013). Their model showed pink shrimp were less abundant in waters with mud sediments compared to other grain sizes. Drexler and Ainsworth (2013) also showed pink shrimp abundance declined most in waters > 120 m in depth and with waters < 15℃. They found a negative relationship with chlorophyll and a positive relationship with dissolved oxygen of ≥ 5 ml $^{-L}$. Concerning adult white shrimp, their habitat use and marine distribution remains poorly documented.

Recent advances in modeling the distribution of brown and pink shrimp have greatly improved our knowledge of the marine distribution of Penaeids. However, even with these species, much remains unknown about their relationships to geomorphology, ocean currents, and the nearby ecosystems on which each species depends. Turner (1977) found a strong positive correlation between coarse measures of inshore brown shrimp landings and emergent wetland area in the nGoM. Diop et al. (2007) found landings of white shrimp over time were positively, but weakly, correlated with late juvenile abundances. Yet, the distribution of Penaeids in the offshore, marine ecosystem has not yet been related to nearby wetlands and estuaries. Overall, testing the relationships of shrimp species' distribution with a comprehensive suite of habitat variables has the potential to inform management, strengthen environmental impact assessments, and refine our knowledge of important waters for each species.

The need to fill these knowledge gaps is exemplified by marine resource extraction activities like dredging. The demand for offshore marine sand is increasing in the United States (Drucker et al. 2004), and sand is commonly used for beach renourishment, barrier island restoration, and wetland restoration. 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. Sand shoals are often preferred resources because of the quantity of sand per unit area, and the dredging of Outer Continental Shelf (OCS) sand shoals is likely to increase in the future as demand increases due to renourishment cycles for beaches, emergency repairs of beaches after storms, and the projected effects of sea-level rise (Nairn et al. 2004). In particular, the nGoM coast benefits greatly from barrier islands that reduce storm surge (Grzegorzewski et al. 2011), and these islands require regular sediment replenishment. The Bureau of Ocean Energy Management (BOEM), as part of the US Department of the Interior, is responsible for the management and development of mineral resources on the OCS, including sediment resources. As demand for OCS sand 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 Essential Fish Habitat (EFH) (Michel et al. 2013).

As part of our project, we have identified shoal locations and developed a classification scheme based on expert opinion and shoal characteristics of interest (Volume 2). The classification includes characteristics hypothesized to be related to fish such as bottom current direction and velocity, slope, depth, sediment grain size, rugosity (i.e., depth heterogeneity), and shell presence. In addition to the effect of shoals themselves, these characteristics are largely untested in relation to fish and shrimp species. Here, our objectives were to:

1) Test for habitat relationships of brown, pink, and white shrimp with a broad suite of environmental factors, including geomorphology, oceanographic characteristics, and nearby ecosystems

2) Model the spatial distribution of white, brown, and pink shrimp with multiscale predictors

1.2 Methods

1.2.1 Study Area

The study area spanned the extent of the nGoM from Texas to Florida, USA. The landward boundary began with federally managed waters (3 nm from the shoreline of Louisiana, Mississippi, and Alabama; 9 nm from the shoreline of Texas and Florida) through the 50-m depth contour. More specifically, we defined the landward boundary of the study area by the 1953 Outer Continental Shelf Lands Act (OCSLA), which distinguishes Federal- and state-managed waters. The oceanic boundary of the study area was defined by a 50-m contour line from National Oceanic and Atmospheric Administration's (NOAA's) Coastal Relief Model (CRM) (NOAA National Centers for Environmental Information 2010). Only waters ≤ 50 m in depth were included in the study because our focus was on the potential impact of sediment dredging and the logistics of dredging limit potential areas to these shallow depths. The study area had a total surface area covered of 162,985 km^2. The benthic substrate of the area consists of unconsolidated sediments ranging from mud to gravel with natural patches of hard bottom reefs.

1.2.2 Biological Data

We obtained offshore shrimp data from fishery-independent trawl surveys of the Southeast Area Monitoring and Assessment Program (SEAMAP) in the nGoM. With the exception of minor changes in sample selection procedures in 2010 and thereafter, surveys have used similar gear, protocols, and a random stratified sampling design based on depth and shrimp statistical area since 1992 (Craig et al. 2005; Gulf States Marine Fisheries Commission 2017). From these data, we used the years of 2003–2017 because we wanted to depict current conditions as best as possible, and this timeframe still provided us with an adequate sample size. Trawl surveys from the SEAMAP program span the entire nGoM and are conducted in the summer (June–August) and fall (October–December) seasons. Trawl surveys targeted shrimp and groundfish using a 12.8-m net in the central and eastern nGoM and with a 6.1-m net near Texas. Surveys were conducted at all hours of the day. Complete counts of shrimp were conducted in summer surveys. In the fall, complete counts were conducted for samples < 22.7 kg. For larger catches, totals counts were estimated by extrapolating from a subset of the catch. We used the centroid of trawl survey tows to represent survey locations and to calculate environmental variables. We removed extremely long or short tow survey lengths, and subsequently, trawls ranged from 11–52 min and 1.0–5.2 km. We calculated the relative abundance of each shrimp species by calculating catch per unit effort (CPUE) as shrimp km^{-1} of survey. Because initial relative abundance models showed below-average results for pink shrimp, we used presence/absence data for modeling this species. Prior to analyses, we used Generalized Additive Models (GAMs) (knots=3) to explore the effects of trawl length and duration on the presence/absence or CPUE of species. All tests showed <1% of the deviance was explained by these effects except for the effect of length on white shrimp CPUE, which explained 3% of the deviance. Given these minimal effects, we proceeded with using CPUE and presence/absence data. We used summer (June–August) and fall (October–December) trawl surveys for analyses of shrimp.

Pink shrimp ranged throughout the nGoM, and we used the entire study area for the analysis. We restricted the analyses of white shrimp and brown shrimp to their primary geographic ranges because the inclusion of hundreds, or thousands, of absence points outside a species' range may skew the results towards predictor variables characterizing the species' geographic range rather than their fine-scale distribution and habitat associations. Additionally, our goal was to add to existing knowledge of distributions to aid in management applications. For white shrimp, we excluded trawl surveys east of a longitude of W 87.9°. Only 1 of 1,653 trawl surveys east of this latitude recorded a white shrimp, and this longitude was > 200 km from that presence location. For brown shrimp, we excluded trawl surveys east of a longitude of W 84.5° because only 6 of 1,203 locations (8 of 570,733 individuals) were found east of this longitude. The next nearest brown shrimp in a trawl survey east of this longitude was > 100 km away.

1.2.3 Geographic Information System (GIS) Methods and Environmental Data

We converted all fish survey and environmental GIS data to the North America Albertson Equal Area Conic map projection. To calculate environmental variables with focal statistics (e.g., mean depth within a 3-km radius), we initially included a 5-km buffer of the study area. We removed this buffer after final maps were developed. Because we anticipated hierarchical relationships of shrimp with predictor variables (e.g., broad oceanographic factors and fine-scale substrate factors and depth), we predicted all SDMs to a 90-m resolution raster. The 90-m resolution was the same as several, but not all environmental variables. For data initially at a resolution > 90 m, we conducted a bilinear resampling and then aligned the data to the other 90-m datasets.

We developed predictor variables to depict oceanographic conditions, substrate, geography, and nearby ecosystems (**Table 1-1**). For bathymetry, and variables derived from bathymetry, we used the CRM (NOAA National Centers for Environmental Information) for offshore waters of Texas, Louisiana, Mississippi, and Alabama. Offshore of Florida, we observed large errors in depth values that spanned tens of kilometers and were usually observed as rectangular boxes with little variability. Therefore, we used sounding data developed into a 50-m raster grid by the US Geological Survey (Robbins et al. 2007). To be consistent with the regional analysis, we resampled to a 90-m resolution using bilinear interpolation. To correspond to the approximate length of trawl surveys, we used ArcGIS focal statistics to calculate the CV (coefficient of variation) of depth, mean slope, mean sediment grain size, proportion of area with shoal, and proportion of area with a positive bathymetric position index (BPI) within a 3-km radius.

Table 1-1. Predictor variables developed to predict the distribution of brown, pink, and white shrimp in the nGoM.

Oceanographic predictors were obtained from aggregations of monthly means spanning 2003–2017 unless otherwise noted.

Variable type	Predictor variable (units)	Resolution	Data source
Substrate	CV of depth	90 m	CRM + modifications
Substrate	Distance to shoal (km)	90 m	Pickens and Taylor, NOAA Biogeography Branch, identification of shoals
Substrate	Proportion of area with shoal	90 m	Pickens and Taylor, NOAA Biogeography Branch, identification of shoals
Substrate	Mean sediment grain size (mm)	370 m	Chris Jenkins, University of Colorado, interpolation of usSEABED data
Substrate	Proportion of area with BPI ≥1	90 m	CRM + modifications
Substrate	Slope (degrees)	90 m	CRM + modifications
Oceanographic	Mean depth (m)	90 m	CRM + modifications
Oceanographic	Bottom temperature (℃)	4.4 km	HYCOM + NCODA
Oceanographic	Chlorophyll-*a* (mg m^{-3})	5.5 km	Aqua MODIS satellite, 8-day composites
Oceanographic	Bottom salinity (psu)	4.4 km	HYCOM + NCODA
Oceanographic	Bottom current, U- and V-directional velocity (m s^{-1})	9.3 km	HYCOM + NCODA
Oceanographic	Mixed layer thickness (m) (depth where temperature change from surface is 0.2℃)	4.4 km	HYCOM + NCODA
Oceanographic	Hypoxia (mean probability of hypoxia)		North Carolina State University
Geography	East or West of W 88º longitude	90 m	
Geography	Distance to shoreline (km)	90 m	Submerged Lands Act
Nearby ecosystems	Nearby wetlands (km^2)	90 m	National Wetlands Inventory
Nearby ecosystems	Nearby estuaries (km^2)	90 m	National Wetlands Inventory

* HYCOM + NCODA = HYbrid Coordinate Ocean Model + Navy Coupled Ocean Data Assimilation; MODIS = Moderate Resolution Imaging Spectroradiometer

We used the ArcGIS Benthic Terrain Modeler (Wright et al. 2012) to calculate the slope and BPI. The BPI is an index that represents underwater hill and valley topography, with values ≥ 1 indicating a cell is higher than surrounding cells and a BPI ≤ -1 indicating a cell is lower than surrounding cells. We used the BPI to calculate the proportion of area as a topographic high. For mean sediment grain size, we used a data interpolation of usSeabed from NOAA/National Ocean Service (NOS) National Centers for Coastal Ocean Science (NCCOS) (Kinlan et al. 2013). Aspect, BPI classified as a valley, and sediment grain size classes were initially explored but did not provide further information beyond slope, sediment grain size, and shoal features that were already depicted.

We obtained oceanographic predictor variables by using Duke University's Marine Geospatial Ecology Toolbox (Roberts et al. 2010) within ArcGIS to summarize monthly means. Variables included bottom water temperature, chlorophyll-*a* (chlorophyll), bottom salinity, mixed layer thickness, and bottom current velocity for U and V directions (**Table 1-1**). All measures were averaged monthly over the period extending from January 2003 through December 31, 2017. Monthly measures were then averaged by seasons: spring = March 1–May 31, summer = June 1–Aug 31, fall = Sept 1–Nov 30, and winter = Dec 1–Feb 29. We used only remote sensing data to characterize oceanographic conditions because these measures are consistent across the geography and variables represented a relatively long-term characterization of the water column (i.e., monthly averages rather than a single day of a specific survey). Therefore, oceanographic measures represented spatial tendencies and related ecological processes rather than instantaneous conditions. Data on bottom temperature, bottom salinity, and bottom currents (U- and V- directions) were obtained from the HYCOM + NCODA Gulf of Mexico 1/25 degree analysis (GLMI0.04) (Chassignet et al. 2009). The Marine Geospatial Ecology Toolbox derived values calculated each day from the following datasets that have identical spatial extents and resolutions:

- 1 January 2003–30 April 2009: HYCOM dataset expt_20.1
- 1 May 2009–1 April 2014: HYCOM dataset expt_31.0
- 2 April 2014 and later: HYCOM dataset expt_32.5

The v-directional raster characterized the north (+) and south (-) currents and the u-directional raster characterized the east (+) and west (-) currents.

To quantify hypoxia (≤ 2 mg L^{-1} dissolved oxygen), we used the results of Matli et al. (2018), who modeled the extent and probability of hypoxic conditions for each year based on three separate data collection efforts. To obtain a relative index of long-term conditions, we used the mean probability of hypoxia for July and August from 2003–2017 as a predictor variable. As the data were initially points, we created an interpolated raster dataset by using ordinary, spherical kriging with calculations including eight adjacent points. Because the dataset ended abruptly at an area of high hypoxia probability, we used the ArcGIS "Expand" tool to extrapolate all values up to 35 km.

Geographic and nearby ecosystem predictor variables considered included distance to shoreline, nearby wetland area, nearby estuarine area, and a longitudinal threshold that depicted whether the location was east (1) or west (0) of the W 88º longitude near Mobile Bay, Alabama, USA. Waters west of this longitude are dominated by mud sediments, large river influences, lower salinities, and higher chlorophyll concentrations that extend farther offshore compared to eastward waters. Waters east of this longitude are dominated by sandy substrates and higher salinities. The abundance of brown shrimp (Montero et al. 2016) and juvenile red snapper (Dance and Rooker 2019) have been shown to differ at this longitude threshold. To determine if coastal wetlands contributed to the distribution of shrimp in the marine environment, we used National Wetlands Inventory (NWI) data (U.S. Fish and Wildlife Service 2018) and its classification of "estuarine and marine wetland". To quantify nearby wetland area (km^2), we first calculated the farthest distance from a wetland in the study area. Using the resulting 160 km distance, we then used ArcGIS focal statistics to sum all wetland cells (90 m resolution) within 160 km of a cell in the marine environment. The metric was converted to wetlands km^2. Unfortunately, we could not use NWI data to define estuaries because of large areas of missing data. Therefore, we defined nearby estuary area from a map of EFH from red drum *Sciaenops ocellatus* (NOAA NMFS n.d.) that characterized all estuaries in the nGoM. As the layer included some nearshore waters, we removed all waters that were seaward of the shoreline position. Similar to nearby wetlands, estuary cells within 160 km of each cell were summarized for the study area. Distance to shoreline was calculated from the boundaries of the Submerged Lands Act (Bureau of Ocean Energy Management 2010; 2012), which depicts a distance of 3 or 9 nm from shoreline, depending on the state. We used the buffer tool to re-create the approximate shoreline boundaries and then calculated the Euclidean distance from the shoreline to each cell in the study area.

1.2.4 Statistical Analysis

We examined predictor variables for multicollinearity, and we removed highly correlated variables (r > 0.80) prior to analyses. In an initial analysis of pink shrimp, we found a strong negative relationship with nearby estuaries. This was likely a result of pink shrimp being common on the Florida shelf, which coincides with low estuarine areas. Because this relationship was likely because of other correlates on the Florida shelf (e.g., sand sediments, higher salinity), we did not include nearby estuaries as a predictor of pink shrimp. In addition to environmental predictors, we used season (summer or fall trawl survey) and start time of surveys as predictors. The time at which surveys are conducted can affect the detectability of species and has previously been documented as affecting brown shrimp catch (Craig and Crowder 2005). Regarding seasonality, changes in summer and fall distributions are likely because of species' natural history. We did not use year as an explanatory factor because the primary objective of our research was to determine long-term value of waters and substrates of the nGoM. Therefore, we assume years of high or low abundance are representative of long-term shrimp distribution.

Our trawl survey dataset had a large number of locations, which led to some being in close proximity to others. Individual surveys are likely to be independent over the 15 years of data collection, but we wanted to ensure our models were robust to specific survey locations. Fourcade et al. (2018) found that a purely random split of training and validation data for SDM assessments led to a high validation accuracy assessment for models derived from fake predictors. They found a "checkerboard" approach of aggregated training or validation locations was helpful, and a "block" approach was best at distinguishing models as being poor when they were truly poor (Fourcade et al. 2018). Similarly, we aggregated training and validation data with alternating bands along a longitudinal gradient (**Figure 1.1**). The use of a longitudinal gradient maintained a depth gradient in each block. More specifically, we reclassified a raster of longitude into 70 equal interval divisions, which resulted in 23-km bands across the study area. We then alternated the delineation of training (two bands) and validation (one band) datasets to achieve the desired ratio of training and validation data.

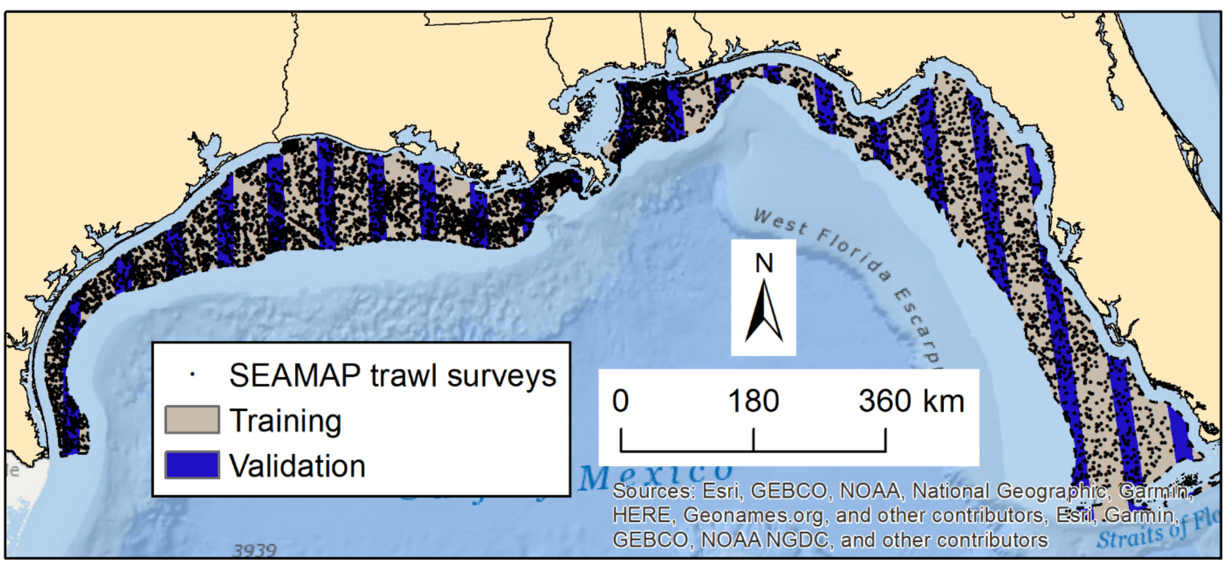

Figure 1.1. The study area with training and validation zones depicted with trawl surveys overlaid.

We used boosted regression trees (BRT) to model species-habitat relationships with the training data. In comparative studies of SDM statistical methods, BRTs and similar techniques have outperformed generalized linear models (GAMs), Maxent, and other techniques (Couce et al. 2012; Rooper et al. 2017; Smolinski and Radtke 2017). Classification and regression decision trees are the basis of BRT. Decision tree analyses are ideal for quantifying nonlinear relationships and complex interactions, which are both inherent in ecological data (De'ath and Fabricius 2000). The predictive power of decision trees are enhanced by boosting, which sequentially adds trees that improve the model; the results are then derived from an ensemble of hundreds of trees (De'Ath 2007; Elith et al. 2008). We followed the general procedures outlined by Elith et al. (2008) to develop BRTs. We iteratively assessed tree complexities of 1–5 and used learning rates that resulted in > 1,000 trees. The tree complexity represents the level of interaction allowed to occur (e.g., 1 = no interaction effects, 2 = interaction between two variables), and the number of trees are iterations. For brown and white shrimp, we used a Poisson log-linear model using CPUE as the dependent variable. For pink shrimp, we used a binomial model to predict probability of presence. We followed the BRT model "simplification" procedure described by Elith et al. (2008). In this procedure, the weakest predictor is dropped sequentially, predictors are ranked in order of importance, and the change in model deviance with each drop is assessed. The inflection point where the model's deviance sharply increases after a drop defines which variables remain in the model with the goal of having a parsimonious model without losing predictive power. To be consistent, we defined an inflection point as an increase of > 2% of the deviance explained when dropping a single variable and ≥ 3% for multiple dropped variables.

To assess accuracy of relative abundance models, we report the percent deviance explained (similar to an r^2 for Poisson regression) from the cross-validation and validation datasets. A Spearman rank correlation (R_s) was also calculated between observations and predictions of the validation dataset. For the pink shrimp presence/absence model, we used a receiver operator characteristic, area under the curve statistic (AUC). The AUC has been commonly used to test predictive ability of SDMs (Guisan and Zimmermann 2000) and is independent of thresholds. Measures of the AUC range from 0.0 to 1.0 and were interpreted as suggested by Manel et al. (2001) and Swets (1988) as follows: < 0.50 = no discriminatory power; 0.50–0.69 = poor power; 0.70–0.89 = good power; and 0.90–1.0 = excellent discriminatory power. In addition, we report the confusion matrix for the presence/absence model at the probability of presence threshold determined by the maximum Kappa that optimally discriminates presence and absence. We report relative importance of each variable in the model as suggested by Elith (2008). The relative importance of all variables in the model sums to 100%. Likewise, we assessed the strength of interaction, and these results are on a continuous scale with zero showing no interaction effect. We report interaction effect (IE) measures with a score of > 10 because these were most straightforward to interpret.

We used the statistical program R (R Core Team 2018) and the package "dismo" (Hijmans et al. 2017) to implement BRT. To predict models to the extent of the study area, or geographic range of species, we used the R packages "rgdal" (Bivand et al. 2019) and "raster" (Hijmans 2019). We assumed the effect of time of survey represented a detectability effect rather than a change in distribution. Therefore, we applied the models during each species' peak time of detectability: 10:30 for brown shrimp, 02:00 for white shrimp, and 06:00 for pink shrimp.

1.3 Results

Brown and white shrimp models explained a substantial amount of variation in CPUE, and the pink shrimp model was very good at predicting presence/absence (**Table 1-2**). Brown shrimp were present on 76.3% of trawl surveys with a total of 570,725 individuals. Mixed layer depth (MLD) in the spring and summer were most influential in the model with a deeper mixing of water related to a higher brown shrimp CPUE (**Figure 1.2 and Figure 1.3**). Brown shrimp CPUE was lowest from 11:00 through 24:00, and the season effect showed CPUE was greatest during summer. We found a slight positive relationship of brown shrimp with fall bottom temperature, and nearby wetlands had a positive effect on CPUE. There was a strong interaction between spring MLD and nearby wetlands (IE = 41), which showed the highest predicted CPUE where high nearby wetland area was combined with a relatively deep spring MLD.

Table 1-2. BRT specifications and measures of accuracy for the shrimp species distribution models.

Species	*n*	Tree complexity	Learning rate	# of trees	Cross-validation	Validation	Validation Spearman correlation
Brown shrimp	4,417	4	0.02	1,850	Deviance explained = 45%	Deviance explained= 37%	0.64
White shrimp	3,967	5	0.01	1,200	Deviance explained = 41%	Deviance explained = 30%	0.55
Pink shrimp	5,620	3	0.01	2,000	AUC = 0.84	AUC = 0.85	NA

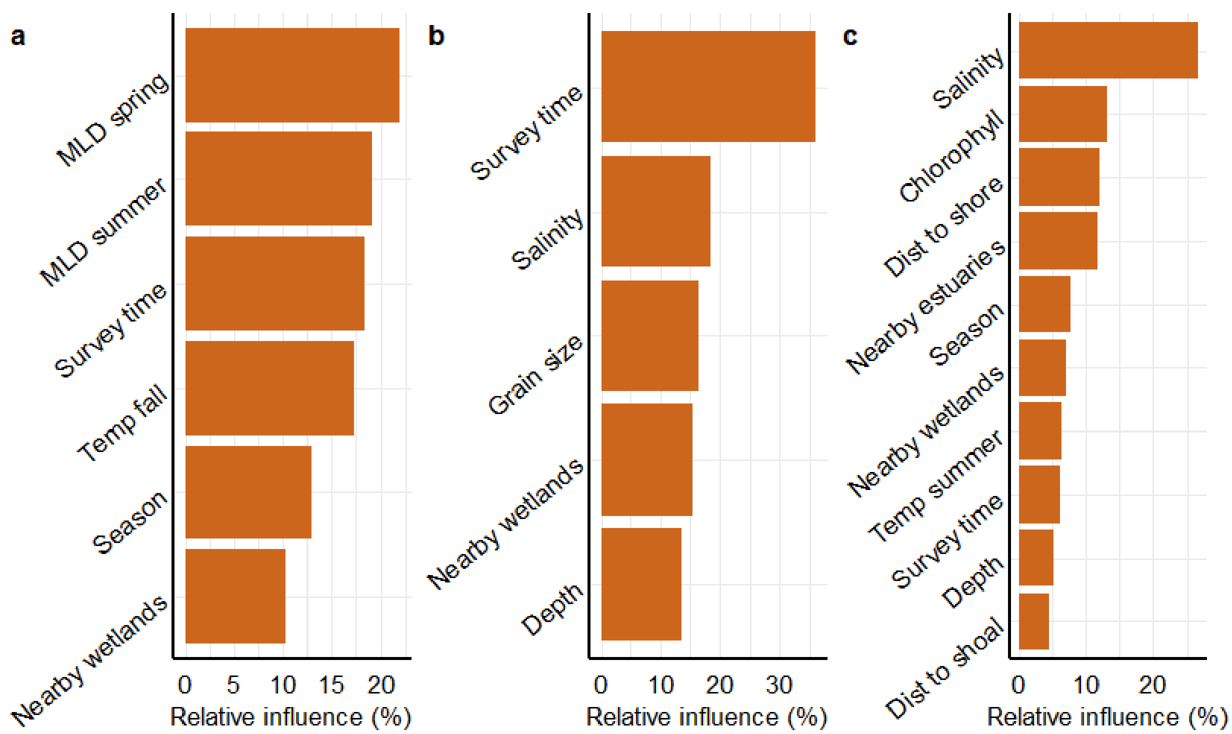

Figure 1.2. Relative importance of variables from the a) brown shrimp, b) pink shrimp, and c) white shrimp SDMs for the nGoM.
MLD = mixed layer depth, Temp = bottom water temperature, Grain size = sediment grain size, Dist = distance

White shrimp were present on 39.1% of trawl surveys, with a total of 44,455 individuals. White shrimp were most abundant with salinities of 22–30 psu, high chlorophyll concentrations, within 30 km of the shoreline, and with greater amounts of nearby estuaries and wetlands (**Figure 1.2 and Figure 1.4**). They had a higher CPUE during the fall season, with higher summer bottom temperatures, and with depths of approximately 15–30 m; they had a lower CPUE near midnight. White shrimp were slightly more abundant farther away from shoals. There was an interaction between salinity and nearby wetland area (IE = 17) that showed CPUE was greatest where high salinity coincided with a high nearby wetland area. The lowest predicted CPUE was at lower salinities with few nearby wetlands. An interaction between chlorophyll and season (IE = 13) showed chlorophyll had a greater influence on CPUE during the fall.

Pink shrimp were present on 22.6% of trawl surveys with a total of 36,015 individuals. Surveys from 11:00 to 23:00 had an extremely low probability of presence (**Figure 1.2 and Figure 1.4**). Pink shrimp presence was associated with salinities spanning 31–36 psu and substrate with sand grain sizes (less likely with silt or granule gravel grain sizes) (see Wentworth 1922). The relationship with nearby wetland area showed substantial variability, but pink shrimp were less likely when wetland area was extremely low and when wetland area was very high. Pink shrimp had the highest likelihood of presence with depths of 18–30 m, although considerable uncertainty existed at shallow depths. Survey time interacted with both salinity (IE = 65) and wetlands (IE = 65). This showed that these habitat variables had the greatest effect during times when pink shrimp were catchable. The pink shrimp model discriminated absence much better than presence (**Table 1-3**), and the overall accuracy represented by the AUC score was very good. The spatial models of the three species showed distinct patterns (**Figure 1.3 to Figure 1.5**).

Table 1-3. Confusion matrix from the validation data of the pink shrimp model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.38.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	1,323	180	88%
Predicted presence	132	241	65%

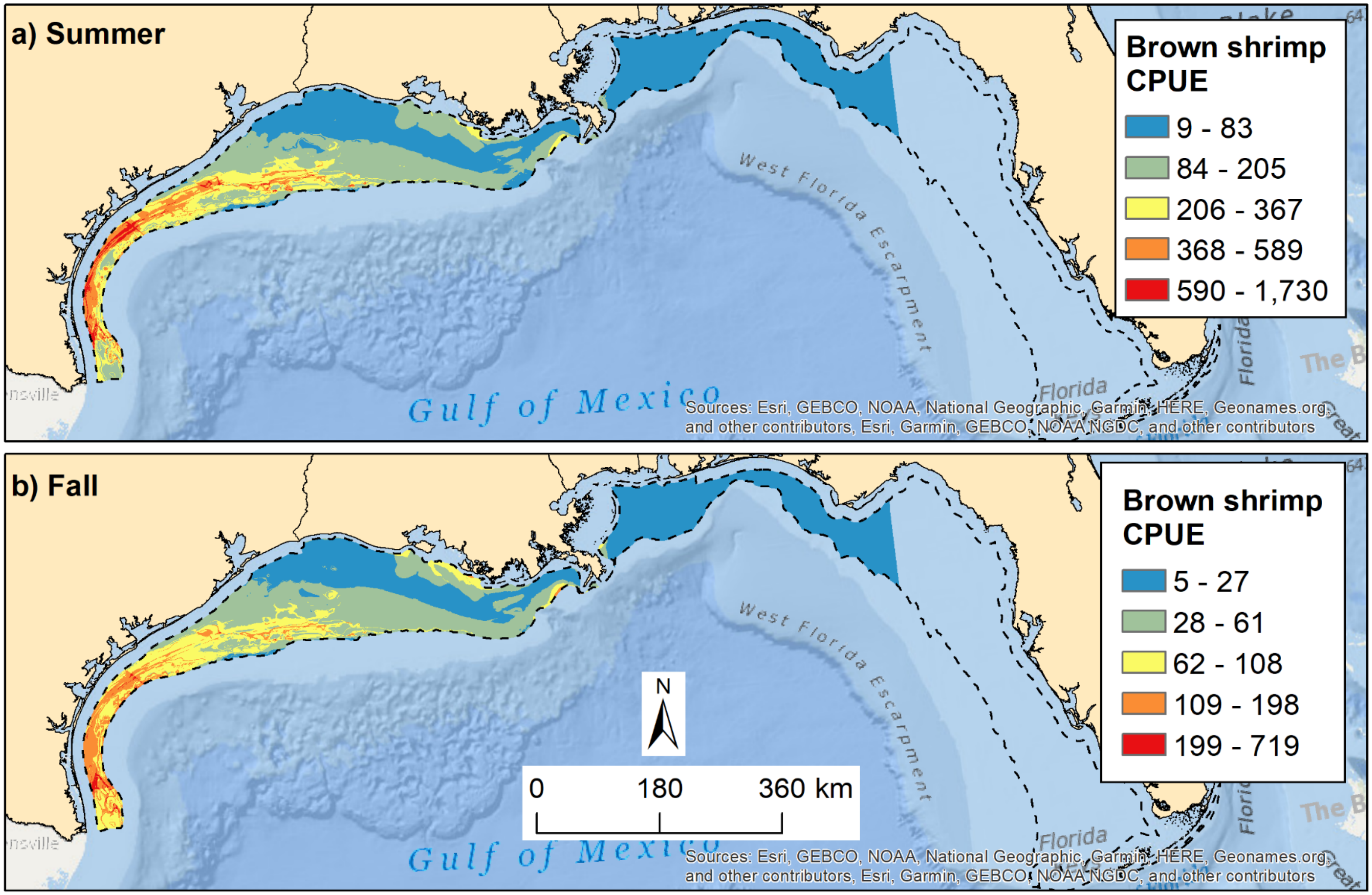

Figure 1.3. Predicted brown shrimp CPUE in the a) summer and b) fall seasons.
The study area is indicated by the dashed line, and CPUE represents the predicted number of shrimp per km of trawl survey.

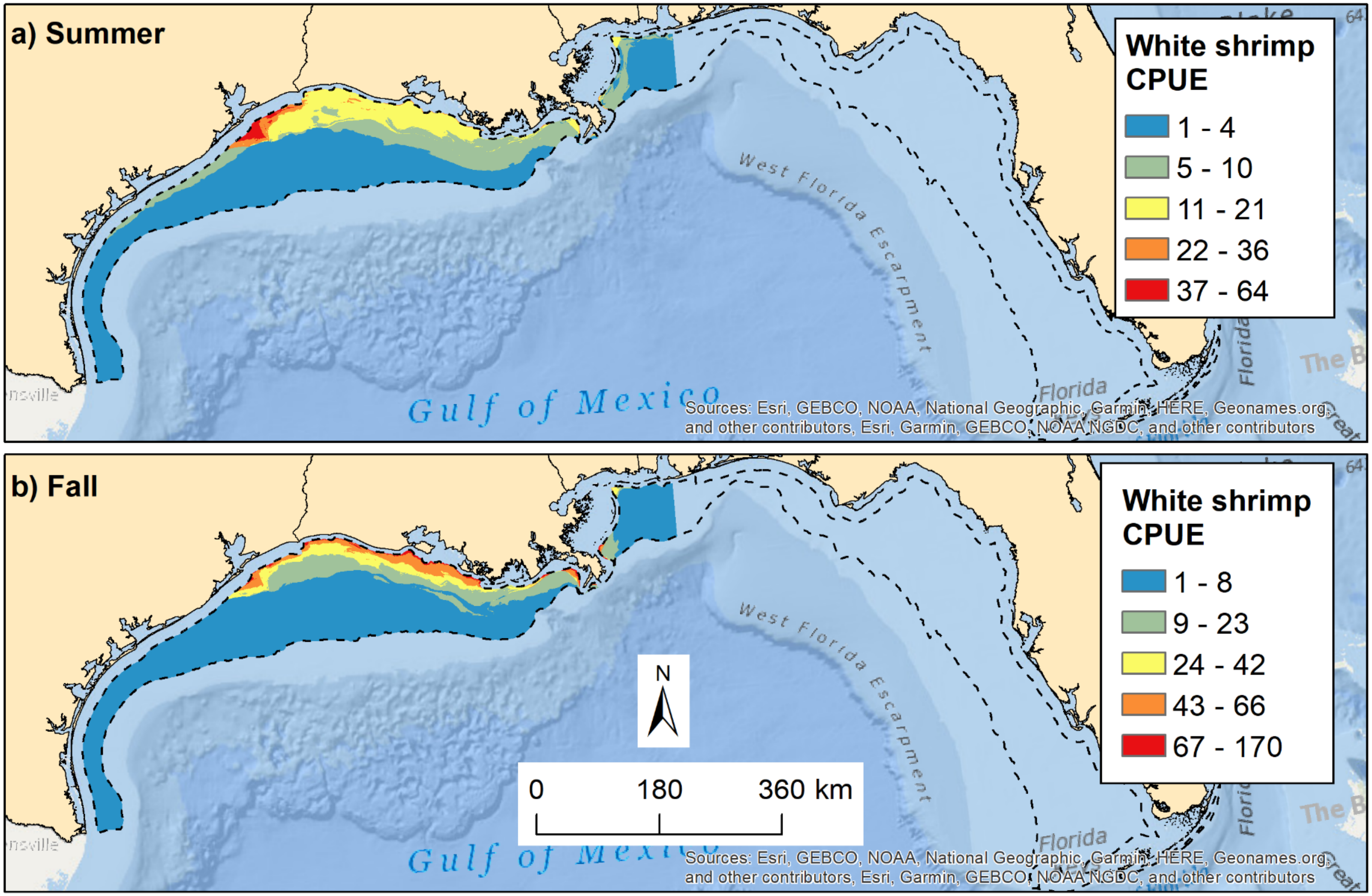

Figure 1.4. Predicted white shrimp CPUE in the a) summer and b) fall seasons.
The study area is indicated by the dashed line, and CPUE represents the predicted number of shrimp per km of trawl survey.

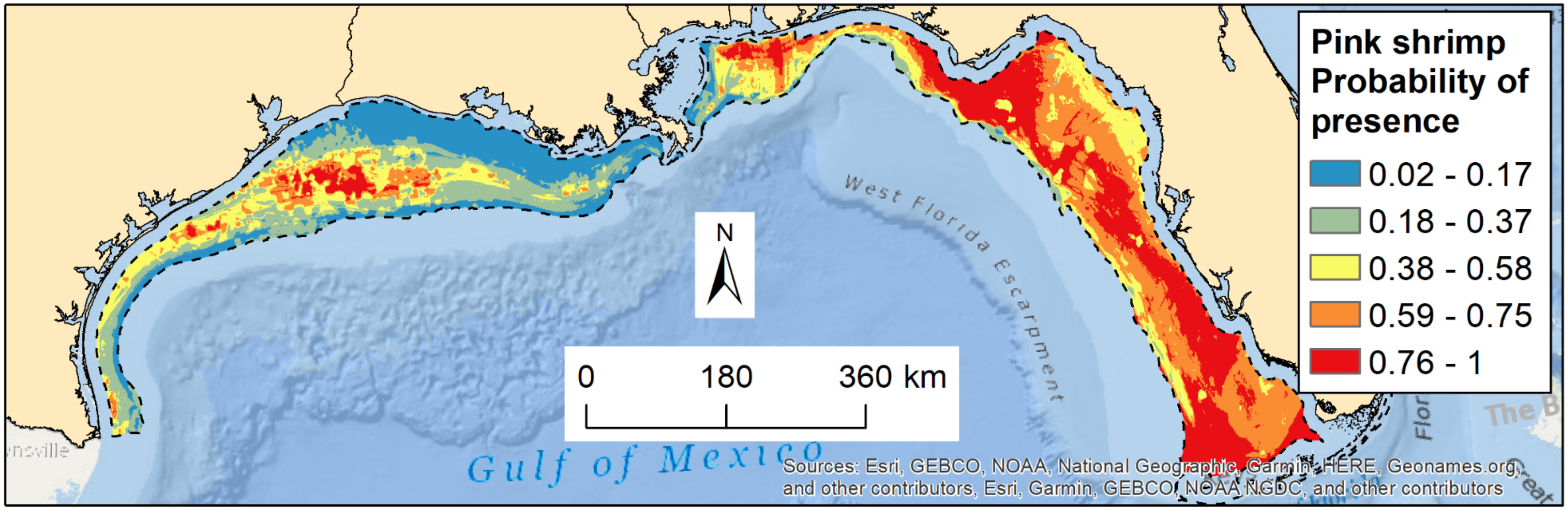

Figure 1.5. Predicted pink shrimp probability of presence in the summer and fall combined.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a trawl survey.

1.4 Discussion

Few studies have linked marine and terrestrial ecosystems despite important linkages such as between coastal estuaries and offshore marine habitats (Beger et al. 2010). By analyzing three Penaeid species, we found a consistent pattern that the distributions of these species in the offshore, marine environment were positively related to the amount of nearby coastal wetlands and estuaries. In fact, brown and white shrimp both had interaction effects that included nearby wetlands and an oceanographic factor (MLD and salinity, respectively). These interactions directly demonstrate the linkages between these two ecosystems. Wetland loss and the loss of open water-marsh edges have been independently identified as contributors of shrimp declines in estuarine (Rozas et al. 2007) and marine environments over time (Diop et al. 2007). However, the relationships we quantified further suggest that wetland loss is likely to lead to changes in shrimp distribution in the marine environment.

None of the three Penaeid species' distributions were related to bottom currents or geomorphology metrics of slope, BPI, or CV of depth. Of the geomorphology variables, only distance to shoal and sediment grain size were selected as predictors, and the importance of these variables were relatively minor. Predicted white shrimp CPUE was less in close proximity to shoals and pink shrimp were positively associated with sand grain sizes. Similar to findings of Drexler and Ainsworth (2013) and a qualitative assessment by Mulholland (1984), we found pink shrimp occurred with sand substrates and had a lower probability of occurrence with mud, silt, or gravel substrates. In contrast to Montero et al. (2016), we did not find brown shrimp were associated with mud and silt sediments. However, we excluded trawl data offshore of Florida because it appeared to be outside of the geographic range of the species. If we had included the hundreds of absence locations in those eastern nGoM waters with sand substrates, then the results may have been similar to Montero et al. (2016). By analyzing the three Penaeid species separately, we observed the spatial distinctions among species. Brown shrimp had the highest CPUE in the western nGoM, while pink shrimp were most common east of the W 88º longitude near Mobile Bay, Alabama. White shrimp mostly occurred near the shoreline of the central nGoM.

1.4.1 Brown Shrimp

A greater CPUE of brown shrimp in summer coincides with their peak spawning season in the nGoM (Gulf of Mexico Fishery Management Council 1981). Notably, there was no direct relationship with the high frequency hypoxic zone. Nonetheless, the predicted CPUE of brown shrimp was extremely low in the hypoxic zone. This is likely because brown shrimp had a higher CPUE with a deeper MLD, and these waters tended to occur outside of the high frequency hypoxic zone. The association with fall bottom temperature is also associated with farther offshore habitats. Craig et al. (2005) found brown shrimp move both farther inshore and offshore when hypoxia is severe (i.e., leaving the intermediate depth zone), and this spatial pattern is observed in our model of brown shrimp distribution. We note that our data describes the long-term average brown shrimp distribution rather than annual events that other studies have considered. Harvest strategies for shrimp also do differ in Louisiana and Texas. In Louisiana, shrimp trawlers work in the most shallow waters (< 20 m depth) near estuaries earlier in the year. For Texas, harvest does not occur until later and shrimp trawlers use areas > 20 m in depth (Craig et al. 2005).

1.4.2 Pink Shrimp

The time of day for surveys was influential for pink shrimp and brown shrimp, as CPUE was far higher from approximately midnight to 11:00 for these species. In a lab study, pink shrimp buried themselves in response to daylight and were active in dark conditions (Hughes 1968). Hughes (1968) also found individuals synchronized emergence within a 20–30 minute timeframe. Studies of other Penaeids worldwide have also documented this phenomenon (Wassenberg and Hill 1994). Our pink shrimp model showed the association with salinity resulted in a distribution skewed towards Florida, although pink shrimp were also predicted to be in other parts of the central nGoM where sand sediments were present. The interaction effects among habitat variables and the time of surveys suggests that surveys at ineffective times are likely to complicate the quantification of species-habitat relationships. The confusion matrix for pink shrimp showed absence can be more accurately predicted compared to presence. This result is expected given that species are detected imperfectly in surveys (see MacKenzie et al. 2002 for details on the topic), and our results depict the predicted catch in a trawl survey. In essence, a species may be present even though a single survey did not detect that species. Applications of these SDMs should consider the results as relative rather than absolute numbers. Waters with a greater probability of occurrence or relative abundance (CPUE) of species should be interpreted in the context of waters with lower probability of occurrence or relative abundance.

1.4.3 White Shrimp

White shrimp CPUE was much higher in the coastal zone where salinity was < 30 psu. White shrimp selected waters closer to the shoreline, with a greater chlorophyll-*a* concentration, and depths of 12–30 m. White shrimp also had a positive association with summer bottom temperatures that were > 28°C. Research on the temporal dynamics of white shrimp in Louisiana coastal waters showed greater water temperatures related to higher abundances of early juveniles, and to a lesser extent, more adult white shrimp. A greater CPUE of white shrimp in the fall coincides with their peak spawning season in the nGoM (Gulf of Mexico Fishery Management Council 1981).

1.4.4 Conclusions and Implications for Dredging

In summary, we have identified habitat associations and modeled the distribution of three federally managed Penaeid shrimp species. Nearby wetlands and oceanographic characteristics, such as bottom temperature, salinity, MLD, and chlorophyll drove the majority of the spatial patterns for these species. Bottom currents, slope, and depth heterogeneity were not associated with species' distributions. In the cases where sediment grain size and distance to shoal were associated with species, the variables explained only a small portion of the variance compared to other variables. We do caution that fishery-independent trawl surveys typically sample ~3 km tows, and microhabitats within these areas were not assessed by our study. Shoals themselves are poorly sampled because of their depth, and, therefore, samples near shoals were assumed to represent those areas. Importantly, the spatial models do show strong spatial patterns that will help inform the identification of important waters for Penaeid species, and their predators, in the nGoM. Broad spatial patterns typical of oceanographic predictors as well as habitat associations with grain size and distance to shoal will help guide impact assessments at a local level, such as at individual dredge sites.

2 Predicting the Marine Distribution of Snappers and Sharks in the Northern Gulf of Mexico

2.1 Introduction

SDMs and similar methods of spatial modeling have proliferated in the last two decades as GIS technologies, remote sensing, availability of spatial data, and computation capacity have rapidly improved. Recent distribution models within marine ecosystems have been applied to ecosystem-based management (Gruss et al. 2018), marine spatial planning (Hattab et al. 2013), scenario assessments (Delevaux et al. 2018), stock assessments (Saul et al. 2013), and climate change (Morley et al. 2018; Su et al. 2013). SDMs can illuminate habitat relationships as well as map the spatial patterns of species' distributions. In turn, mapping is critical for the identification of EFH (Moore et al. 2016; Pennino et al. 2016), defined as those waters and substrates required for a species to spawn, breed, feed, or grow to maturity (US Sustainable Fisheries Act, 1996, Public Law 104-297). EFH is inclusive of prey species, although we are not aware of EFH mapping that explicitly includes the distribution of prey species.

A decade after Robinson et al. (2011) declared that marine ecosystems provide a prime opportunity to test the influence of prey on species' distributions, few studies have explored this possibility (see Volume 2). The nGoM presented an opportunity to test predator-prey relationships with data on a variety of trophic levels. Food webs in the nGoM have been summarized by Tarnecki et al. (2016), including common linkages among Penaeid shrimp, menhaden, squid, and other small demersal and pelagic fish. In particular, the abundance of Gulf menhaden has been shown to have major effects on fisheries (Geers et al. 2016; Robinson et al. 2015). Estuaries, subaquatic vegetation, oyster reefs, and coastal wetlands in the nGoM also contribute towards offshore marine productivity by providing habitat for estuarine-dependent life stages and producing common prey species of the marine environment (e.g., menhaden, shrimp, crabs) (Spies et al. 2016). Much like the effect of prey distribution, the effect of such nearby ecosystems has rarely been considered in SDMs.

In addition to these knowledge gaps, we previously have documented biases towards testing particular habitat variables for particular fish guilds (Volume 1). More specifically, there has been a trend to test numerous substrate predictor variables—but few oceanographic variables—with reef fish, such as snapper. Conversely, the trend for sharks is to test more oceanographic variables, but few substrate variables. Therefore, testing a comprehensive suite of habitat variables has the potential to test these biases and lead to an improved understanding of EFH.

We selected snapper and shark species to study based on their breadth of designated EFH, overlap of a species' EFH with our study area (Federal waters, ≤ 50 m), socio-economic importance, data availability, and potential vulnerability to sand dredging (i.e., demersal species with an affinity to soft sediments). Below, we outline characteristics of each of the selected species:

Sharks:

- Blacktip shark is a large coastal shark that prey on teleost fishes (Cortés 1999) and is listed as globally "vulnerable" by the International Union for Conservation of Nature (IUCN) (Burgess and Branstetter 2000). In the nGoM, juvenile blacktip shark diet is dominated by Gulf menhaden (Bethea et al. 2004). Blacktip shark is the second most valuable shark to commercial fisheries of the southeastern US (Castro 1996).

- Spinner shark is listed as globally "near threatened" by the IUCN (Burgess 2009) and is a common target by commercial fisheries. Spinner shark primarily prey on teleost fishes (Cortés 1999), particularly Gulf menhaden (Bethea et al. 2004).

- Atlantic sharpnose shark is a relatively small, demersal shark that feeds on crustaceans and teleost fishes (Cortés 1999). In the nGoM, Atlantic sharpnose are regularly caught in recreational and commercial fisheries.

Snappers:

- Red snapper juveniles are demersal before inhabiting natural and artificial reefs as adults (Gallaway et al. 2009). In 2016, commercial landings totaled $26.5 million in the nGoM (NOAA NMFS Office of Science and Technology 2019); the species is also important for recreational fisheries.

- Lane snapper is a subtropical, reef-associated species that has a demersal juvenile life stage that inhabits shallow waters with sand/mud bottoms, including shoal habitats (Mikulas and Rooker 2008; Wells et al. 2009). Commercial landings of lane snapper are modest, primarily in Florida, where landings totaled $86,219 in 2016 (NOAA NMFS Office of Science and Technology 2019). They are a regular recreational catch offshore of Florida as well.

The objectives of our study were the following:

1) Test for habitat relationships of snapper and coastal shark species with a broad suite of environmental factors, including prey, nearby ecosystems, geomorphology, and oceanographic characteristics.

2) Model the spatial distribution of snapper and shark species.

2.2 Methods

2.2.1 Study Area

The study area is described in Section 1.2.1.

2.2.2 Biological Data

Fish data were derived from fishery-independent surveys of SEAMAP and the National Marine Fisheries Service (NMFS) Mississippi Laboratories, which included regular surveys as well as a 1-year survey from the Congressional Supplemental Sampling Program (CSSP) (**Table 2-1**). From these data, we only used data from 2003 and later because we wanted to depict current conditions as best as possible, and this timeframe still provided us with an adequate sample size.

Trawl surveys from the SEAMAP program span the entire nGoM and are conducted in the summer (June–August) and fall (October–December). With the exception of minor changes in sample selection procedures in 2010 and thereafter, surveys have consistently used similar gear and protocols, and a random stratified sampling design based on depth and shrimp statistical area (Craig et al. 2005; Gulf States Marine Fisheries Commission 2017). Trawl surveys targeted shrimp and groundfish using a 12.8-m net in the central and eastern nGoM and a 6.1-m net near Texas. Trawls were conducted consistently during day and night hours. For processing, complete counts were conducted for samples < 22.7 kg, and a sample was taken and the count extrapolated for most species if the sample was larger. Snapper in summer surveys were counted in their entirety. In the fall, only a portion of the biological catch was measured for large samples (> 22.7 kg). For those surveys, the full trawl catch was projected based on a measured sample (e.g., when only half catch was sampled, the number of individuals for each species was doubled). We assumed that this proportionate extrapolation applied to each age group.

Table 2-1. Sources of fisheries-independent survey data spanning 2003–2015.
Sample sizes are provided for waters within our study area.

Data source	Acronym for dataset	Dates	Gear used for survey (*n* = sample size)	Species modeled
Southeast Fisheries Science Center, NMFS, Mississippi Laboratory (MSLABS)	MSLABS-BL	2003-2017; 8 April–16 Nov	Bottom longline; *n* = 1,014	Atlantic sharpnose shark, blacktip shark, and spinner shark
Congressional Supplemental Sampling Program (CSSP)	CSSP-BL	2011; 7 April–25 Oct	Bottom longline; *n* = 498	Atlantic sharpnose shark, blacktip shark, and spinner shark
SEAMAP-trawl	SEAMAP trawl	2003-2017; 30 May–19 Dec	Trawl; *n* = 5,620	Red snapper (age 0 & age 1), lane snapper (age 0)

We removed extremely long or short trawl survey lengths, and subsequently, trawls ranged from 11–52 min and 1.0–5.2 km. We transformed trawl survey counts to CPUE, calculated as fish km^{-1} of survey. Prior to analyses, we used GAMs (knots=3) to explore the effects of trawl length and duration on the presence/absence or CPUE of species. All tests showed <2% of the deviance was explained by these effects except for a negative association with red snapper age-0 (2.7%) and a positive association with lane snapper age-1 (4.5%). Given these mixed effects, we proceeded with using CPUE and presence/absence data. We used summer (June–August) and fall (October–December) trawl surveys.

Surveys from the CSSP-Bottom Longline (CSSP-BL) and MSLABS-BL programs used the same sampling methodology. The methods are described in detail by Driggers III et al. (2012) and are outlined here. All bottom longline surveys used a 15/0 circle hook baited with Atlantic mackerel. The MSLABS-BL and CSSP-BL surveys were randomly placed throughout the nGoM. For BL surveys, gear soak times were targeted to be 1 hour, as defined by the time elapsed between completion of deployment and initiation of retrieval. We used the centroid of trawl tows and bottom longline surveys to depict fish survey locations. For bottom longline surveys, we removed extreme survey efforts of ≥134 min, and survey effort ranged from 37–107 min. We also removed surveys with low hook counts (<80 hooks); the remaining surveys had 86–104 hooks per survey. Fish from bottom longline surveys were measured as count 100 $hooks^{-1}$ hr^{-1}. We used GAMs (knots=3) to explore the effects of survey length (km) and duration (min) on each species. All tests showed < 2% of the deviance were explained by these factors.

2.2.3 Fish Age Classification

The age of red and lane snapper were distinguished for the analysis because these species undergo ontogenetic shifts between early juvenile stages and the adult stage, when they inhabit reefs. The vast majority of sharks were juveniles and adults, both of which occur in similar habitats. Therefore, we did not distinguish age for sharks, but simply had the goal to estimate the proportion of each life stage present. During trawl surveys, fish lengths were measured as total length (TL), fork length (FL), or standard length (SL). We converted all lengths to TL for age classification. For lane snapper, we used the conversion equations from the species' stock assessment (SEDAR 2016). We categorized age classes based on a Bermuda study of lane snapper because the study measured numerous juveniles (Luckhurst et al. 2000), which were not well described elsewhere. Luckhurst et al. (2000) found a minimum length of 185 mm FL (199 mm TL) for age-1 individuals. Based on this finding, we defined lane snapper age-0

(< 199 mm TL) and age-1 or greater (≥ 199 mm TL). Given the dramatic reduction in growth after age one (Luckhurst et al. 2000), we did not distinguish further age groups.

For red snapper length conversions, we used equations from the Gulf of Mexico red snapper stock assessment report (SEDAR 2018). Red snapper of 19–50 mm TL are distinguished as post-settlement juveniles (Gallaway et al. 2009), and their prey items differ from older age classes that begin to prey on fish (Szedlmayer and Lee 2004). Because only 0.01% of red snapper captured by trawl were <50 mm TL, we discarded this age class from the analysis. Red snapper < 172 mm TL have mostly been found in open habitats, whereas the majority of those ≥ 172 mm inhabit reef habitats (Szedlmayer and Lee 2004). This threshold is further supported by Powers et al. (2018), who aged red snapper captured in trawl surveys and found age-0 red snapper = 30–170 mm, age-1 = 175–297 mm, and age-2 = 320–360 mm TL. Therefore, we categorized red snapper into the following juvenile age classes (range of catch 18–893 mm TL):

Age-0 = 51–172 mm TL (*n* = 25,528)
Age-1 = 173–300 mm TL (*n* = 5,670)

The majority of sharks were measured by natural total length (nat TL). When only FL was measured, we converted it to nat TL using equations dervived from available MSLABS-BL data. For Atlantic sharpnose, the conversion equation was: nat TL mm = 57.308 + 1.118 (FL mm); r^2= 0.952. For blacktip shark, the equation was nat TL mm = 44.129 + 1.162 (FL mm); r^2= 0.954. For spinner shark, the equation was TL mm = 25.811 + 1.170 (FL mm); r^2 = 0.972). To further classify the age classes of sharks surveyed, we investigated the literature on length-age class associations. For Atlantic sharpnose shark, we followed the classifications used by Drymon et al. (2012) and Hoffmayer and Parsons (2003), who used the following age classes: Young-of-year = 330–590 mm nat TL, Juvenile = 600–840 mm nat TL, Adults ≥ 850 mm nat TL. Castro (1996) reported blacktip shark neonates ranged from 530–660 mm nat TL offshore of Florida, and we used 660 mm nat TL as the maximum length to classify young-of-year blacktip shark. To distinguish juveniles from adults, we classified adults as those ≥ 1,407 mm nat TL. This is the median length reported for female blacktip sharks at maturity (Carlson et al. 2006). Carlson et al. (2006) showed male median length at maturity was slightly lower (1,246 mm TL); therefore, our estimate is likely to be slightly biased towards juveniles rather than adults. However, not all sharks were sexed, so detailed evaluation could not be conducted. With an approximation based on Carlson's study (Carlson and Baremore 2005), we defined those spinner shark < 70 cm as young-of-year, 70–116 cm as juvenile, and those > 116 cm as adults.

Of the sexed blacktip sharks, 54% were females and 46% were males. Blacktip shark lengths ranged 656–2,466 mm nat TL with a median of 1,350 mm nat TL in MSLABS-BL data and 1,252 mm nat TL in CSSP surveys. Of blacktip sharks with an age classification, 0.002% (3 individuals) were classified as young-of-year, 65% were juvenile, and 35% were adults. Of the sexed Atlantic sharpnose sharks, 49% were males and 51% were females. Atlantic sharpnose shark length ranged 340–1,223 mm nat TL with a median of 920 mm nat TL in MSLABS-BL surveys and 895 mm nat TL in CSSP surveys. Of Atlantic sharpnose sharks with an age classification, 2% were young-of-year, 27% were juvenile, and 71% were adults. Of sexed spinner shark, 49% were female and 51% were male. Spinner shark age classification showed 3% as young-of-year, 43% as juvenile, and 54% as adults.

2.2.4 GIS Methods and Environmental Data

All GIS analysis procedures and data collection methods were followed as described in **Section 1.2.3** with the exceptions described below. Overall, we developed predictor variables to depict oceanographic conditions, geomorphology, nearby ecosystems, geography, and prey species (**Tables 2-2**, **2-3**). Natural reef locations were mapped during SEAMAP reef fish video surveys and were acquired through several mechanisms, including available charting, historical knowledge from fishermen, and bathymetric mapping (i.e., side-scan sonar and multi-beam sonar) (pers. communication, Matthew Campbell and

Brandi Noble, NOAA National Marine Fisheries). The point density of and distance to artificial structures, including artificial reefs as well as oil and gas platforms, were calculated for the study area. Sea surface temperature (SST) variables were obtained as predictors using the same methods as used for other oceanographic data (**Section 1.2.3**). We used SST data processed from GHRSST (Group for High Resolution Sea Surface Temperature) version 4.1 of the Multiscale Ultrahigh Resolution Level 4 analysis, which obtains high resolution data via a blend of satellite measures (JPL MUR MEaSUREs Project 2015).

Prey species were first identified from the literature. Then, we developed predictor variables for prey that were either readily sampled by the SEAMAP trawl surveys or were available from models developed during our project (**Table 2-3**). We derived data on menhaden (primarily Gulf menhaden), croaker, spot croaker, mantis shrimp, and squid from SEAMAP trawl data. The SEAMAP survey locations were interpolated to create a continuous surface. We synthesized data on prey species from 2003–2017 and conducted ordinary kriging with a spherical semivariogram model. Eight points were used for analysis within a maximum distance of 10 km. The ArcGIS expand tool was used to further extrapolate prey distributions when trawl surveys were > 10 km from a location in the study area.

Table 2-2. Environmental variables developed to predict snappers and sharks in the nGoM.
Oceanographic predictors were obtained from aggregations of monthly means spanning 2003–2017.

Variable type	Variable (units)	Radius of analyses (km)	Resolution of data	Data source
Substrate	CV of depth	3	90 m	CRM + modifications
Substrate	Density of oil platforms & artificial reefs (structures km^{-2})	5	90 m	BOEM, Marine Cadastre
Substrate	Distance to oil platforms & artificial reefs (km)	NA	90 m	BOEM, Marine Cadastre
Substrate	Distance to natural reef (km)	NA	90 m	Matthew Campbell, NOAA National Marine Fisheries
Substrate	Distance to shoal (km)	NA	90 m	Pickens and Taylor, NOAA Biogeography Branch, identification of shoals
Substrate	Proportion of area with shoal	3	90 m	Pickens and Taylor, NOAA Biogeography Branch, identification of shoals
Substrate	Mean sediment grain size (mm)	3	370 m	Chris Jenkins, University of Colorado, interpolation of usSEABED data
Substrate	Proportion of area with BPI ≥ 1	3	90 m	CRM + modifications
Substrate	Slope (degrees)	3	90 m	CRM + modifications
Oceanographic	Mean depth (m)	3	90 m	CRM + modifications
Oceanographic	Sea surface temperature (℃)	NA	1.2 km	GHRSST blend of satellite measures, MUR-JPL_L4-GLOB-v4.0, NASA Jet Propulsion Lab
Oceanographic	Bottom temperature (℃)	NA	4.4 km	HYCOM + NCODA
Oceanographic	Chlorophyll-*a* (mg m^{-3})	NA	5.5 km	Aqua MODIS satellite, 8-day
Oceanographic	Bottom salinity (psu)	NA	4.4 km	HYCOM + NCODA
Oceanographic	Mixed layer thickness (m) (depth where temperature change from surface is 0.2℃)	NA	4.4 km	HYCOM + NCODA
Oceanographic	Bottom current velocity—U & V directions(m s^{-1})	NA	9.3 km	HYCOM + NCODA
Oceanographic	Hypoxia (mean probability of hypoxia)	NA	90 m	North Carolina State University
Geography	East or west of longitude W 88º	NA	90 m	-
Geography	Distance to shoreline (km)	NA	90 m	-
Nearby ecosystems	Nearby wetlands	NA	90 m	NWI
Nearby ecosystems	Nearby estuaries	NA	90 m	NWI

* HYCOM + NCODA = HYbrid Coordinate Ocean Model + Navy Coupled Ocean Data Assimilation; GHRSST = Group for High Resolution SST Level 4 analysis, Multiscale Ultrahigh Resolution (MUR) based on nighttime; MODIS = Moderate Resolution Imaging Spectroradiometer; CRM = Coastal Relief Model; NWI = National Wetlands Inventory

Table 2-3. Biological predictor variables developed to predict the distribution of snappers and sharks.

Species	Prey species predictor variables	Justification for prey inclusion
Red snapper age-0	Brown shrimp (fall), pink shrimp, mantis shrimp (*Squilla* spp.), squid (*Loligo* spp.)	(Bradley and Bryan 1975; Szedlmayer and Lee 2004; Wells et al. 2008a)
Red snapper age-1	Searobin (*Prionotus* spp.), lizardfish (*Synodus* spp.), squid	(Szedlmayer and Lee 2004; Wells et al. 2008a)
Lane snapper age-0	Brown shrimp (fall) & pink shrimp	(Franks and VanderKooy 2000)
Lane snapper age-1	None identified	NA
Atlantic sharpnose shark	Menhaden (*Brevoortia* spp.), croaker (*Micropogonias undulatus*), pink shrimp, brown shrimp (summer)	(Bethea et al. 2004; Drymon et al. 2012; Harrington et al. 2016)
Blacktip shark	Menhaden, croaker	(Barry et al. 2008; Bethea et al. 2004)
Spinner shark	Menhaden	(Bethea et al. 2004; Cortés 1999)

2.2.5 Statistical Analysis

We examined predictor variables for multicollinearity, and we removed highly correlated variables ($r > 0.80$) prior to analyses. In addition to environmental predictors, we used season (summer or fall) for trawl surveys of snappers, day-of-year for shark species, and start time of surveys as predictors. The time at which surveys are conducted can affect the detectability of species and has previously been documented as affecting shark catch (Driggers III et al. 2012). Seasonality of trawls represented changes in summer and fall distributions because of species' natural history. We did not use year as an explanatory factor because the primary objective of our research was to determine long-term value of waters and substrates of the nGoM. Therefore, we assume years of high or low abundance are representative of long-term fish distribution.

Because improving our understanding of species-habitat relationships was one of our objectives, the variables in shark and snapper models differed by a few variables. We excluded variables with no hypothesized relationship to particular fish species to help in minimizing issues with moderately correlated variables. Chlorophyll-*a*, nearby wetlands, and nearby estuaries were tested only for sharks. SST variables were only tested for blacktip shark and spinner shark, as the other species have demersal habits. We only used density of artificial structures, distance to artificial structures, and distance to natural reefs to test with snappers. To summarize predictor variable results, we calculated the frequency of each variable type as depicted in **Table 2-2**.

Training (70%) and validation (30%) data were selected by the same process as used with the statistical analyses of Penaeid shrimp (**Section 1.2.4**). The BRT analysis procedures outlined in **Section 1.2.4**. were used to quantify species-habitat relationships, relative importance of predictors, and interaction effects for snappers and sharks. We also assessed the accuracy of models using the same procedures. We assumed the effect of survey time represented a detectability effect for blacktip shark and applied the model at the peak time of 02:00. For Atlantic sharpnose shark, day-of-year was a factor in the model, and we predicted at the peak time of year in the model (9 April).

2.3 Results

The frequency of species ranged from red snapper age-1 on 18.7% of trawl surveys to Atlantic sharpnose shark on 58.6% of bottom longline surveys (**Table 2-4**). The relative abundance models for red snapper age-0 and Atlantic sharpnose shark explained 41–43% of the deviance in the validation data with an R_s of 0.59–0.60 in their respective models (**Table 2-5**). All species modeled with presence/absence data had an AUC value of ≥ 0.80 in the validation data, indicating the models were very good at discriminating presence and absence (**Table 2-4**). The confusion matrices showed that absence was consistently predicted more accurately than presence (**Table 2-6** to **Table 2-10**).

Across all snapper and shark models, 45 variables were selected across the following variable types: oceanographic (22 variables selected), prey (6), substrate (6), geographic (4), temporal variables (4), and nearby ecosystems (3). Of the 22 oceanographic predictors, the most common variables were MLD (6), bottom temperature (5), and salinity (5). Of the substrate variables, three were related to artificial or natural reefs; sediment grain size, BPI, and distance to shoal were each selected one time. Variable importance varied considerably among variables, though sediment grain size, BPI, and distance to shoal were of relatively minor importance (**Figures 2-1, 2-2**).

Table 2-4. Frequency of select snapper and shark species in the trawl and bottom longline surveys from 2003–2017.

Common name	Number of fishery-independent surveys (*n*)	Type of modeling	Percent of surveys with presence	Total count of species
Red snapper (age 0)	5,620	Count	36	23,076
Red snapper (age 1)	5,620	Presence/absence	18.7	4,753
Lane snapper (age 0)	5,620	Presence/absence	25.7	9,784
Lane snapper (age 1)	5,620	Presence/absence	20.3	1,143
Atlantic sharpnose shark	1,506	Count	58.6	8,765
Blacktip shark	1,506	Presence/absence	28.2	1,831
Spinner shark	1,506	Presence/absence	12.6	872

Table 2-5. BRT specifications and percent deviance explained for marine SDMs of snappers and sharks.

Species	Tree complexity	Learning rate	# of trees	Cross-validation	Validation	Validation Spearman correlation
Red snapper (age 0)	5	0.01	1,400	Deviance explained = 50%	Deviance explained = 41%	0.59
Red snapper (age 1)	2	0.01	1,550	AUC = 0.83	AUC = 0.80	NA
Lane snapper (age 0)	3	0.01	1,950	AUC = 0.84	AUC = 0.83	NA
Lane snapper (age 1)	2	0.02	2,550	AUC = 0.91	AUC = 0.89	NA
Atlantic sharpnose shark	5	0.005	1,900	Deviance explained = 45%	Deviance explained = 43%	0.60
Blacktip shark	1	0.02	1,250	AUC = 0.84	AUC = 0.80	NA
Spinner shark	2	0.005	1,400	AUC = 0.90	AUC = 0.87	NA

Table 2-6. Confusion matrix from the validation data of the red snapper age-1 model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.31.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	1,323	174	88%
Predicted presence	172	207	55%

Table 2-7. Confusion matrix from the validation data of the lane snapper age-0 model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.37.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	1,185	200	86%
Predicted presence	201	290	59%

Table 2-8. Confusion matrix from the validation data of the lane snapper age-1 model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.44.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	1,379	154	90%
Predicted presence	104	238	70%

Table 2-9. Confusion matrix from the validation data of the blacktip shark model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.47.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	323	61	84%
Predicted presence	34	67	66%

Table 2-10. Confusion matrix from the validation data of the spinner shark model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.33.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	333	76	81%
Predicted presence	24	52	68%

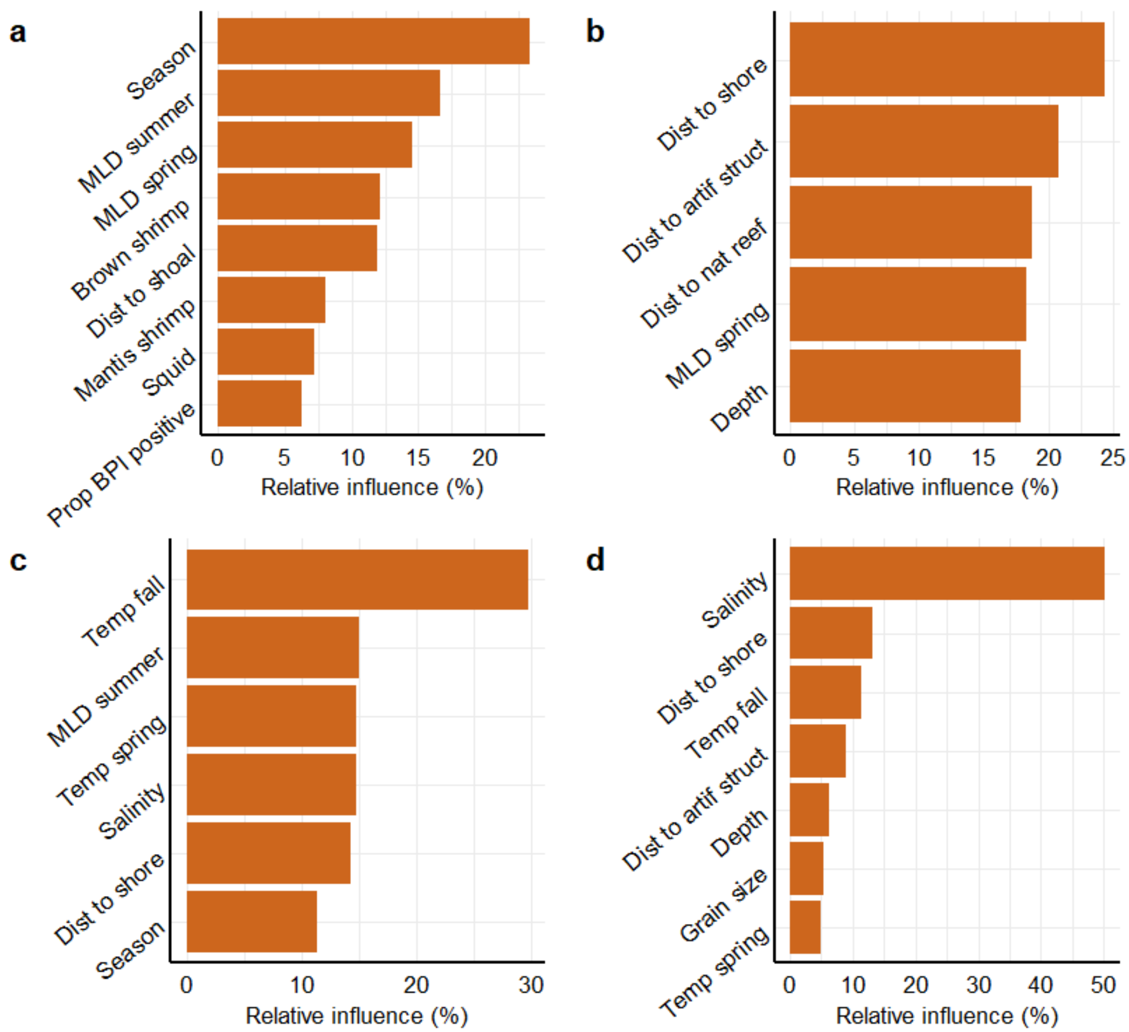

Figure 2.1. Relative importance of variables in models of a) red snapper age-0, b) red snapper age-1, c) lane snapper age-0, d) lane snapper age-1.

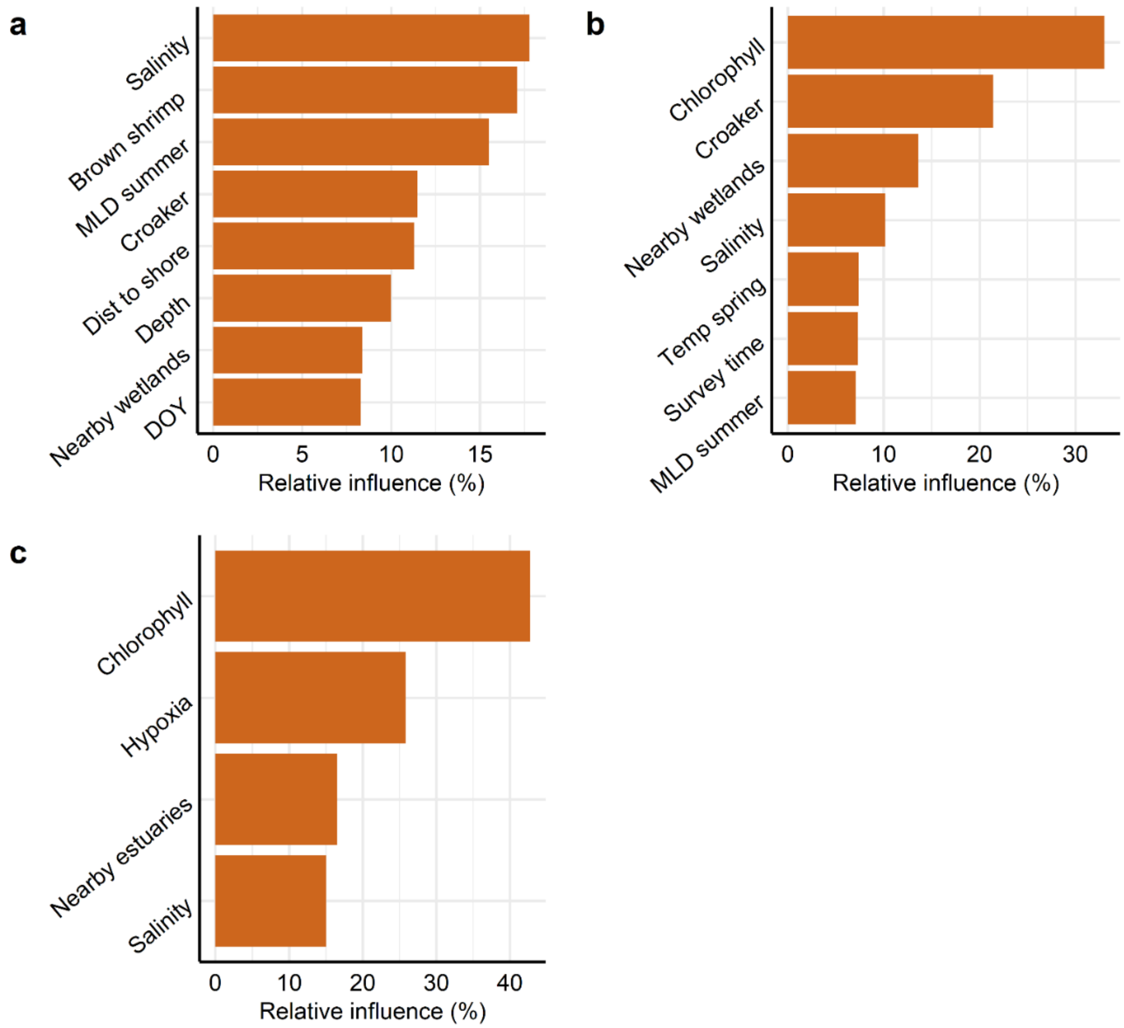

Figure 2.2. Relative importance of variables in models of a) Atlantic sharpnose shark, b) blacktip shark, c) spinner shark.

2.3.1.1 Red Snapper Age-0

As expected, red snapper age-0 (**Figure 2.3**) were more abundant in the fall as individuals become large enough to be captured by trawl sampling. MLD in the spring and fall had strong, positive relationships with red snapper. These young juveniles were positively associated with prey species of brown shrimp, mantis shrimp, and squid. In regard to geomorphology, red snapper age-0 had a higher CPUE within the first few km of shoals and with a high proportion of area with a positive BPI. An interaction between BPI and distance to shoal (IE = 49) showed red snapper had a high CPUE within a few km of a shoal regardless of BPI. At greater distances from shoals, the BPI had a linear, positive effect when the BPI had a positive value positive for > 40% of the surrounding area. MLD in summer primarily affected red snapper during the fall season (IE = 31), and CPUE was highest when a deeper MLD in spring was coincided with a higher brown shrimp CPUE (IE = 19).

2.3.1.2 Red Snapper Age-1

Red snapper age-1 (**Figure 2.4**) shifted farther offshore compared to age-0, but still had a positive association with spring MLD. Individuals moved closer to artificial structures (often within 25 km) and natural reefs (often within 75 km). In regard to interactions, red snapper age-1 moved farther offshore when artificial structures were within 20–40 km (IE = 44). Interpreting these distance from mapped variables, we observed that this meant red snapper age-1 moved farther offshore near Alabama, Mississippi, Louisiana, and Texas (where oil platforms and other artificial reefs were common), but individuals did not move farther offshore in Florida shelf waters. Distance to shoreline interacted with MLD spring (IE = 32) and showed MLD spring had the strongest effect farther from the shoreline. Red snapper age-1 had the highest probability of presence at greater depths up to a 50 m depth.

2.3.1.3 Lane Snapper Age-0

As expected, lane snapper age-0 (**Figure 2.5**) were more abundant in the fall as individuals become large enough to be captured by trawl sampling. Lane snapper age-0 had a higher probability of presence with higher fall and spring bottom temperatures. They had a higher probability of presence farther from the shoreline, with more shallow MLD, and with salinities in the range of 20-34 psu compared to salinities > 34 psu. An interaction between salinity and spring bottom temperature (IE = 61) showed low probability of occurrence at higher salinities (> 32 psu) except where spring bottom temperatures were > 22°C. Probability of occurrence was highest where a shallow MLD in summer coincided with high fall bottom temperatures (IE = 61).

2.3.1.4 Lane Snapper Age-1

Lane snapper age-1 (**Figure 2.6**) had a greater probability of presence in waters < 40 m in depth and with cooler spring bottom temperatures. They were less common with mud and silt substrates (particularly, < 0.03 mm grain size) and when grain sizes became larger than granule gravel (see Wentworth 1922). No interactions were observed.

2.3.1.5 Blacktip Shark

Blacktip shark (**Figure 2.7**) were positively related to chlorophyll and nearby wetlands; they had the highest probability of occurrence with low salinity waters spanning 27–34 psu. Blacktip shark were positively related to croaker prey, but did not directly select for their primary prey of menhaden. The species had a higher probability of occurrence where MLD in summer was relatively deep, and they were more likely to be caught from 00:00–03:00. Interactions were minimal.

2.3.1.6 Spinner Shark

Spinner shark (**Figure 2.8**) had the highest probability of occurrence with salinities of ≤ 30 psu, and they were positively related to chlorophyll and nearby estuaries. Spinner shark had a positive relationship with hypoxia, as probability of occurrence was highest in waters with 20–40% probability of hypoxia. We examined the hypoxia map, and the 20-40% range corresponded well with the edge of highest probability hypoxic zone. Spinner shark had a moderate interaction between hypoxia and nearby estuaries (IE = 38), showing that spinner shark selected waters with 20–40% hypoxia probability that coincide with a relatively high amount of nearby estuaries.

2.3.1.7 Atlantic Sharpnose Shark

Atlantic sharpnose shark CPUE (**Figure 2.9**) was positively related to the prey species of brown shrimp and croaker as well as the amount of nearby wetlands. Atlantic sharpnose shark CPUE was highest in the spring, farther from the shoreline, and at greater depths up to our 50-m maximum in the study area.

Bottom salinity showed a dichotomy where salinities of 27.5–30 psu and 33.5–36 psu had the highest CPUE, but intermediate salinities had fewer Atlantic sharpnose shark. Interaction effects were minimal.

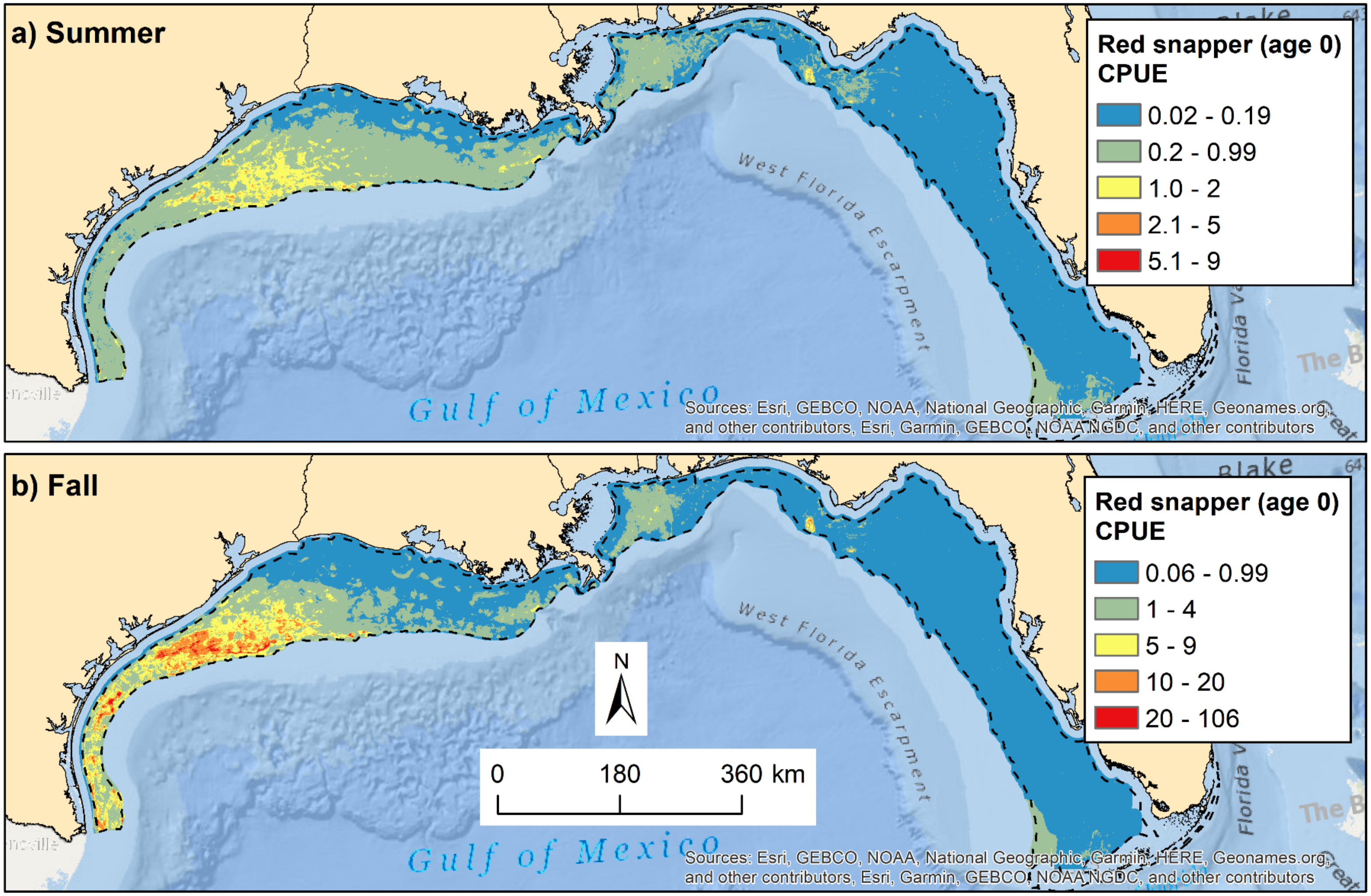

Figure 2.3. Predicted red snapper age-0 CPUE in the a) summer and b) fall seasons.
The study area is indicated by the dashed line, and CPUE represents the predicted number of red snapper per km of trawl survey.

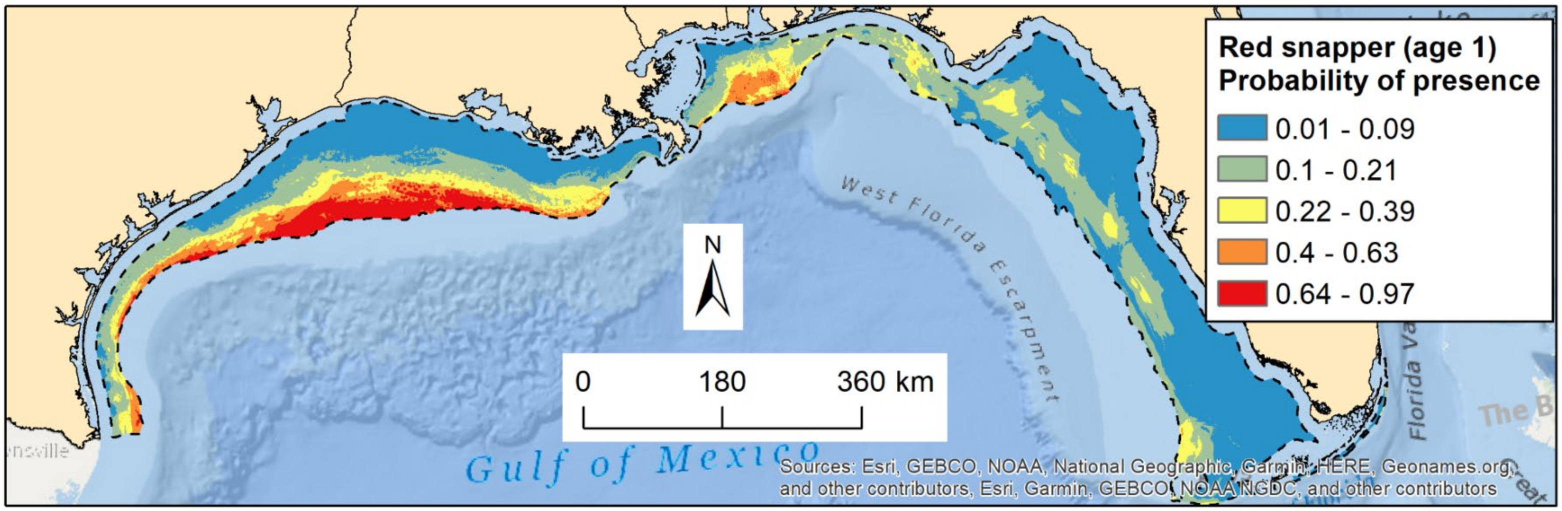

Figure 2.4. Predicted red snapper age-1 probability of presence in the summer and fall seasons combined.
The study area is indicated by the dashed line, and CPUE represents the predicted number of red snapper per km of trawl survey.

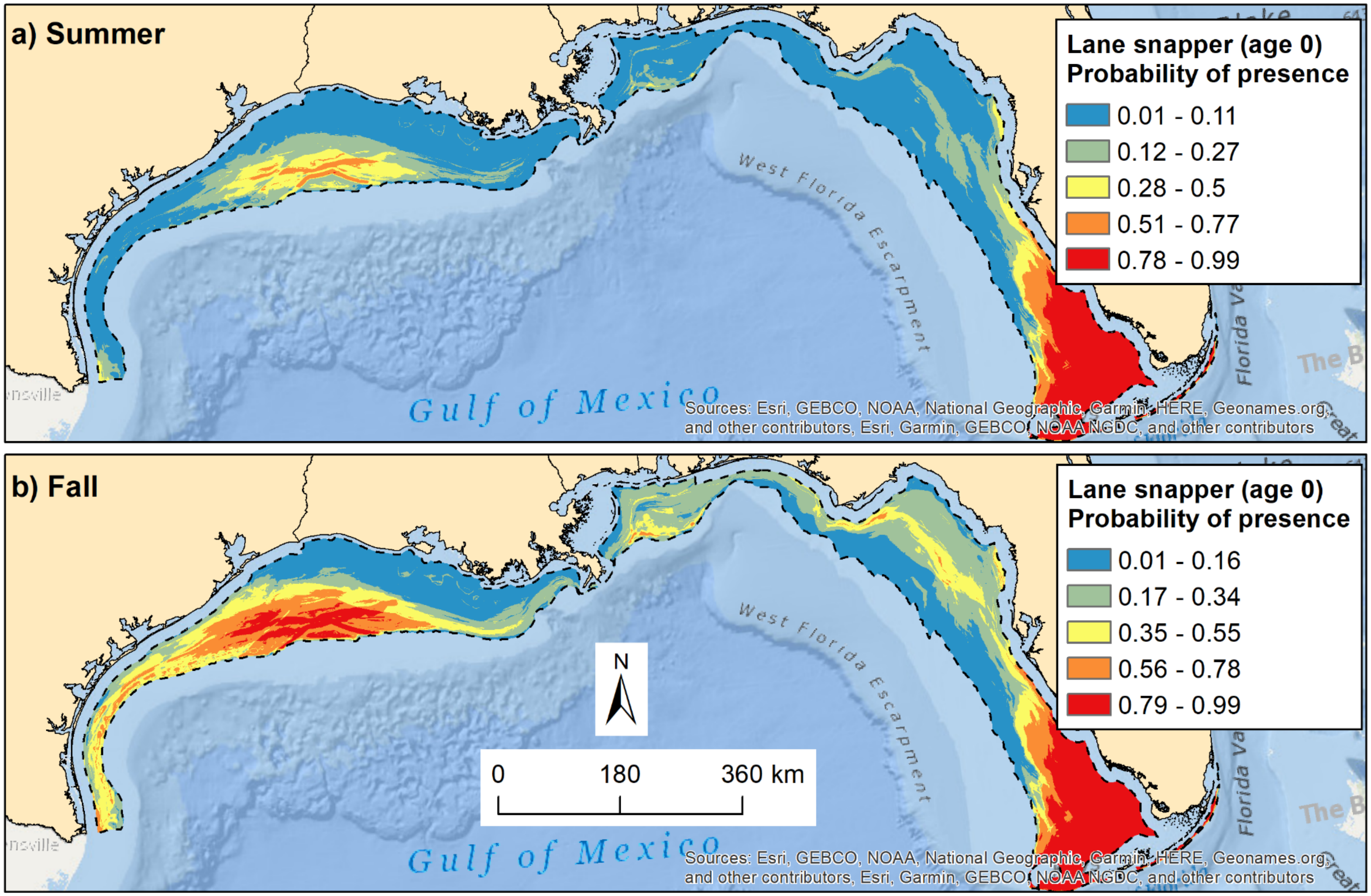

Figure 2.5. Predicted lane snapper age-0 probability of presence in the a) summer and b) fall seasons.
The study area is indicated by the dashed line, and CPUE represents the predicted number of lane snapper per km of trawl survey.

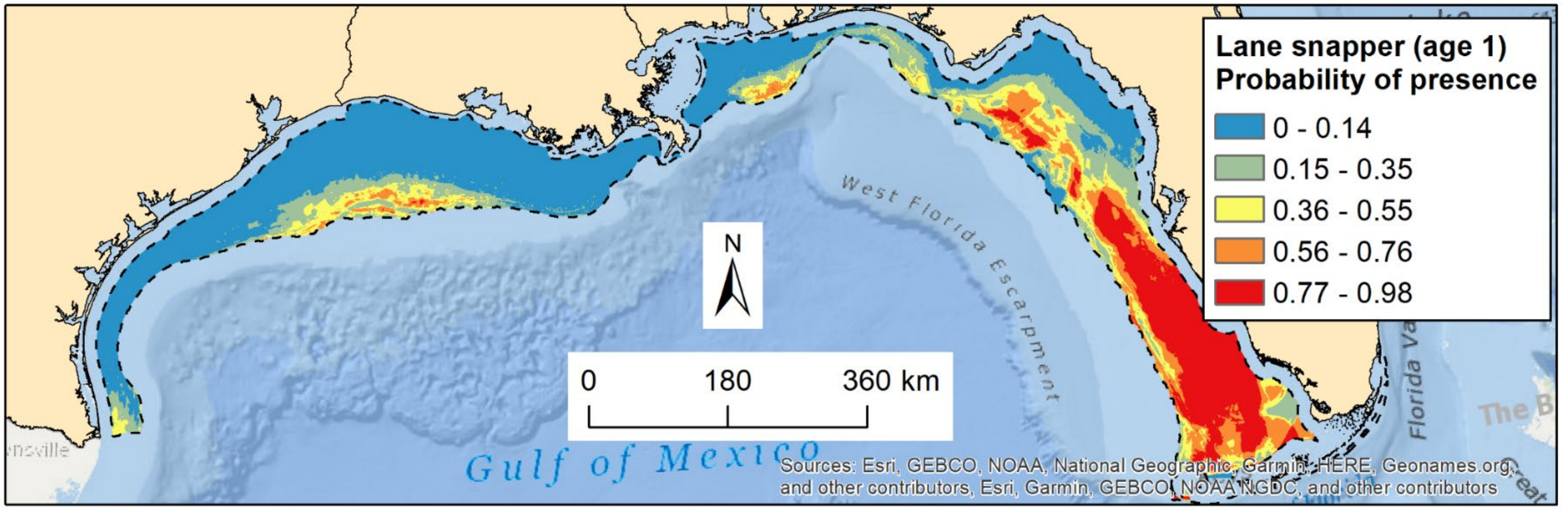

Figure 2.6. Predicted lane snapper age-1 probability of presence in the summer and fall seasons combined.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a trawl survey.

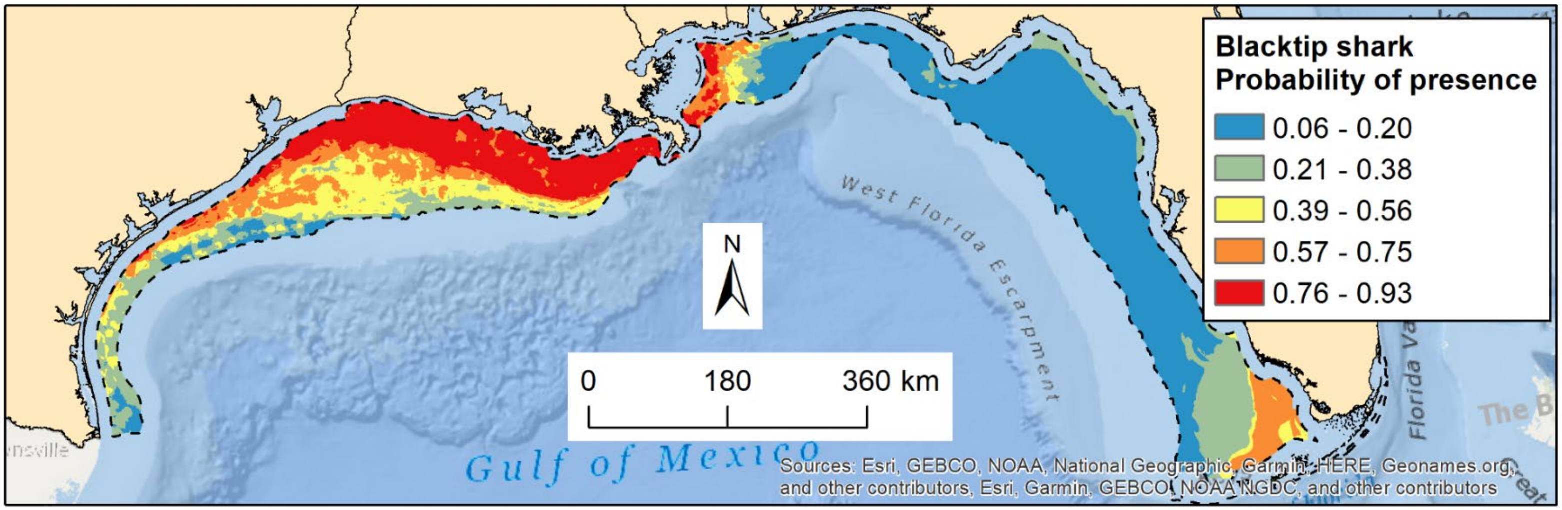

Figure 2.7. Predicted blacktip shark probability of presence in the spring–fall seasons.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a bottom longline survey.

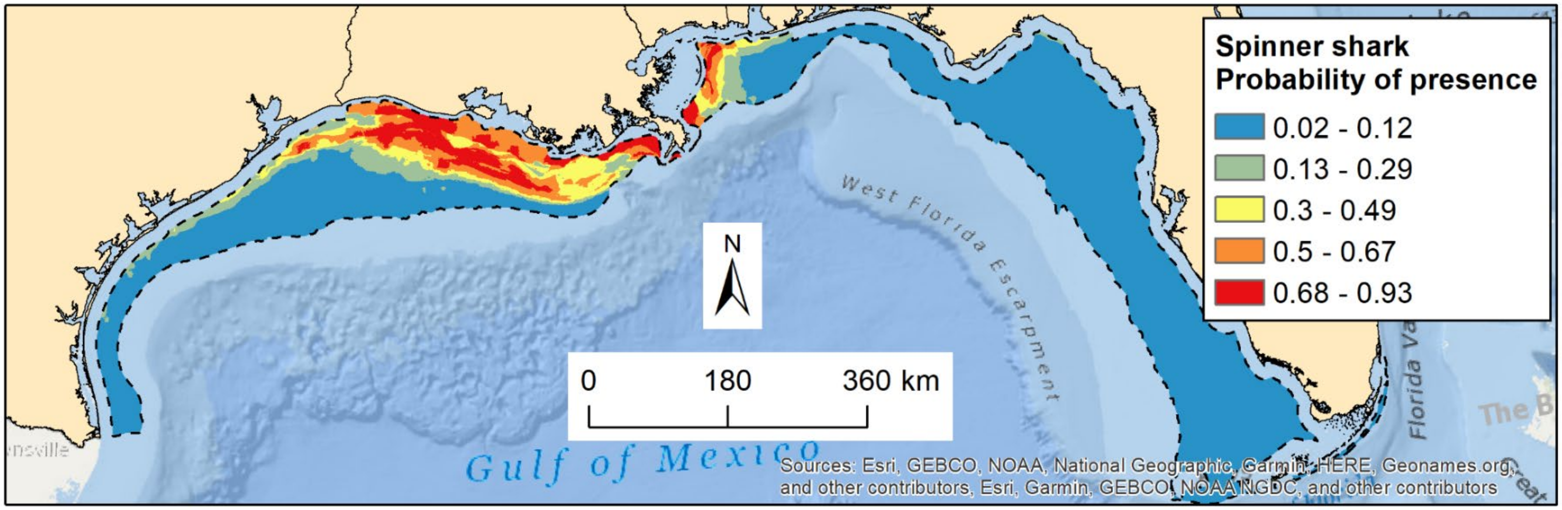

Figure 2.8. Predicted spinner shark probability of presence in the spring–fall seasons.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a bottom longline survey.

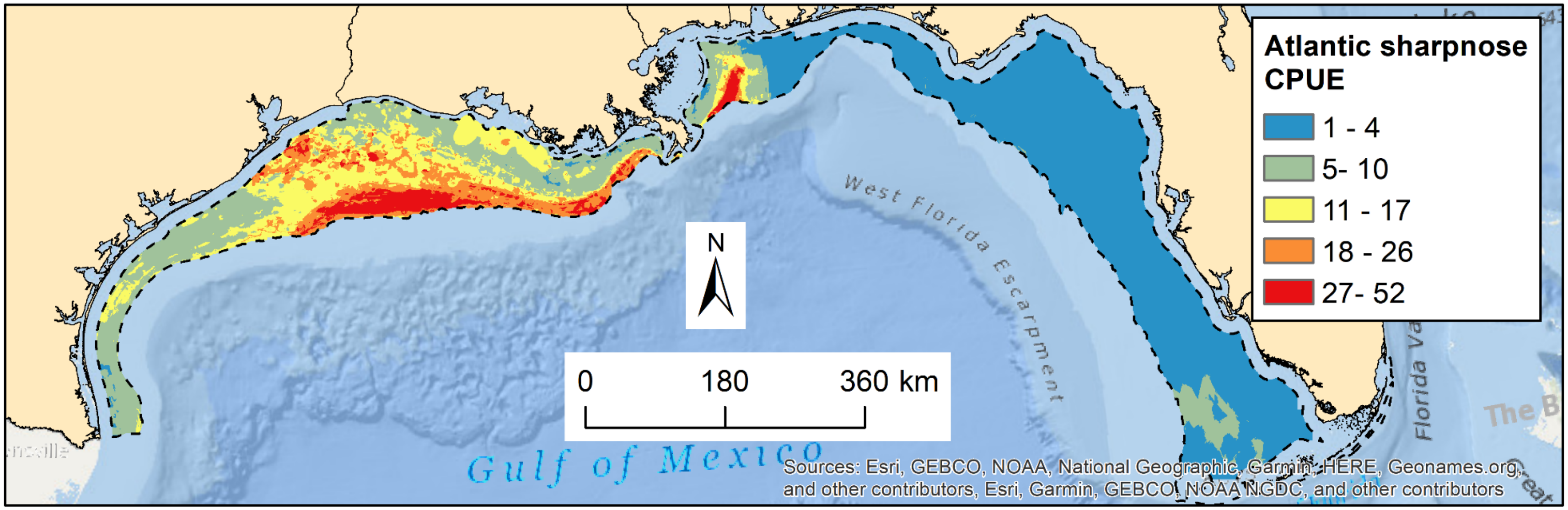

Figure 2.9. Predicted Atlantic sharpnose shark CPUE in the spring–fall seasons.
The study area is indicated by the dashed line, and CPUE represents the predicted number of sharks hooks^{-100} hr^{-1} of bottom longlne survey.

2.4 Discussion

By testing a comprehensive set of predictors, we discovered new species-habitat relationships with MLD, the area of nearby wetlands and estuaries, prey species, and substrate characteristics. Across all species considered, oceanographic variables were the most commonly selected predictors, and they often scored high in variable importance. Although distance to artificial or natural reefs were important predictors in the red snapper age-1 model, the other geomorphology predictors had low variable importance. Geomorphology predictors only appeared to refine fish distributions where oceanographic characteristics were already suitable for fish. For example, red snapper age-0 had a higher CPUE in close proximity to shoals and where the BPI predominately showed a hill topography. Yet, their spatial distribution was highly skewed towards the western GoM, where MLD was deeper and brown shrimp prey were most common. This inference of variable influence across spatial scales is supported by Mannocci et al. (2017), who shows that oceanographic features operate at intermediate scales of 1–100 km and factors such as prey operate at a scale of 10 m–1 km. We further suggest that geomorphology also affects species at this local scale. We acknowledge that prey species may be also be correlated with geomorphology, oceanography, and nearby ecosystems characteristics. SDMs are correlative models, but the species-habitat relationships we quantified here provide for refined distributions of species, reveal patterns of important habitat factors, and provide a basis for future hypothesis testing concerning EFH. As with the shrimp results (above), no species analyzed here was associated with benthic characteristics of bottom current velocity, slope, or depth heterogeneity.

We discovered MLD was an influential predictor of red snapper age-0 and age-1, lane snapper age-0, blacktip shark, and Atlantic sharpnose shark. MLD was also a predictor of brown shrimp CPUE (**Section 1.3.**), which were related to red snapper age-0 and Atlantic sharpnose shark. Measures of MLD have rarely been included as predictor of marine fish SDMs (Volume 1). In the nGoM, MLD appears to be influenced by the Loop Current, eddies, and wind stress. The Loop Current enters the Gulf of Mexico through the Yucatan Channel, goes northward in the basin, and then exits via the Straits of Florida. Importantly, the Loop Current produces large spin-off eddies that often take a westward path originating near the Mississippi Delta (Johnson et al. 2017). To a lesser extent, wind influences water vertical structure and 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), and upwelling or downwelling can contribute towards biological productivity (Spies et al. 2016). Therefore, the relationships with MLD and shrimp, snapper, and sharks may be the result of enhanced productivity in these waters.

We found prey species and nearby ecosystems were common predictors of species. In our review of spatial distribution models, prey were only considered in 4% of studies (Volume 1). Of these, none were studies of sharks or reef fish. Because SDMs are correlative models, we do not have direct evidence of drivers of species' distributions. However, the statistical models suggest that prey species can be better predictors than other environmental attributes. There are two probable reasons for these results: 1) fish-prey relationships represent a causal relationship, or 2) prey species represent a combination of environmental conditions, which are better correlated to species compared to single predictor variables. Future studies could further test predator-prey relationships to determine their applicability to EFH and ecosystem-based management. The productivity of nearby ecosystems, such as wetlands and estuaries, are usually ignored in marine fish research. Only a few studies have linked offshore marine fish or shrimp distribution to the proximity of mangroves (Barbier and Strand 1998) or estuaries (Beger and Possingham 2008). Nonetheless, interactions among coastal and marine ecosystems have been cited as important areas for conservation (Pickens et al. 2017), and trophic energy exchanges are thought to be important among these ecosystems (Zuercher and Galloway 2019).

As with pink shrimp (**Section 1.3**), the confusion matrices showed predictions of absences were consistently more accurate than predicted presences. Given that detectability is imperfect during surveys (MacKenzie et al. 2002), the SDMs presented here should be considered as relative rather than absolute numbers. Waters with a greater probability of occurrence or relative abundance (CPUE) should be interpreted in comparison to waters with lower probability of occurrence or relative abundance.

2.4.1 Red Snapper

Research on juvenile red snapper in the nGoM has focused on geomorphology, including the chronological selection of sand, shell, low-relief shell, and high-relief shell (Lingo and Szedlmayer 2006; Szedlmayer and Howe 1997; Wells et al. 2008b). Our study adds to these known associations with substrate complexity by quantifying topography at a landscape scale in the form of high relief (i.e., a positive BPI) and the proximity of sand shoals. Observational evidence has suggested a role between juvenile red snapper and sediment grain size (Powers et al. 2018; Szedlmayer and Mudrak 2014); however, we did not find evidence of such an association at the scale of our study area. Spatial patterns of red snapper age-0 were primarily determined by MLD and the predicted CPUE of brown shrimp, with minor contributions from other prey species and substrate complexity. A distinct ontogenetic shift occurred from age-0 to age-1 red snapper. Age-1 red snapper were still associated with spring MLD, but they moved to greater depths, farther offshore, and closer to natural and artificial structures. Our spatial models have similar patterns as Dance and Rooker's (2019) results, though we validated our model and took a more mechanistic approach, rather than testing variables like latitude and longitude.

Switzer et al. (2015) found the abundance of juvenile red snapper near Louisiana were reduced during years with severe hypoxia, and juveniles moved to deeper, cooler waters during those years. Although hypoxia was not selected as a variable for red snapper age-0 in our study, the model showed a low predicted CPUE in waters with a high frequency of hypoxia. The brown shrimp prey species is well known to avoid hypoxic waters (Craig 2012; Craig et al. 2005), and the correlation between brown shrimp and red snapper age-0 may have indirectly resulted in the low predicted CPUE in waters prone to hypoxia. Furthermore, the correlations between brown shrimp and predatory species provides evidence that hypoxia could affect fish farther up the food chain. We caution that trawl surveys were near shoals, but shoals themselves are usually too shallow to survey. Therefore, our habitat models do not account for fine-scale variation directly at the shoal. Dubois et al. (2009) does present evidence that Ship Shoal offshore of Louisiana is a hypoxia refuge for benthic invertebrates.

2.4.2 Lane Snapper

We found both age groups of lane snapper were strongly associated with oceanographic variables, particularly bottom temperature and salinity. They shifted from using salinities of ≤ 34 psu at age-0 to waters with > 34 psu at age-1. Given their subtropical / tropical range, the relationship with bottom temperature was not surprising. However, we expected more relationships with geomorphology. Juvenile lane snapper had been associated with sand shoals offshore of Louisiana (Brooks et al. 2005) and Texas (Mikulas and Rooker 2008; Wells et al. 2009). Our findings of juvenile lane snapper age-1 did show an association sandy sediments, but an association with shoals was lacking.

2.4.3 Blacktip, Spinner, and Atlantic Sharpnose Shark

Spinner and blacktip shark were not correlated with their main prey, menhaden, but bottom trawl surveys may not sample menhaden very well. Both shark species were associated with factors that indirectly describe menhaden habitat. Gulf menhaden utilize estuary and nearshore waters of moderate salinity, where they prey directly on phytoplankton and zooplankton (Olsen et al. 2014). These prey are likely correlated with chlorophyll-*a* measures. Gulf menhaden use estuaries and open water-marsh edges (Costanza et al. 1989; Rozas and Minello 2015; Rozas et al. 2007); therefore, a shark association with these habitats is expected. Drymon et al. (2013) found the distribution of blacktip shark was best explained by crustacean biomass, but noted these sharks eat small fish. Here, we infer that wetland productivity likely leads to greater blacktip shark CPUE. Spinner shark have previously been found in waters with relatively low dissolved oxygen (Drymon et al. 2013). We found a positive relationship between spinner shark and hypoxia with a peak probability of occurrence with a 25-40% chance of hypoxia. These results suggest that spinner shark may feed on prey that either aggregate at the edge of hypoxic zones (e.g., Craig 2012) or aggregate towards the surface (e.g., Hazen et al. 2009). In lab experiments, Atlantic menhaden (*B. tyrannus*) have been shown to avoid waters with low dissolved oxygen (Wannamaker and Rice 2000), but the specific behavior of menhaden due to hypoxia in the nGoM needs further study.

For Atlantic sharpnose shark, we found habitat relationships that differed from studies that found positive correlations with chlorophyll concentration (Drymon et al. 2013) and depths < 30 m (Drymon et al. 2010). Similar to red snapper, the association with brown shrimp and greater depths meant that low CPUEs of Atlantic sharpnose shark were predicted for frequently hypoxic waters. Harrington et al. (2016) found juvenile Atlantic sharpnose shark fed heavily on Penaeid shrimp offshore of Texas, and other studies in the nGoM have shown they feed on a wide range of species (Bethea et al. 2006; Drymon et al. 2012). The lack of a relationship with chlorophyll in our results may be because salinity, croaker, and nearby wetlands better explained Atlantic sharpnose shark distribution in nearshore waters where chlorophyll is high.

2.4.4 Conclusions and Implications for Dredging

In our wide-ranging study, we can conclude that not all shoals have equal value to fish. This is supported by the fact that Rutecki et al. (2014) reported lane snapper and red snapper in the GoM were either frequent, common, or rare, depending on the shoal. Oceanographic factors, prey species, and nearby wetlands and estuaries all play key roles in determining species' distributions. Evidence shows geomorphology only plays a minor or localized role in determining fish species' distributions. We caution that our analyses were based on fish surveys that typically span 3 km in length and microhabitat selection within this range may have been missed. Yet, our models showed a high predictive ability and were able to quantify suspected relationships between red snapper and substrate complexity. SDMs can illuminate habitat relationships as well as map the spatial patterns of species' distributions. We have demonstrated that modeling the distribution of species can be accomplished at a relatively fine scale. These spatial patterns can be applied to make management decisions and apply a strategic, regional perspective on natural resource use in the nGoM.

3 Predicting the Marine Distribution of Red Snapper, Black Seabass, and Shark Species in the South Atlantic

3.1 Introduction

The US Southeast Atlantic OCS (hereby South Atlantic) supports a diverse assemblage of fishes that provide high economic value to commercial and recreational fisheries. Fisheries economic impact in recent years has exceeded $4.4 billion annually and supports more than 77,000 jobs (National Marine Fisheries Service 2017). Geographically identified along the coastlines of the states of North Carolina, South Carolina, Georgia, and Florida, the region is bounded by prominent geographic features of Cape Hatteras and Cape Canaveral and oceanographically by the Gulf Stream current to the east. The region is home to species that use an array of seafloor and oceanographic habitats dominated by unconsolidated sediments forming sheets, ridges, and shoals, with interspersed emergent rocky reefs and other hard structures like artificial reefs and shipwrecks. Managed fish species are grouped into complexes based upon their general habitat preference (e.g., "snapper-grouper complex" that use reef and rocky habitats), migratory behaviors ("coastal pelagics" transient predators that seasonally migrate along the coastline), and migratory top predators (sharks).

Fishery-independent surveys have assessed managed species to derive indices of abundance that contribute to stock assessments for managing the populations. However, details on habitat selection and drivers of the broader distribution patterns remain poorly understood for the vast majority of species, including members of the snapper-grouper complex that are surveyed around rocky reefs throughout the region. For the snapper-grouper complex, there is evidence that some species are likely to be observed over soft sediments, though those sediment habitats may be adjacent to structured habitats (Bacheler et al. 2014). Studies of shark distributions are rare in the South Atlantic and mostly focus on seasonality, size and ages in coastal waters and estuaries (Thorpe et al. 2004; Ulrich et al. 2007).

Improvements in species distribution information and a greater understanding of relationships of fish to seafloor characteristics are needed to guide planning and permitting for ocean activities that may impact fish and their habitats. The demand for offshore marine sands is increasing in the United States (Drucker et al. 2004), and sand is commonly used for beach renourishment, barrier island restoration, and wetland restoration. Sand shoals are often preferred sand resources because of the quantity of sand per unit area, and the dredging of OCS sand shoals is likely to increase in the future as demand increases due to renourishment cycles for beaches, emergency repairs of beaches after storms, and the projected effects of sea-level rise (Nairn et al. 2004). BOEM is part of the US Department of the Interior and is responsible for the management and development of mineral resources on the OCS, including sediment resources. As demand for OCS sand 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).

For this paper, we focus on two primary objectives:

1) Test for habitat relationships of red snapper, black seabass, and selected shark species with a broad suite of environmental factors, including geomorphology, oceanographic characteristics, and nearby ecosystems

2) Model and predict the spatial distribution of five species of hard bottom fish and sharks to identify overlap with sand shoals that may be targeted for dredging activities

3.2 Methods

3.2.1 Study Area

The study area depicted the South Atlantic region of the United States spanning from Cape Hatteras, North Carolina through the South Atlantic Bight. The landward boundary of the study area was defined by OCSLA, which distinguishes Federal- and state-managed waters. The oceanic boundary of the study area was defined by a 50-m contour line from NOAA's CRM (NOAA National Centers for Environmental Information). This particular area was the focus because sand dredging operations authorized by the BOEM for beach and barrier island restoration does not exceed 50 m. The study area had a total surface area of 84,924 km^2. The benthic substrate of the area consists of fine to coarse sand sediments with patchy areas of hard bottom reefs. The most prominent feature of the South Atlantic is the Gulf Stream current, which creates a cross-shelf mixing of waters and strong water stratification (Castelao 2011). Surface and bottom water intrusions from the Gulf Stream 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).

3.2.2 Biological Data

Fish data were derived from fishery-independent surveys of SEAMAP, the Southeast Reef Fish Survey (SERFS) and the NOAA Southeast Fisheries Science Center, Mississippi Laboratories (**Table 3-1**). From these data, we only used data from 2004 and later because we wanted to depict current conditions as best as possible, and this timeframe still provided us with an adequate sample size. The vast majority of surveys were conducted from spring through fall, and, therefore, our SDMs do not represent snapper or shark distributions in the winter.

Surveys from the MSLABS-BL program used the methodology outlined in detail by Driggers III et al. (2012), with details briefly outlined here. All bottom longline surveys used a 15/0 circle hook baited with Atlantic mackerel. Surveys were randomly placed throughout the study area. Gear soak times were targeted to be 1 hr, as defined by the time elapsed between completion of deployment and initiation of retrieval. We used the centroid of bottom longline surveys to depict fish survey locations. For MSLABS-BL data, we removed extreme survey lengths. More specifically, we removed one survey that was 25 km in length and one that was 0.08 km. The remaining surveys ranged from 0.50–3.7 km. Additional bottom longline data were obtained from the SEAMAP program; however, the survey methodology differed, so these data were only used as presence data for validation purposes. Of the SEAMAP data, we only used surveys offshore of Florida, Georgia, and South Carolina for analysis because waters in our study area were not sampled offshore of North Carolina. The SEAMAP bottom longline methods offshore of Florida and Georgia included using either a 15/0 or a 12/0 circle hook baited with squid. For South Carolina, SEAMAP used a 15/0 circle hook and had a soak time target of 30 min, but they used Atlantic mackerel and striped mullet as bait.

The SERFS program surveys hard bottom locations in the South Atlantic with Chevron traps, and, since 2010, video attached to Chevron traps (Bacheler et al. 2014). The determination of sampling locations and details of both methods are provided by Bacheler et al. (2014), with a brief outline given here. Chevron traps were baited with 24 menhaden, and traps were deployed in groups of six, with each > 200 m from each other. The soak time for traps was targeted for 90 min. Since 2010, video cameras have been mounted over the mouth of traps with a view of approximately 60º from the trap. Video frames were read to identify fish at 30 second intervals over 20 min, which resulted in 41 frames read per sampling effort. For SERFS video surveys, we removed any video surveys that had < 41 frames read. For SERFS trap surveys, we removed traps with a duration of < 10 min and those > 150 min, because they had a much lower catch rate in preliminary models.

Table 3-1. Sources of fisheries-independent survey data. Sample sizes are provided for waters within our study area.

Data source	Acronym for dataset	Dates	Gear used for survey (*n* = # of surveys)	Species modeled
Southeast Fishery-Independent Survey	SEFIS video	2010–2017; 21 Apr–27 Oct	Video; *n* = 6,264	Red snapper Validation only for sandbar shark, tiger shark
Southeast Area Monitoring and Assessment: reef fish traps	SEAMAP traps	2010–2017; 23 Apr–26 Oct	Chevron traps; *n* = 5,210	Black sea bass
Southeast Fisheries Science Center, NMFS, Mississippi Laboratory	MSLABS-BL	2004–2017; 21 Apr–25 Sep	Bottom longline; *n* = 386	Blacknose shark, sandbar shark, tiger shark
Southeast Area Monitoring and Assessment: bottom longline	SEAMAP longline	2006–2016; 22 Apr–29 Dec	Bottom longline	Validation only for blacknose shark, tiger shark

3.2.3 GIS and Environmental Data Sources

All GIS analysis procedures and data collection methods were followed as described in **Section 1.2.3**, with the following modifications and additions. We developed predictor variables to depict oceanographic conditions, geomorphology, nearby ecosystems, and geography (**Table 3-2**). We used a bathymetry dataset derived as part of the South Atlantic Marine Bight Assessment (Conley et al. 2017). Despite these data being the best available for the South Atlantic, a few notable errors persisted. We updated a 1,600 km^2 and a 400 km^2 region offshore of northeastern North Carolina with sounding points acquired in 2016 (NOAA National Centers for Environmental Information 2016). To be consistent, we used the same kriging methods as the South Atlantic Marine Bight Assessment (Conley et al. 2017). To describe aspect, we calculated the aspect cosine and sine. We calculated depth, CV depth, slope, sediment grain size, proportion of area with shoal, slope, aspect, and the BPI within a 3-km radius using ArcGIS focal statistics.

Hard bottom habitats were depicted by three datasets that were combined. A predictive model (developed by Matthew Poti, NOAA Biogeography Branch) depicting probability of hard bottom occurrence was converted into a presence/absence model by representing areas with ≥ 36.3% probability of occurrence as hard bottom presence locations. This threshold was derived as the maximum AUC value, which is the threshold that corresponds with the model's optimal discrimination ability. We then added polygon data from South Atlantic Marine Bight Assessment mapping of hard bottom features. With this dataset, we removed the classifications of "possible" and "potential" hard bottom or hard bottom slope; these low confidence areas depicted seabed features > 2 km from known hard bottom locations and had substantial uncertainty. All "probable" hard bottom locations (≤ 2 km from known hard bottom), "high confidence" (≤ 1 km from known hard bottom), and very confident (mapped) locations were added to the spatial model. The probable and high confidence locations often depicted polygons where hard bottoms were known to be present along the periphery. The third dataset was the original synthesis of hard bottom point locations that were used as input into NOAA's predictive model. We removed point locations already classified as hard bottom (from above) and those within 90 m of hard bottom. These adjacent points often surrounded areas depicted as hard bottom. These restrictions resulted in 687 additional points. Because these were often isolated points, we calculated a 1 km buffer around these locations to add to the hard bottom model.

Table 3-2. Environmental variables developed to predict fish species in the South Atlantic.
Oceanographic predictors were from aggregations of monthly means spanning 2003-2017.

Variable type	Variable	Resolution of data	Source data
Substrate	Mean depth (m)	90 m	CRM plus modifications
Substrate	CV of depth	90 m	CRM plus modifications
Substrate	Density of artificial reefs (reefs km^{-1})	90 m	BOEM, Marine Cadastre
Substrate	Distance to artificial reef (km)	90 m	
Substrate	Distance to shoal (km)	90 m	Pickens and Taylor, NOAA Biogeography Branch, identification of shoals
Substrate	Proportion of area with shoal	90 m	Pickens and Taylor, NOAA Biogeography Branch, identification of shoals
Substrate	Mean sediment grain size (mm)	90 m	The Nature Conservancy, South Atlantic Marine Bight Assessment, substrate data
Substrate	Proportion hills (with BPI ≥1)	90 m	CRM plus modifications
Substrate	Slope (°)	90m	CRM plus modifications
Substrate	Proportion hard bottom	90 m	Matt Poti, NOAA Biogeography Branch + The Nature Conservancy- SAMBA
Substrate	Distance from hard bottom (km)	90 m	Matt Poti, NOAA Biogeography Branch + The Nature Conservancy- SAMBA
Substrate	Aspect (cosine)	90 m	CRM plus modifications
Substrate	Aspect (sine)	90 m	CRM plus modifications
Oceanographic	Bottom temperature (℃)	9.8 km	HYCOM + NCODA
Oceanographic	Chlorophyll-*a* (mg m^{-3})	5.1 km	Aqua MODIS satellite, 8-day composites
Oceanographic	Surface current direction- U (eastward water velocity, m s^{-1})	8.9 km	HYCOM + NCODA
Oceanographic	Surface current direction- V (northward water velocity, m s^{-1})	8.9 km	HYCOM + NCODA
Geography	Distance to shoreline	90 m	-
Nearby ecosystems	Nearby wetlands (km^2)	90 m	NWI
Nearby ecosystems	Nearby estuaries (km^2)	90 m	NWI

***** HYCOM + NCODA = HYbrid Coordinate Ocean Model + Navy Coupled Ocean Data Assimilation; MODIS = Moderate Resolution Imaging Spectroradiometer. CRM = Coastal Relief Model; NWI = National Wetlands Inventory

Salinity data were initially explored, but were excluded because there was minimal variation in our study area. To define the Gulf Stream, and subsequently the distance to the Gulf Stream, we examined summer U- and V- current velocity (m s^{-1}) for the summer season. The mean current velocity was highest during this season. We visually examined the data in 0.5 m s^{-1} increments and classified waters averaging ≥ 0.5 m s^{-1} for either the U or V direction as part of the Gulf Stream. This classification created a unified Gulf Stream current in our study area. We used Euclidean distance to calculate distance to the Gulf Stream.

To determine if coastal wetlands contributed to the distribution of fish in the marine environment, we used the classification of "estuarine and marine wetland" to depict estuarine wetlands and "estuarine and marine deepwater" to depict estuaries from the NWI dataset (U. S. Fish and Wildlife Service 2018). First, we calculated the maximum distance to wetlands within our study area, which was approximately 130 km in the South Atlantic. Then, we summed the number of 90-m cells depicting wetlands, and estuaries, within a radius of 130 km of each cell. The metric was converted to wetland km^{-2} and estuaries km^{-2} within each 130-km cell. Distance to shoreline was calculated by back-transforming boundaries of the Submerged Lands Act (SLA). The boundaries of the SLA represent a distance of 3 nm from the shoreline of South Atlantic states. We used the buffer tool to re-create the approximate shoreline boundaries (3–9 nm from SLA), and then calculated the Euclidean distance from the shoreline to the entirety of the study area.

3.2.4 Statistical Analysis

We investigated predictor variables for multicollinearity and removed highly correlated variables ($r > 0.80$) prior to analyses. Of note, we removed summer bottom temperature because of a strong correlation with depth ($r = -0.83$). In regard to temporal predictors, we used day-of-year and start time as variables in all models. The time at which surveys are conducted can affect the detectability of species and has previously been documented affecting shark catch (Driggers III et al. 2012). We did not use year as an explanatory factor because our primary objective was to determine long-term use of waters and substrates of the South Atlantic. Therefore, we assume years of high or low abundance are representative of long-term fish distribution.

Because improving our understanding of species-habitat relationships was one of our objectives, the variables in species models differed by a few variables. We excluded variables with no hypothesized relationship to species to help minimize issues with moderately correlated variables. We used chlorophyll as predictor of species except for red snapper. Red snapper are usually farther offshore than a relationship with chlorophyll would suggest. In the Gulf of Mexico (section 2), red snapper were found to be related to shrimp distribution, so we did include nearby wetlands in the South Atlantic models of red snapper. In models of demersal species (red snapper, black sea bass, blacknose shark), we did not test SST. Given the nGoM results (**Sections 1 and 2**), we did not test bottom current velocity. Instead, we tested surface current velocities because of the observed association between surface currents and MLD in the Gulf of Mexico. Data on MLD was not available for the South Atlantic.

For reasons described in **Section 1.2.4**, we aggregated training (70%) and validation (30%) data with alternating bands along a latitudinal gradient (Error! Reference source not found.). The use of a latitudinal gradient maintained a depth gradient in each block. More specifically, we reclassified latitude into 70 equal interval divisions, which resulted in 15-km bands across the study area. We then alternated the delineation of training (two bands) and validation (one band) datasets to achieve the desired ratio of training and validation data. The BRT analysis procedures outlined in **Section 1.2.4** were used to quantify species-habitat relationships, the relative importance of predictors, and interaction effects for snappers and sharks. We also assessed the accuracy of models using the same procedures as described from presence/absence modeling. For blacknose shark, the threshold obtained from the maximum Kappa statistic provided an extremely small sample size of predicted presences. Therefore, we used the presence/absence threshold obtained from minimum distance on the receiver operating characteristic plot. We assumed the effect of time of survey represented a detectability effect of tiger shark. Therefore, we applied the model at their peak time of detectability, which was 23:00.

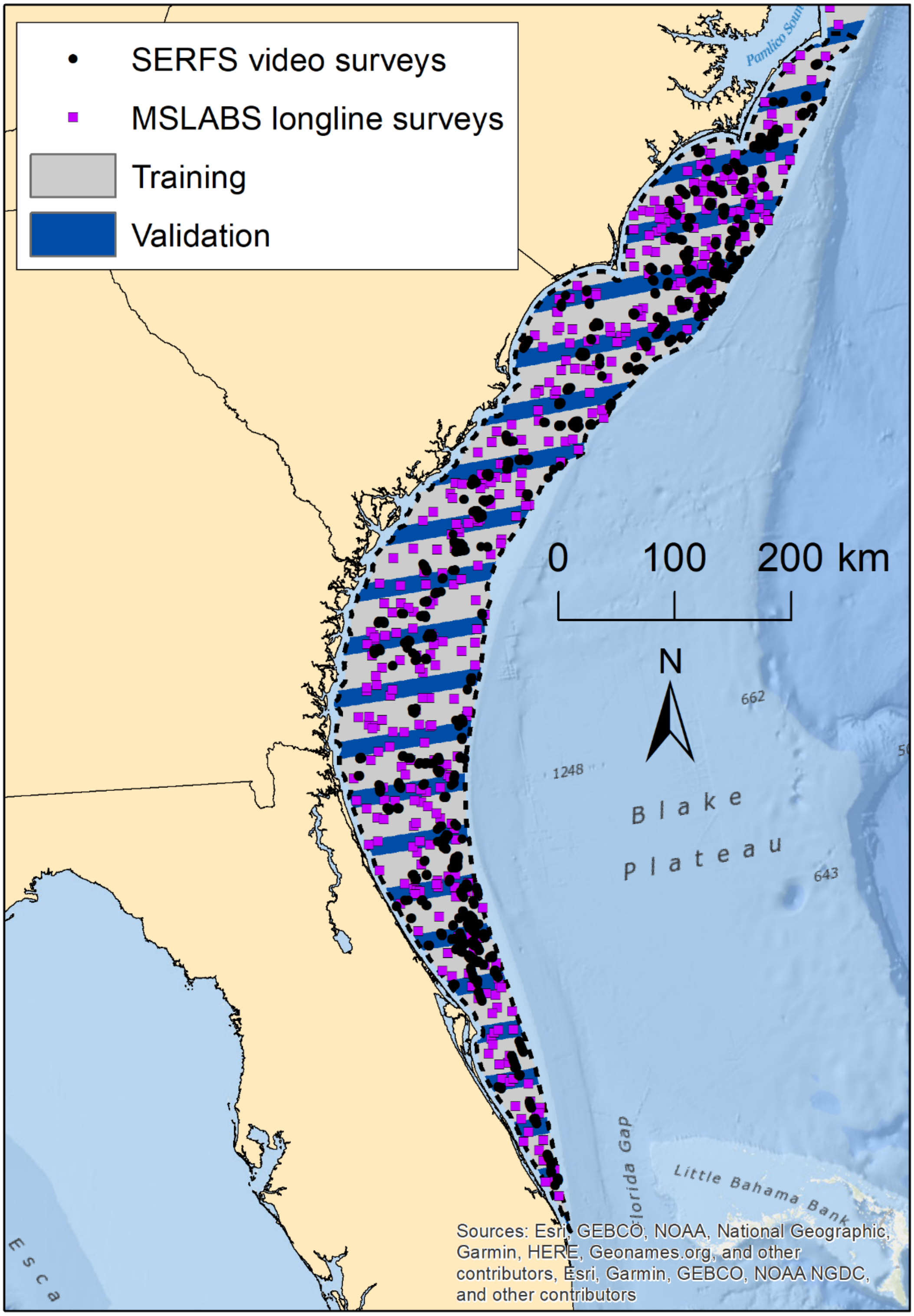

Figure 3.1. The study area with training and validation zones depicted with surveys overlaid. SERFS trap surveys were often at video survey locations.

3.3 Results

The validation results showed AUC statistics ranged from 0.73–0.89, which indicated a good or very good ability of models to discriminate present and absent locations of species (**Table 3-3**). The confusion matrices (**Table 3-4** to **Table 3-8**) showed absences were better predicated than presences for blacknose shark and red snapper. Tiger and sandbar shark models had similar abilities to discriminate presence and absence. For the most common species, black sea bass, presence was predicted better than absences. No geomorphology variables were selected by species, and variable importance measures showed a mixed of oceanographic condition, nearby estuaries and wetlands, and depth all contributed to the SDMs (**Figure 3.2**).

3.3.1 Red Snapper

Red snapper (**Figure 3.3**) were present on 24% of reef video surveys. Red snapper were positively related the amount of nearby wetlands and had the highest probability of occurrence with a depth range of 25–38 m. Red snapper were more likely to occur within approximately 45 km of the Gulf Stream and with westward surface currents. Interaction effects were particularly strong. Red snapper were more likely to occur with a combination of a high amount of nearby wetlands and a greater depth (IE = 212). A combination of westward currents and a depth range of approximately 25–35 m also had a strong positive effect on red snapper occurrence (IE = 82).

3.3.2 Black Sea Bass

Black sea bass (**Figure 3.4**) were present on 66% of reef trap surveys. Black sea bass had a strong, positive relationship with chlorophyll and primarily a positive relationship with nearby estuaries. However, they had a negative relationship with very high amounts of nearby estuaries (> 7,000 km^2), which only occurred offshore of North Carolina. Black sea bass had a particularly low probability of occurrence > 75 km from the shoreline (**Figure 3.4**). They had a greater likelihood of occurrence either near the Gulf Stream (< 25 km) or much farther away from the Gulf Stream (> 85 km). Strong interactions occurred among variables. Distance to shoreline had a minimal effect where nearby estuaries were > 7,000 km^2 (near North Carolina) (IE = 126). For waters with > 7,000 km^2 of nearby estuaries, the Gulf Stream had a greater effect on black sea bass probability of occurrence (IE = 116).

3.3.3 Blacknose Shark

Blacknose shark (**Figure 3.5**) were present on 10% of MSLABS-BL bottom longline surveys. An additional 56 blacknose shark presence locations from SEAMAP bottom longline surveys were used for validation. The blacknose shark had positive relationships with chlorophyll and the amount of nearby estuaries. They had a higher probability of occurrence with a higher velocity of U-direction surface currents, particularly westward currents; however, eastward currents also had a positive effect. Interactions were not present.

3.3.4 Sandbar Shark

Sandbar shark (**Figure 3.6**) were present on 25% of MSLABS-BL surveys. An additional 50 sandbar shark presence locations from SERFS video were used for validation. For sandbar shark, probability of presence was highest with water depths of 42–50 m and probability declined steadily at more shallow depths. Sandbar shark were related to higher fall bottom temperatures with a sharp increase in their predicted occurrence with temperatures > 24.5°C. Interactions were not present.

3.3.5 Tiger Shark

Tiger shark (**Figure 3.7**) were present on 42% of MSLABS-BL surveys. An additional 37 tiger shark presence locations from SERFS video surveys and 16 SEAMAP bottom longline survey presence locations were used for validation. Tiger shark presence was associated with a greater amount of nearby wetlands and with greater water depths. Peak tiger shark occurrence was predicted with depths of 25–50 m. Survey time also affected the presence of tiger sharks in surveys, as 11:00–23:00 was predicted to have the highest probability of occurrence. There was an interaction with nearby wetlands and survey time (IE = 10). This interaction showed that depth did not have as much of an effect during low probability survey times.

Table 3-3. BRT specifications and AUC statistics for marine species distribution models of snappers and sharks.

Species	Tree complexity	Learning rate	# of trees	Cross-validation AUC	Validation AUC
Red snapper	5	0.02	1,850	0.86	0.79
Black sea bass	4	0.03	1,150	0.86	0.81
Blacknose shark	5	0.001	4,300	0.85	0.88
Sandbar shark	1	0.03	4,950	0.83	0.73
Tiger shark	2	0.002	1,700	0.73	0.73

Table 3-4. Confusion matrix from the validation data of the red snapper model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.41.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	1,501	331	82%
Predicted presence	135	168	55%

Table 3-5. Confusion matrix from the validation data of the black sea bass model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.58.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	274	153	64%
Predicted presence	239	1,157	83%

Table 3-6. Confusion matrix from the validation data of the blacknose shark model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.15.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	101	7	94%
Predicted presence	24	61	72%

Table 3-7. Confusion matrix from the validation data of the sandbar shark model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.33.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	73	34	68%
Predicted presence	21	59	74%

Table 3-8. Confusion matrix from the validation data of the tiger shark model at the optimal threshold to distinguish presence/absence, which was with a probability of 0.42.

	Observed absence	Observed presence	User's accuracy (% correct)
Predicted absence	52	25	68%
Predicted presence	34	79	70%

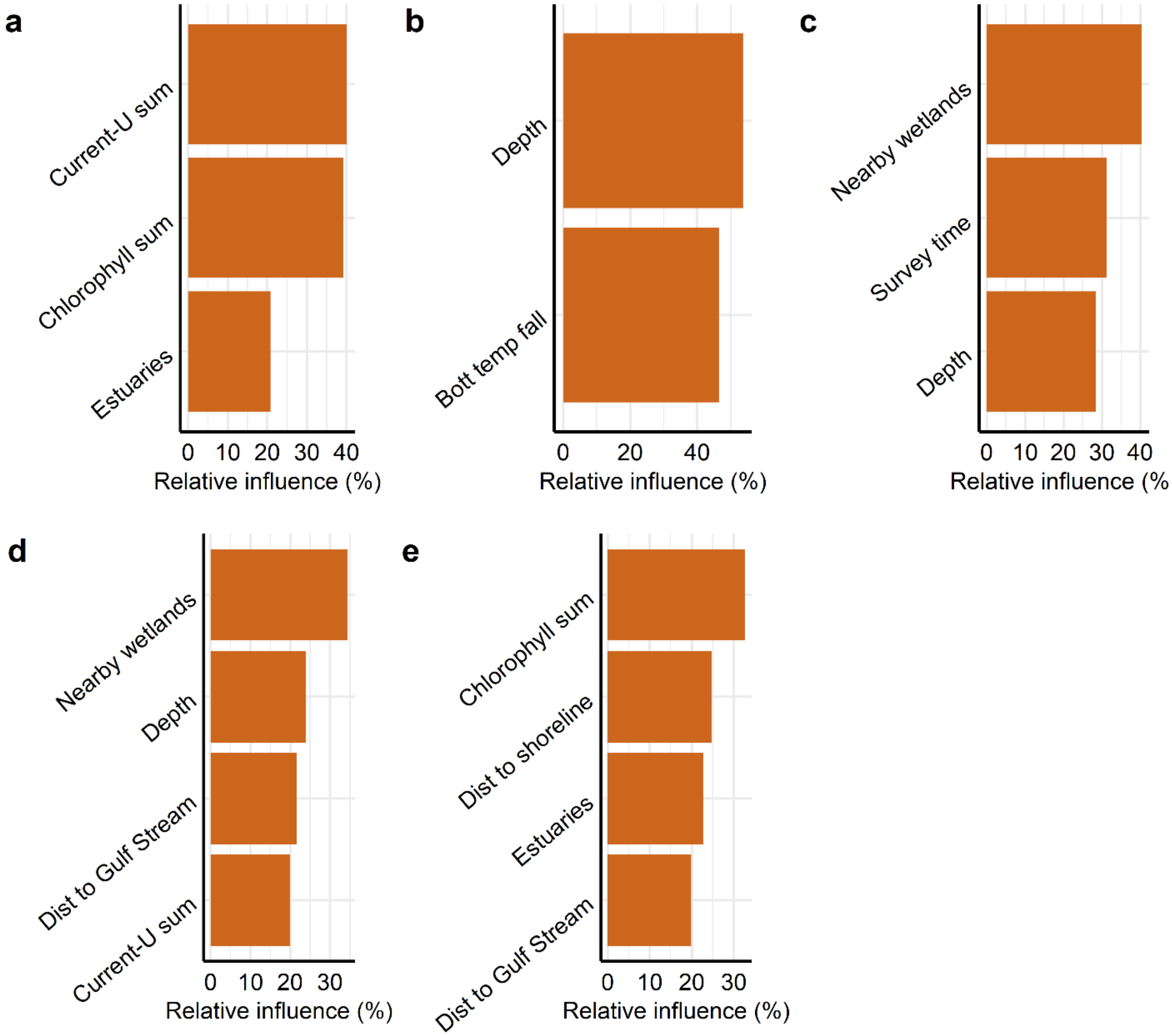

Figure 3.2. Relative importance of variables in models of a) blacknose shark, b) sandbar shark, c) tiger shark, d) red snapper, and e) black sea bass.

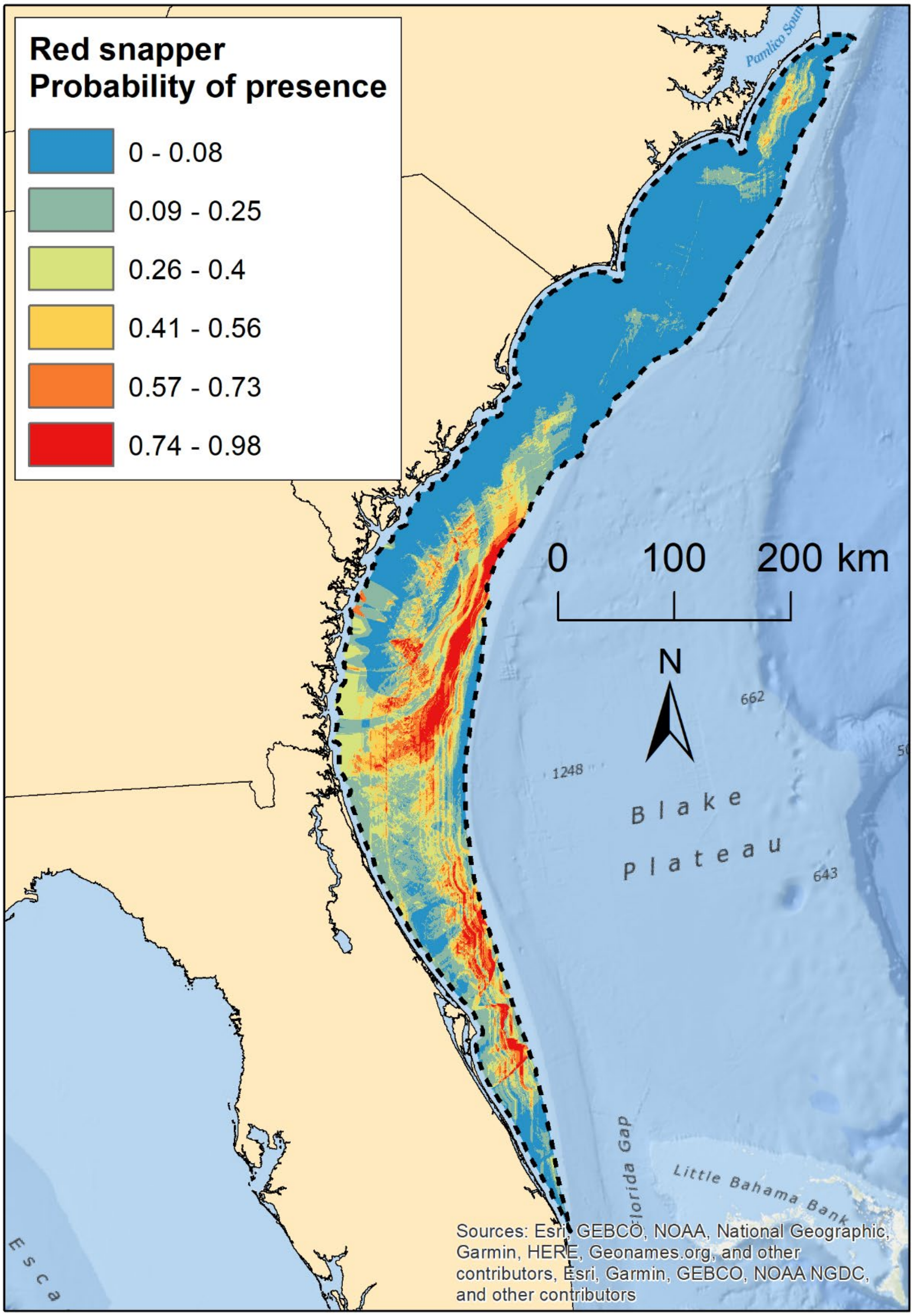

Figure 3.3. Predicted red snapper probability of presence in spring–fall seasons.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a video survey at a hard bottom location.

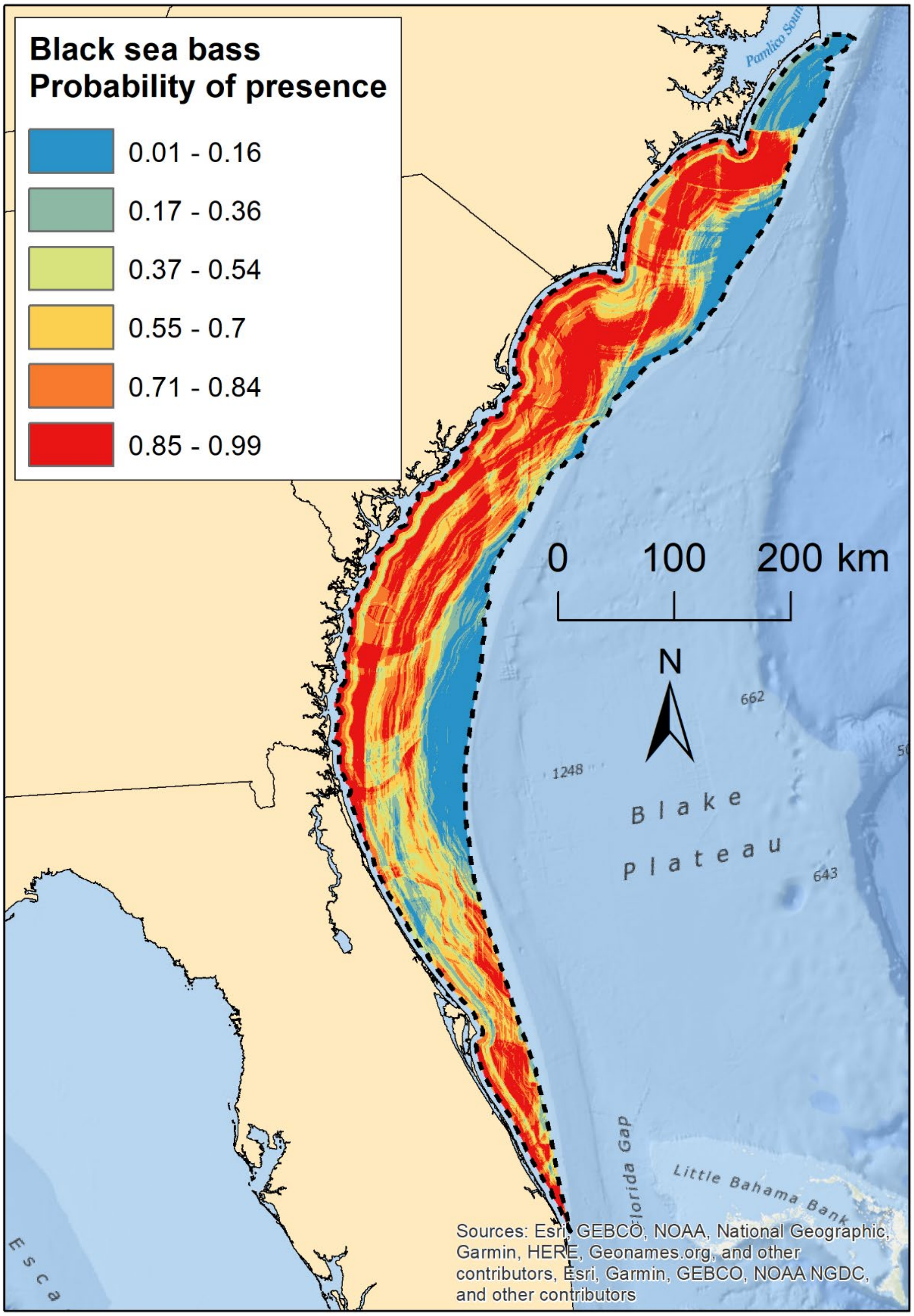

Figure 3.4. Predicted black sea bass probability of presence in spring–fall seasons.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a trap survey at a hard bottom location.

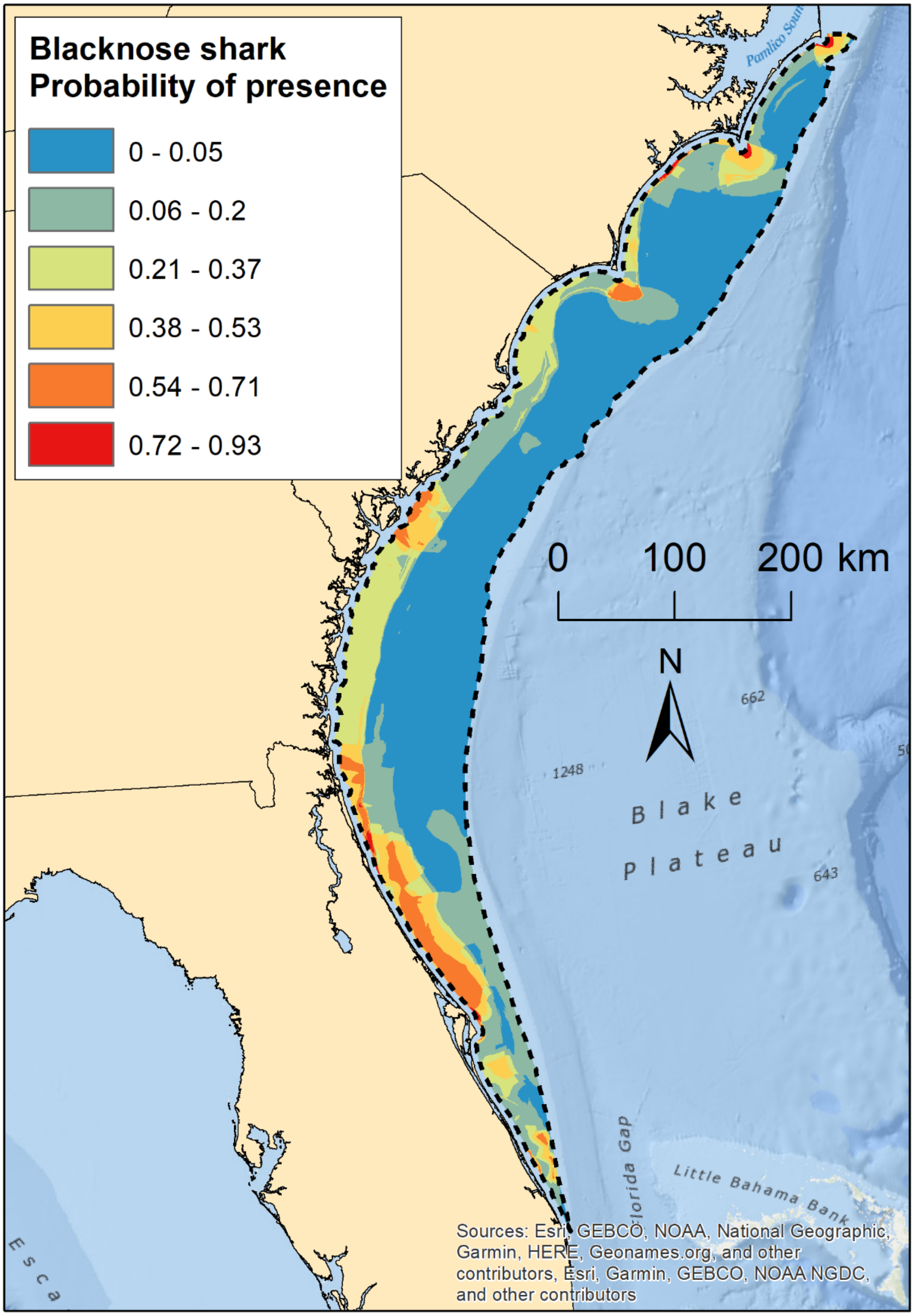

Figure 3.5. Predicted blacknose shark probability of presence in spring–fall seasons. The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a bottom longline survey.

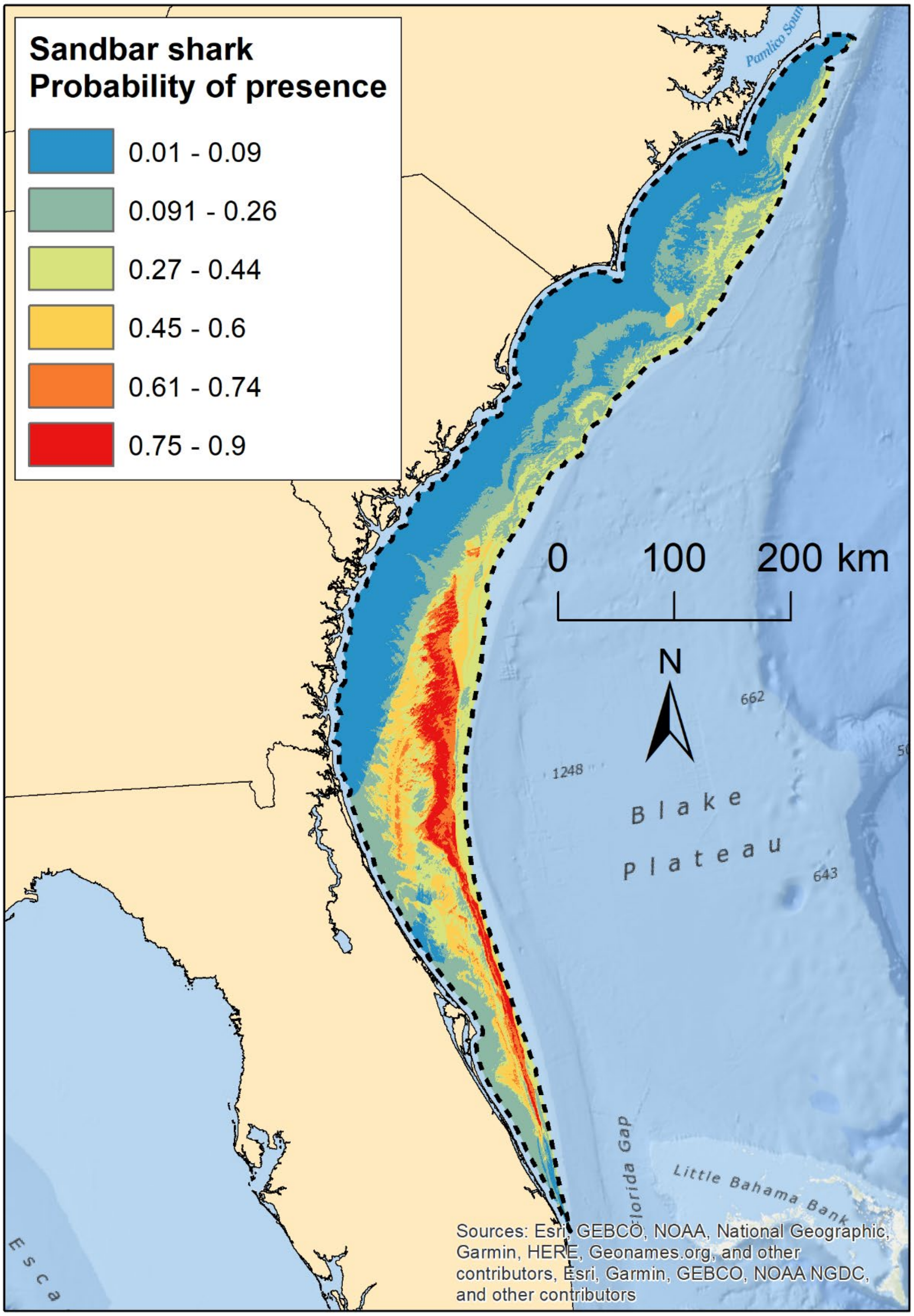

Figure 3.6. Predicted sandbar shark probability of presence in spring–fall seasons.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a bottom longline survey.

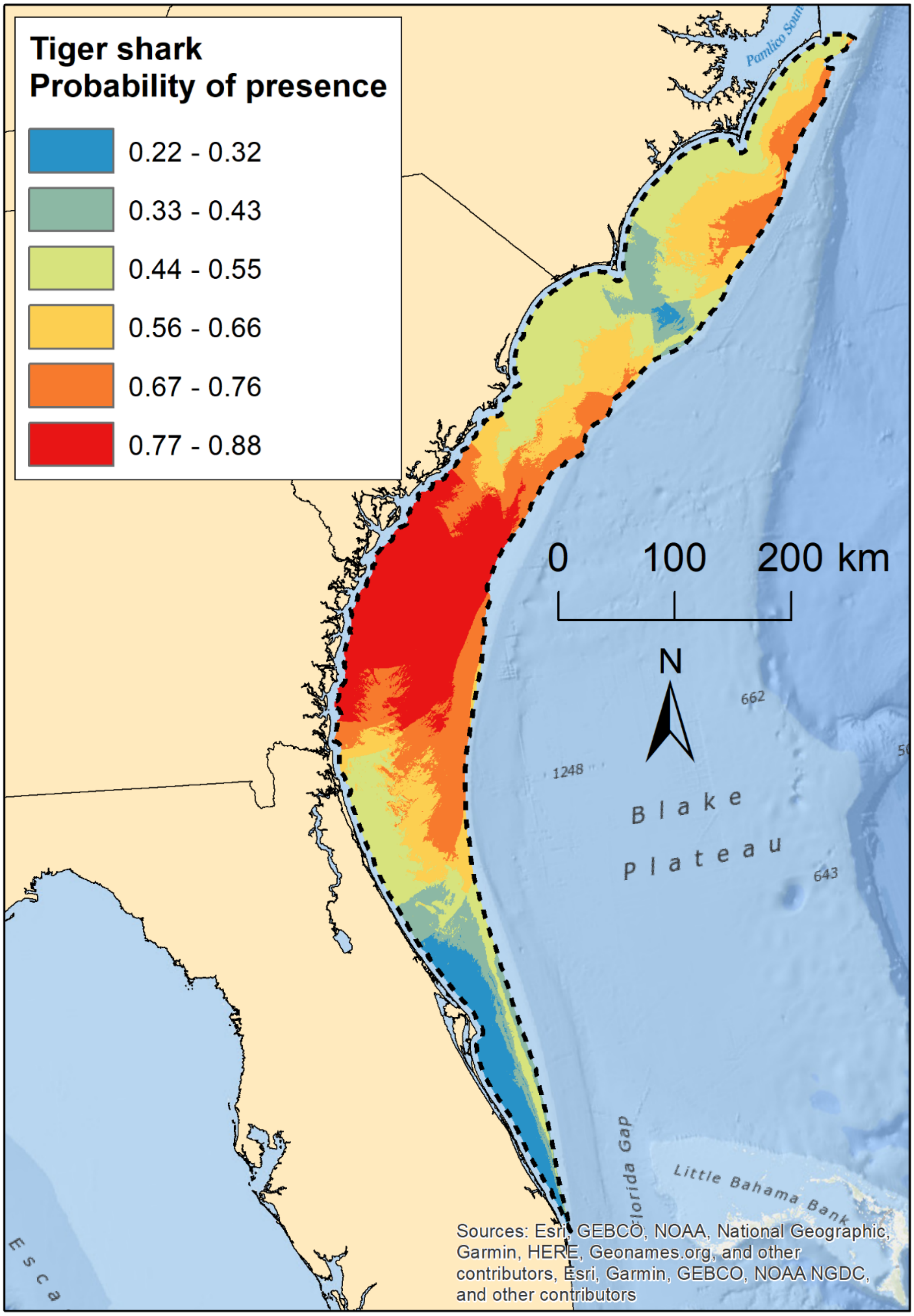

Figure 3.7. Predicted tiger shark probability of presence in spring–fall seasons.
The study area is indicated by the dashed line, and probability of presence represents the probability of capture given a bottom longline survey.

3.4 Discussion

Overall, we discovered new species-habitat relationships that incorporated the nearby ecosystems of wetlands and estuaries along with the most prominent oceanographic feature, the Gulf Stream. Concerning geomorphology, our results with the five South Atlantic fish species analyzed here are consistent with the results from analyses of nGoM shrimp, snapper, and sharks (**Sections 1 and 2**). Oceanographic conditions and nearby ecosystems are the dominant correlates with fish distribution, and the value of shoals as fish habitat is highly variable based on those factors (rather than geomorphology characteristics). In the South Atlantic, none of the five species were associated with the benthic characteristics of slope, aspect, BPI, depth heterogeneity, distance to shoal, or proportion of area as shoal.

3.4.1 Red Snapper and Black Sea Bass

We discovered the proximity to the Gulf Stream was an influential predictor of red snapper and black sea bass. The Gulf Stream is a prominent feature of the South Atlantic, and it creates a cross-shelf mixing of waters as well as strong water stratification (Castelao 2011). 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). These characteristics of Gulf Stream water intermixing with shelf waters is consistent with the red snapper habitat relationships we observed here. Red snapper presence was most likely within approximately 40 km of the Gulf Stream and with a relatively strong western surface current. In the nGoM, we observed a positive relationship with red snapper and a deeper MLD. In the Gulf of Mexico, surface currents and the Loop Current were correlated with deeper MLD (unpublished data). Data sources for MLD were lacking in the South Atlantic, but the surrogates of distance to Gulf Stream and surface current velocity demonstrate the importance of this process. Black sea bass had a greater probability of presence both within 20 km of the Gulf Stream and much farther from the Gulf Stream (> 80 km). The association with the latter probably characterizes their affinity towards high chlorophyll concentrations and being closer to the shoreline. Red snapper were associated with more nearby wetlands, which is consistent with their association with shrimp prey species in the nGoM, whereas the association black sea bass and estuaries may be a result of their use of estuaries as juveniles (Mercer and Moran 1989).

3.4.2 Blacknose, Sandbar, and Tiger Shark

Studies conducted in the shallow waters offshore of North Carolina (Thorpe et al. 2004) and South Carolina (Ulrich et al. 2007) found blacknose shark were common in waters of 3-15 m depths. Our results support these observations, however, we identified more refined habitat characteristics that contributed to their presence within shallow waters. In the nGoM, Drymon et al. (2013) quantified habitat associations of blacknose shark and found depth (or temperature as a correlate) supported a depth preference of 10–30 m. Here, a positive association with chlorophyll meant that blacknose shark were in a close proximity to the shoreline. The positive association with the amount of nearby estuaries and a relatively high velocity of U-direction surface currents resulted in a predicted distribution near many inlets and sounds in the South Atlantic. This finding builds on a previous study that observed blacknose shark near inlets in South Carolina waters (Ulrich et al. 2007). These inlets may be particularly important habitats for blacknose shark.

Our model of sandbar shark showed they primarily used waters ≥ 35 m in depth. In support of this result, observations of sandbar shark in the Gulf of Mexico showed they are rarely encountered at shallow depths but have been recorded in depths of > 30 m (Drymon et al. 2010). We emphasize that we did not analyze data from nearshore, state-managed waters where juveniles may be found (e.g., Ulrich et al. 2007). Seasonal differences in sandbar shark depth use also appear to exist. Conrath and Musick (2008) reported that 80% of summer sandbar shark locations were in depths of < 12 m (range of 0–24 m) in the water column, but winter observations ranged 0–172 m. Total depth of the water column itself was not reported in this study. In our study, depth was negatively related to bottom temperature. Therefore, the selection for depths up to 50 m is also indicative of sharks using relatively cooler water temperatures in the spring and summer. Additionally, we found an association between sandbar shark and fall bottom temperature, which were the lowest temperatures of the year. Sandbar shark are known to leave waters that get too cold. For example, 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 this is because of North Carolina's proximity to the warm Gulf Stream.

For tiger shark, we found a greater probability of presence with higher amounts of nearby wetlands. Worldwide, tiger sharks are known to use nearshore habitats (e.g., seagrasses) and to prefer shallow waters; presumably, these preferences characterize waters with a high prey density (Heithaus et al. 2006). Similarly, coastal wetlands of the South Atlantic likely are productive waters for tiger shark prey species.

3.4.3 Conclusions and Implications for Dredging

In our wide-ranging study, we can conclude that not all shoals have equal value to fish in the US Southeast Atlantic. Oceanographic factors, such as distance to Gulf Stream, and surface currents play a substantial role, as well as the amount of nearby wetlands and estuaries. Evidence shows geomorphology only plays a minor role in the distribution of hard bottom fish and sharks. We caution that our analyses were based on fish surveys that typically span 3 km in length, and microhabitat selection within this range may have been missed. Yet, our models showed a high predictive ability. For example, three of five of our models had an AUC of ≥ 0.79. We have demonstrated that modeling the distribution of species can be accomplished at a relatively fine scale. These predicted spatial patterns can be used in decision support tools to identify potentially sensitive habitats, and relative hotspots for species, and allow for a strategic, regional perspective on natural resource use in the South Atlantic.

4 Predicting the Marine Distribution of Demersal Species in the Greater Atlantic

4.1 Introduction

SDMs are state-of-the-art statistical models that predict the distribution of species based on species-habitat relationships (Guisan and Zimmermann 2000; Robinson et al. 2011). Extending survey samples across the complex habitat mosaics allow for determination of relative hotspots for species and can identify particularly sensitive areas to avoid when planning offshore uses that potentially impact benthic habitats. The Northeast Fisheries Science Center (NEFSC) has recently developed SDMs for the Greater Atlantic (personal communication and unpublished data, Kevin Friedland, NEFSC). They had the objective of modeling the spatial distribution of North Atlantic species for the spring and fall seasons. Their work also examined the broad importance of predictor variables categorized as the following: physical (e.g., depth, temperature), primary production, secondary production (i.e., zooplankton), and benthic habitat complexity (i.e., substrate measures). The NEFSC provided us with a subset of species models derived from the last decade of fishery-independent trawl data, while their broader work covers over 30 years of fishery and oceanographic survey data.

The multispecies spatial modeling effort led by Kevin Friedland (NEFSC) is in the preparation phase for a peer-reviewed manuscript; therefore, we only provide a brief overview here. Currently, there is a manuscript concerning American lobster that is in review, and this manuscript will describe more detailed methodology for the species described here and included in the ShoalMATE tool (see Volume 4). Here, we provide a brief overview of the modeling data sources, methodology, and results.

4.2 Methods

The NEFSC used the following components to build SDMs for the Greater Atlantic:

- Fishery-independent trawl survey data from 2009–2018 were used for statistical modeling. The NEFSC has a longstanding trawl survey program, and the latest decade of data were used for modeling. All species with an adequate sample size from trawl surveys were modeled.

- Predictor variables sampled during trawl surveys included remote sensing data on oceanographic conditions, substrate descriptors, zooplankton abundance, and *in situ* measures of depth, water temperature, and salinity. Oceanographic predictor variables, such as chlorophyll concentration, were summarized on a monthly basis, and each month was tested as a predictor.

- The NEFSC used Random Forest (Cutler et al. 2007) for statistical models. Similar to BRT (**Sections 1, 2, and 3**), Random Forest is a machine-learning method that uses an ensemble of regression trees to make predictions. Random Forest has been demonstrated as an effective predictive modeling method (Rooper et al. 2017). Model variables were reduced using the model selection criteria of Murphy et al. (2010). Individual models often had > 20 predictor variables. Further details on the modeling methodology will be provided by Friedland et al. (2020).

- The predictive models quantify the probability of occurrence for each species by season (spring and fall), and the accuracy of models were tested with a cross-validation procedure. With this method, species were assessed with a receiver operating characteristic, AUC. The AUC has been commonly used to test the predictive ability of SDMs (Guisan and Zimmermann 2000), and the analysis is independent of thresholds. Measures of the AUC range from 0.0 to 1.0 and were interpreted as suggested by Manel et al. (2001) and Swets (1988) as follows: < 0.50 = no discriminatory power; 0.50–0.69 = poor power; 0.70–0.89 = good power; and 0.90–1.0 = excellent discriminatory power.

4.3 Summary of Model Outputs

Thirty-four federally managed species were modeled for the spring and fall seasons, and these 68 models all had > 70% accuracy based on the AUC statistics for each season (**Table 4-1**). Overall, the models showed inshore-offshore movements of species were common, so distributions were often strikingly different between seasons. Importantly, these models do not depict summer and winter seasonal distributions of species.

A few inferences can be drawn from all the species modeled by the NEFSC (EFH species and other species) with the longer timeframe of modeling results. Among the categories of predictors, benthic substrate variables were selected least often, and measures of variable importance showed these substrate variables had less importance in models compared to other categories of predictor variables. Secondary production, primary production, and physical variables had varying importance depending on season and type of species (e.g., benthivore vs. planktivore).

Table 4-1. Federally managed species and the accuracy of models provided by the NEFSC.

Common name	Fall AUC	Spring AUC	Common name	Fall AUC	Spring AUC
Acadian redfish	0.94	0.92	Scup	0.91	0.9
American plaice	0.94	0.91	Sea scallop	0.84	0.84
Atlantic cod	0.89	0.84	Silver hake	0.82	0.81
Atlantic herring	0.9	0.76	Smooth skate	0.88	0.89
Black sea bass	0.86	0.86	Spiny dogfish	0.8	0.79
Bluefish	0.86	0.88	Summer flounder	0.9	0.84
Butterfish	0.77	0.86	White hake	0.88	0.87
Clearnose skate	0.92	0.92	Windowpane	0.85	0.83
Goosefish	0.77	0.76	Winter flounder	0.87	0.88
Haddock	0.84	0.85	Winter skate	0.88	0.81
Little skate	0.86	0.83	Witch flounder	0.9	0.84
Longfin squid	0.85	0.89	Yellowtail flounder	0.88	0.87
Northern shortfin squid	0.81	0.88	Atlantic mackerel	0.77	0.77
Offshore hake	0.93	0.91	Barndoor skate	0.88	0.86
Red hake	0.82	0.8	Ocean pout	0.82	0.79
Rosette skate	0.93	0.93	Pollock	0.86	0.82
Scup	0.91	0.9	Atlantic angel shark	0.91	0.92

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Appendix A: Common and Scientific Names Cited in the Text

Common Name	Scientific Name	Common Name	Scientific Name
Acadian redfish	*Sebastes fasciatus*	Lizardfish	*Synodus spp.*
American plaice	*Hippoglossoides platessoides*	Longfin squid	*Loligo pealeii*
Atlantic cod	*Gadus morhua*	Mantis shrimp	*Squilla spp.*
Atlantic herring	*Clupea harengus*	Menhaden	*Brevoortia spp.*
Atlantic mackerel	*Scomber scombrus*	Northern shortfin squid	*Illex illecebrosus*
Black sea bass	*Centropristis striata*	Ocean pout	*Macrozoarces americanus*
Bluefish	*Pomatomus saltatrix*	Offshore hake	*Merluccius albidus*
Butterfish	*Peprilus triacanthus*	Pollock	*Pollachius virens*
Croaker	*Micropogonias undulatus*	Red drum	*Sciaenops ocellatus*
Flounder, summer	*Paralichthys dentatus*	Searobin	*Prionotus spp.*
Flounder, windowpane	*Scophthalmus aquosus*	Sea scallop	*Placopecten magellanicus*
Flounder, winter	*Pseudopleuronectes americanus*	Scup	*Stenotomus chrysops*
Flounder, witch	*Glyptocephalus cynoglossus*	Shark, Atlantic angel	*Squatina dumeril*
Flounder, yellowtail	*Limanda ferruginea*	Shark, Atlantic sharpnose	*Rhizoprionodon terraenovae*
Goosefish	*Lophius americanus*	Shark, blacknose	*Carcharhinus acronotus*
Haddock	*Melanogrammus aeglefinus*	Shark, blacktip	*Carcharhinus limbatus*
Hake, red	*Urophycis chuss*	Shark, sandbar	*Carcharhinus plumbeus*
Hake, silver	*Merluccius bilinearis*	Shark, spinner	*Carcharhinus brevipinna*
Hake, white	*Urophycis tenuis*	Shark, spiny dogfish	*Squalus acanthias*
Shark, tiger	*Galeocerdo cuvier*	Skate, little	*Leucoraja erinacea*
Shrimp, brown	*Farfantepenaeus aztecus*	Skate, clearnose	*Raja eglanteria*
Shrimp, pink	*Farfantepenaeus duorarum*	Skate, smooth	*Malacoraja senta*
Shrimp, white	*Litopenaeus setiferus*	Snapper, lane	*Lutjanus synagris*
Skate, barndoor	*Dipturus laevis*	Snapper, red	*Lutjanus campechanus*

OCS Study
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Regional Essential Fish Habitat Geospatial Assessment and Framework for Offshore Sand Features

Volume 4: Development of ShoalMATE: Shoal Map Assessment Tool for Essential Fish Habitat

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Volume 4: Development of ShoalMATE: Shoal Map Assessment Tool for Essential Fish Habitat

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Bureau of Ocean Energy Management

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List of Abbreviations and Acronyms

AOI	area of interest
BMP	best management practices
BOEM	Bureau of Ocean Energy Management
DOI	US Department of the Interior
EFH	Essential Fish Habitat
ETL	Extract-Transform-Load
fGDB	file geodatabase
GIS	Geographic Information System
HAPC	Habitat Areas of Particular Concern
MGET	Marine Geospatial Ecology Tools
MMP	Minerals Management Program
NOAA	National Oceanic and Atmospheric Administration
OCS	Outer Continental Shelf
ShoalMATE	Shoal Map Assessment Tool for EFH
SST	sea surface temperature

Abstract

QSI was contracted to build a standardized reporting tool to facilitate better communication between BOEM and NOAA during Essential Fish Habitat (EFH) assessments required for dredging projects on the Outer Continental Shelf (OCS). QSI initiated development by gathering requirements from BOEM's Marine Minerals Program and NOAA's Habitat Conservation Division. We then designed the database architecture and workflow to meet the needs of access and usability for stakeholders with varying levels of familiarity with GIS. We ran the data necessary to support the tool (e.g., habitat descriptors, species models, project boundaries) through a series of custom scripts that store information describing each identified shoal in a database specifically designed for expedited queries within the front-end application. The front-end application presents this queried information within a web browser and generates a template report, as a Microsoft Word document, that can be edited by analysts to create a final, tangible product.

1 Background to the Tool Development Process

One of the primary goals of this project was to develop a standardized geographically and temporally based reporting tool for use by the Bureau of Ocean Energy Management's (BOEM's) Marine Minerals Program (MMP) practitioners in the Atlantic and Gulf of Mexico region to support Essential Fish Habitat (EFH) consultations for dredging. The ShoalMATE (Shoal Map and Assessment Tool for EFH) tool allows a user to share their assessment logic in a consistent manner. Having the information readily available to review will improve communications between agencies and provide more power and transparency in the EFH consultation process.

The results of the literature review completed as part of Volume 1 of this report, and additional data exploration associated with the tool development revealed numerous data sources that could help to characterize bottom habitats, particularly those of sandy shoals utilized for dredging operations. The development team identified a set of required information to be included in a template version of an assessment document in consultation with members of the MMP and the National Oceanic and Atmospheric Administration (NOAA) Habitat Conservation Division, as well as external subject matter experts, and through review of previous EFH Assessments for dredging projects, (**Table 1-1**).

Table 1-1. List of required components in an EFH Assessment document.

Requirement
A description of the proposed project area
Overview of the location
Bathymetry
Bottom current direction
Substrate type
Recovery potential/accretion of sand resource
Previous dredge events
A list of federally managed species with overlapping EFH polygons (from NOAA)
Evaluation of potential impacts on those species based on known habitat affinities or predicted distribution of fish and shrimp species
Proposed mitigations and best management practices
Results and conclusions
References

Data to support these requirements were compiled into an ESRI file geodatabase (fGDB) if hosted web services were not available. Data sources included MarineCadastre.gov, BOEM's Marine Minerals Information System, and personal communications with BOEM and NOAA stakeholders. Remotely sensed data (sea surface temperature (SST), chlorophyll-*a*, current velocity, etc.) were compiled as 10-year monthly averages using the Marine Geospatial Ecology Tools (MGET) developed by the Marine Geospatial Ecology Lab at Duke University (Roberts et al. 2010). We utilized NOAA's EFH polygons but created additional related tables to store information digitized from EFH source documentation compiled by regional Fishery Management Councils. The table also documents where Volume 1 identified additional sources of information on managed species exceeding the information in the official documentation. Once all available datasets were combined into a fGDB, the data was loaded into ESRI MXD files and published as web services for use in the application.

2 Application Development

BOEM was interested in creating a simple interactive mapping application for users with minimal to moderate Geographic Information System (GIS) skills and experience. This ruled out developing an add-in package to be used in conjunction with desktop mapping software such as ESRI ArcMap, as access to software licenses would be limiting. We determined the solution to be a web-based mapping application that could be operated through any internet browser. The chosen technologies (**Figure 2-1**) were selected to be consistent with other applications developed for BOEM. A more detailed description of the technical architecture can be found in **Appendix A**.

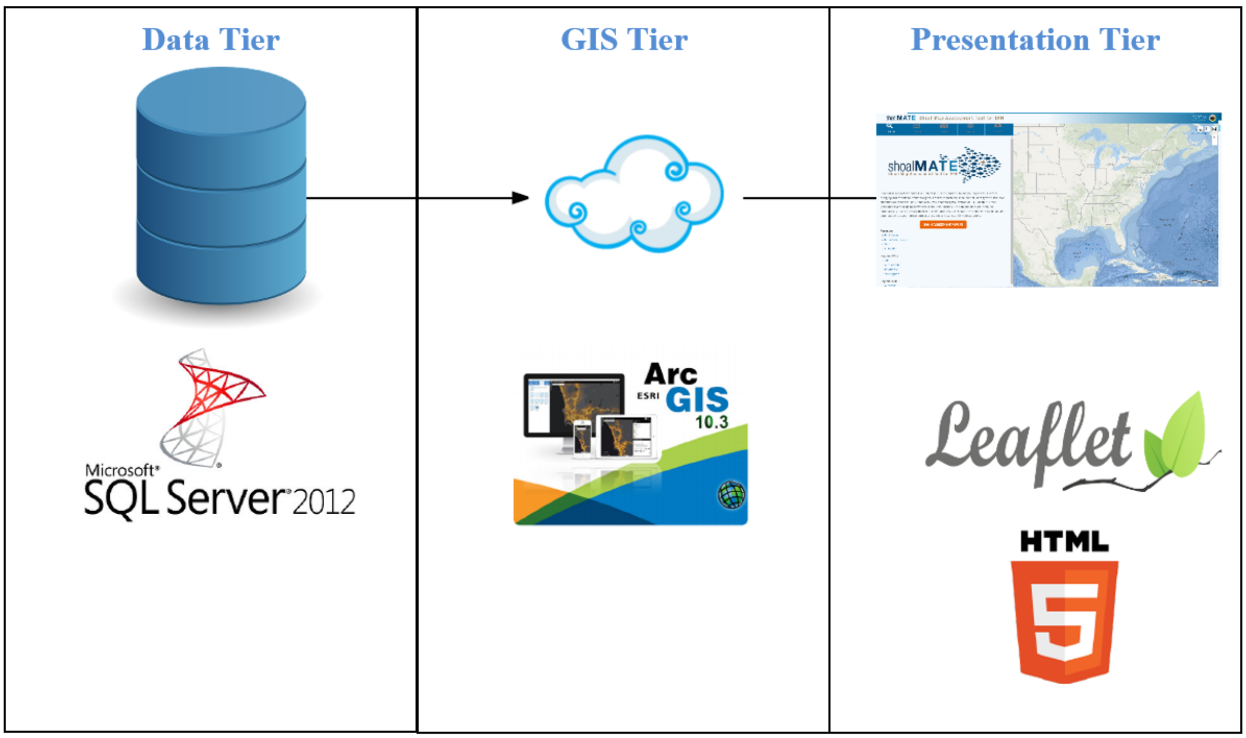

Figure 2-1. High-level architecture for the ShoalMATE application.

The high-level workflow for ShoalMATE involves five main steps (**Figure 2-2**).

- **Step 1: Select Shoal** – The user chooses an area of interest (AOI) and selects the relevant seasons in which dredging may occur.
- **Step 2: Review Results** – The user can review the results of intersecting the selected shoal feature with various data that will be utilized in the generated report.
- **Step 3: Review Maps** – The user can select and review a set of default maps with preset layers.
- **Step 4**: **Create Custom Maps** – To tell a more detailed story of the shoal, the user can choose to generate additional maps to include in the report by choosing from a variety of provided data layers.
- **Step 5: Generate Report** – The Reporting Tool compiles all the user inputs and results into an editable report in Microsoft Word document format.

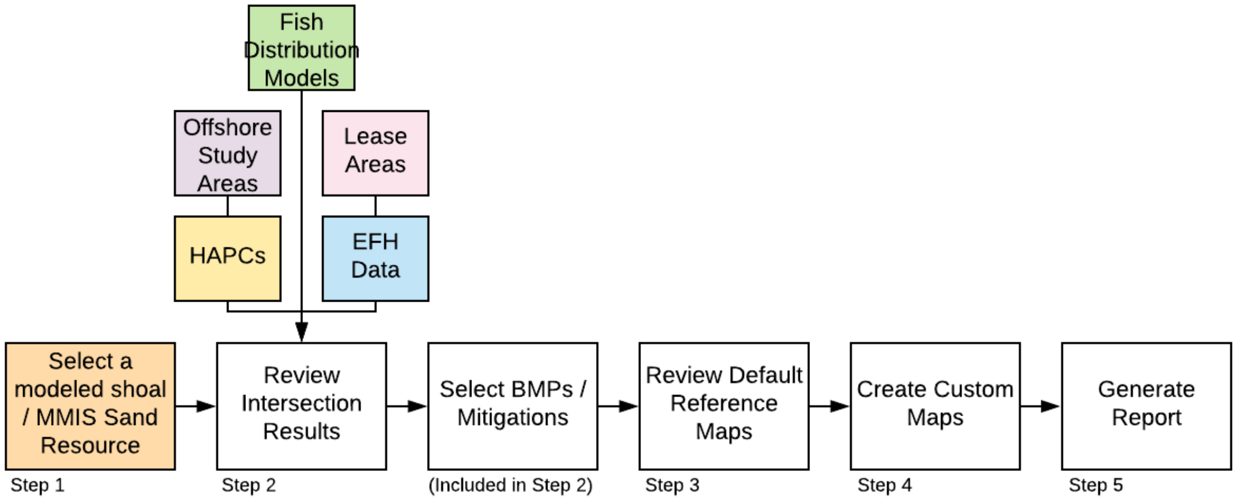

Figure 2-2. High-level workflow for the ShoalMATE application.

2.1 Data Development

To optimize the application's processing time, an Extract-Transform-Load (ETL) script was developed utilizing Python to compile data layer values into a shoal feature class, which became the scale of analysis for the tool. The feature class is a combination of the modeled shoals developed as part of this project and existing Marine Minerals Sand Resources. The analysis area available for ShoalMATE (**Figure 2-3**) includes Federal waters of the OCS to a 50-m depth. This range was driven by the depth limitations of dredge operations. Because of the large area covered, this "canning" of the data allows for significant performance improvements over conducting the analyses on the fly with each run of the tool.

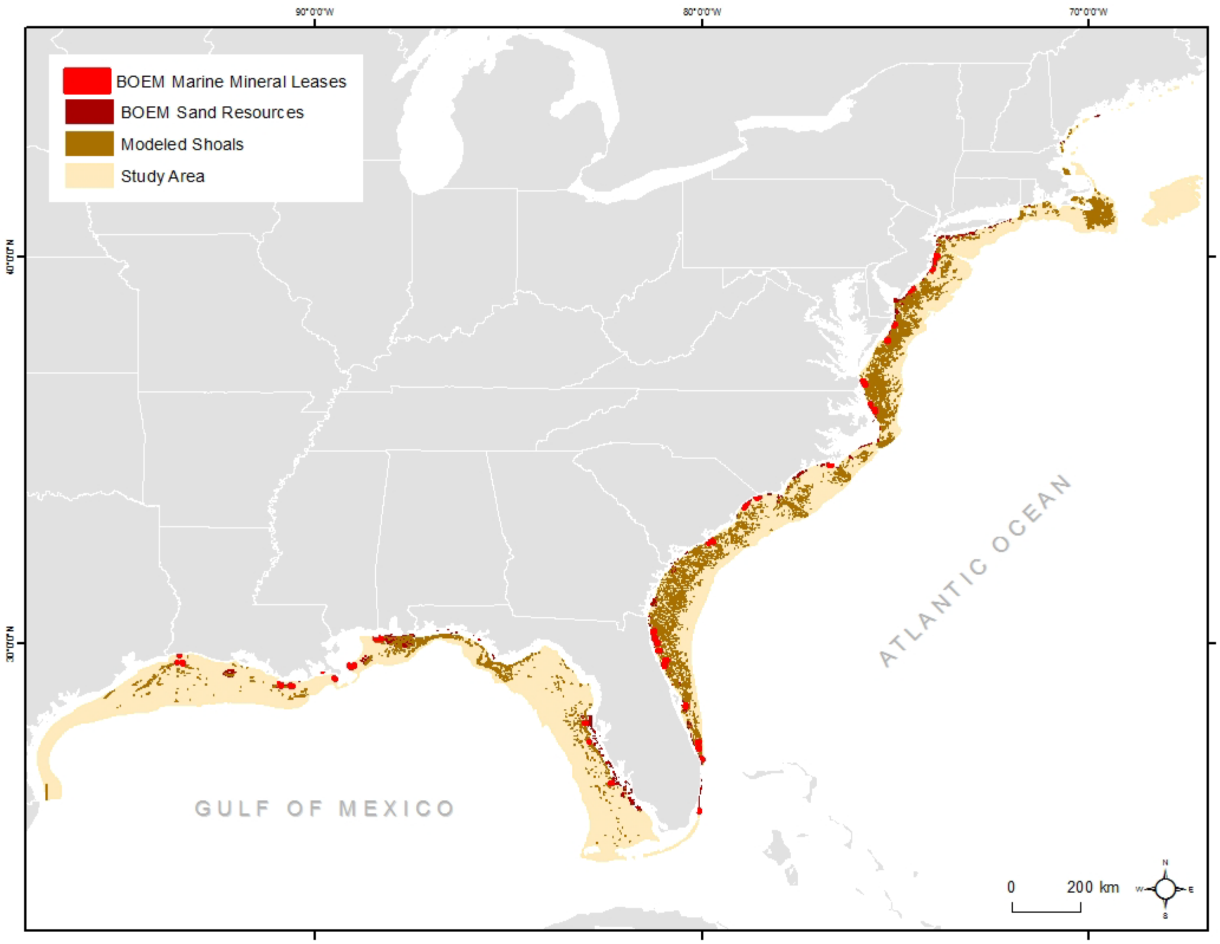

Figure 2-3. Extent of the ShoalMATE Tool represented in beige. Shoals and MMIS sand resources within this boundary are available for analysis within the tool.

Two primary categories of data exist within the ShoalMATE source database: vector data that characterizes presence, absence, or count of a seabed or political feature (e.g., seagrass or MMP leases) and continuous data (point and raster) that indicate a value (e.g., depth and SST).

For each shoal, the ETL performs one of two analyses on each data layer. For vector data, the script runs a spatial intersect and records the intersecting features as attributes for each shoal. For continuous data, a minimum, maximum, average or sum is calculated over the extent of the shoal and the value is stored as an attribute for each feature. This process was iterated for temporally discrete data layers so that an attribute exists for SST in January, SST in February, etc. The result is a set of over 10,000 shoals with a variety of information associated that can be used to describe the habitat of each one.

A second ETL was also developed to store the results of six intersections completed between individual shoals and six key datasets that provide critical information about a particular shoal's use (by fish species and humans) along with external resources to aid in further describing the shoal habitat in sufficient detail for meeting the requirements of a complete EFH Assessment. These results are presented in the tool and are populated in the generated report to resolve the requirements established in **Table 1-1** or to provide the user with additional resources to reference in the completion of the EFH Assessment document.

- **EFH Species Intersection** – Comparison between the shoal feature and the EFH polygons. Generates a list of species/life stage combinations that intersects with the shoal. Perform additional analysis to rank the potential for a species/life stage to be impacted by dredge operations within a certain time frame. Assign a qualitative value of High, Medium, or Low to each combination based on if the shoal habitat meets the documented habitat preferences of the species/lifestage (Figure 2-4).

- **Predicted Relative Abundance Models Intersection** – Summarizes shoal features with fish and invertebrate species distribution models developed or acquired as part of this study. Reports values of predicted mean relative abundance within the shoal alongside the predicted mean abundance for the surrounding area (within 20 km) and predicted abundance within each species' geographic range within each region (e.g., Gulf of Mexico). In this way, the data shows the importance of the shoal in the context of other available habitat in the region.
- **Predicted Probability of Presence Models Intersection** – Summarizes shoal features with fish and invertebrate species distribution models developed or acquired as part of this study. Reports values of predicted probability of presence within the shoal alongside the predicted probability of presence for the surrounding area (within 20 km) and within each species' geographic range within each region (e.g., Gulf of Mexico). In this way, the data shows the importance of the shoal in the context of other available habitat in the region.
- **Habitat Areas of Particular Concern (HAPC) Intersection** – Knowing what, if any, HAPCs intersect the shoal will allow for additional consideration of those areas and the species that may be affected.
- **MMIS Lease Area Intersection** – The intersection of a shoal with a previous lease indicates that the shoal has likely been dredged in the past. Information on the volume removed and what is still available as well as information to direct the user to the lease documentation which may include prior EFH Assessments that can aid in the completion of the manual portions of the generated report. This table is empty if there has been no prior dredging at the site.
- **MMIS Study Intersection** – This intersection may provide additional resources the user can reference when developing the report. A list of BOEM-funded studies by the MMP is provided and a link to the reports are included if available.

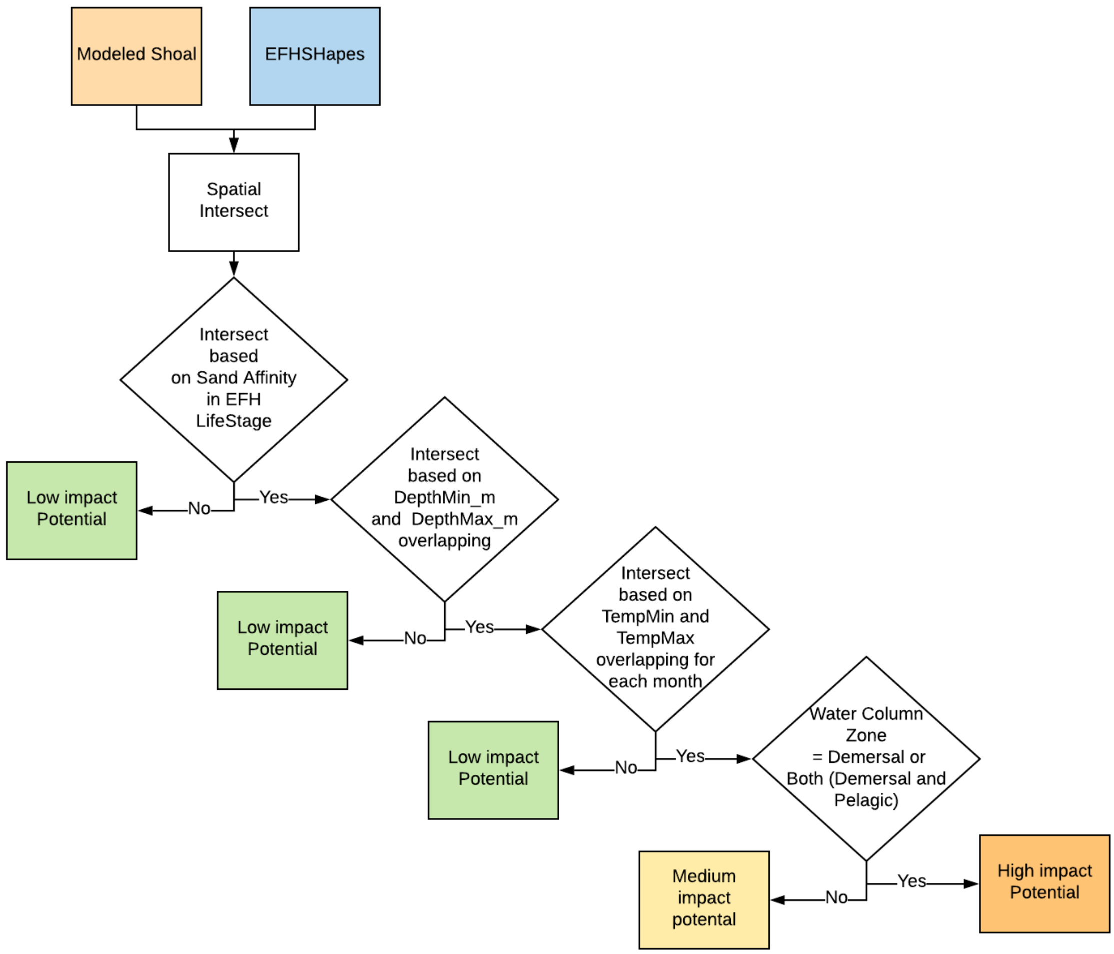

Figure 2-4. Impact potential logic.
Sand dredging impact potential for marine fish is assumed to be based on four main factors as depicted in the diagram: sand affinity, depth range, temperature range, and water column zone. The rankings result in either low, medium, or high potential impact.

A detailed user manual is provided in the Appendix, but a summary of the tool workflow is provided below.

2.2 Step 1 – Select Shoal

After initiating ShoalMATE, the user can zoom in to their AOI to select a shoal or sand resource of their choice. If multiple shoals are present where the user clicked, the user will have to specify by selecting one (**Figure 2-5**). The user must also select one or more seasons during which dredge is anticipated to occur.

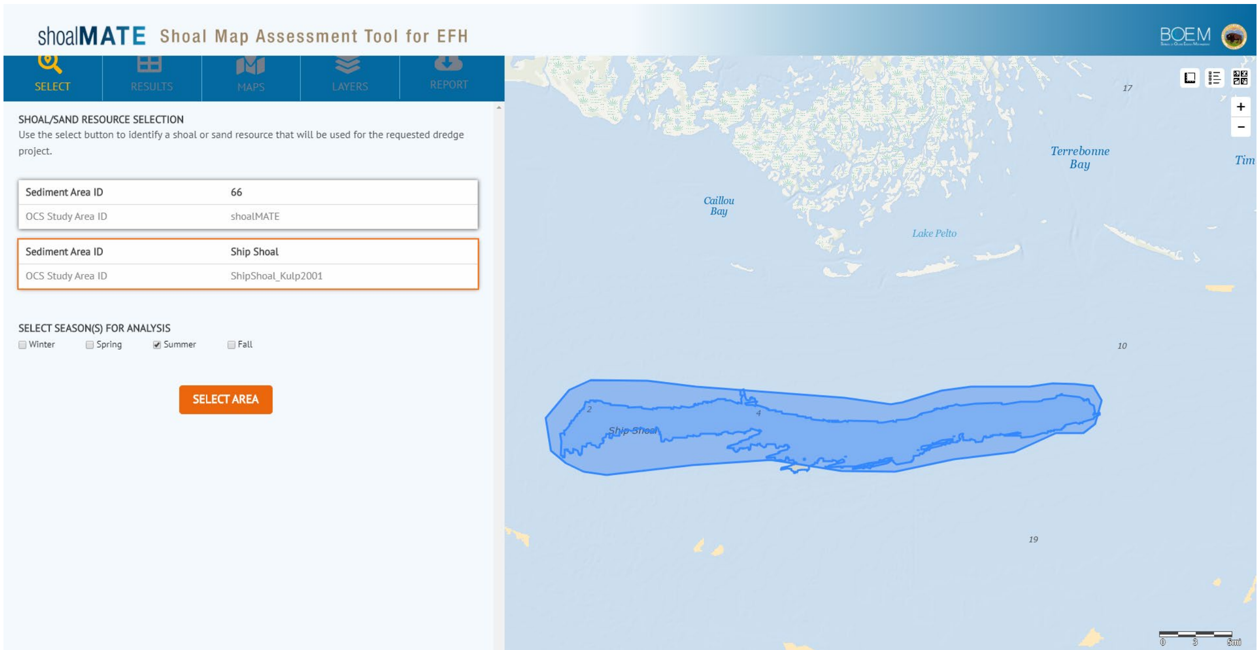

Figure 2-5. Selecting shoal/sand resource.
Available shoals in the selected area are shaded in blue. The shoal highlighted in orange on the left is the one selected for analysis during the summer season.

2.3 Step 2 – Review Results

Each "View" button will display the tabular information that will be carried into the generated report. The user can review this information before selecting mitigations and best management practices from the final "Continue" button. The tool provides a list of standard options that are found among the many existing EFH Assessment documents reviewed for this study (**Figure 2-6**). Several Best Management Practice (BMPs) and Mitigation Measure options have been included to capture past NOAA Conservation Recommendations, and these should be considered for each project.

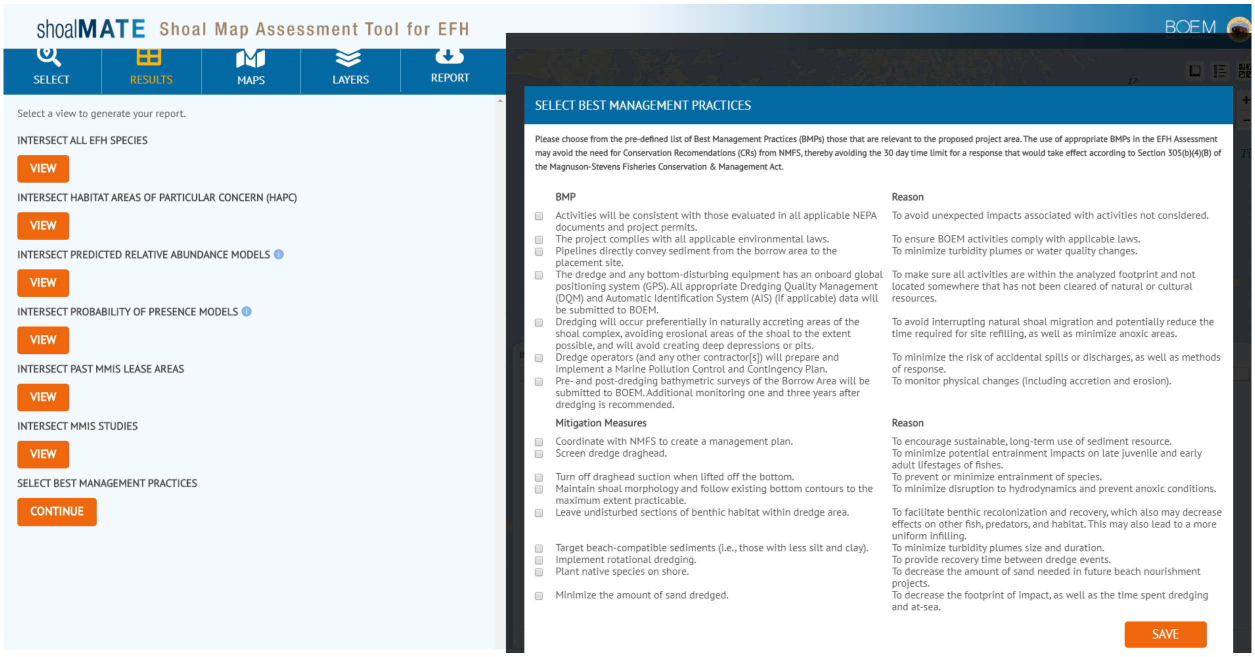

Figure 2-6. Pop up window for selecting BMPs and mitigation measures to be included in the report.

2.4 Step 3 – Review Maps

Step three of the workflow contains five preset maps (**Figure 2-7**) developed to meet the requirements of the EFH Assessment.

1) Overview Map – to provide a sense of location of the shoal
2) Bathymetry Map – to provide a view of the surrounding elevation as well as the prevailing current directions for the seasons selected at the start of the tool.
3) Substrate Map – to indicate the characteristics of the surrounding substrate and any substrate features that are known to influence fish distribution (e.g., artificial reef, oil platforms, natural reefs)
4) Accretion Map – for areas where two or more previous dredge events have occurred, accretion maps are generated by determining the difference between dredge events using pre- and post-dredge surveys. This allows some insight into how the area has recovered between events. Note that these data are still in prototype and not available in all dredged sites yet.
5) Dredge Exposure Map – displays in time units how long a dredge vessel was within an area. This data is currently only available for hopper dredges and is not available for all dredged sites.

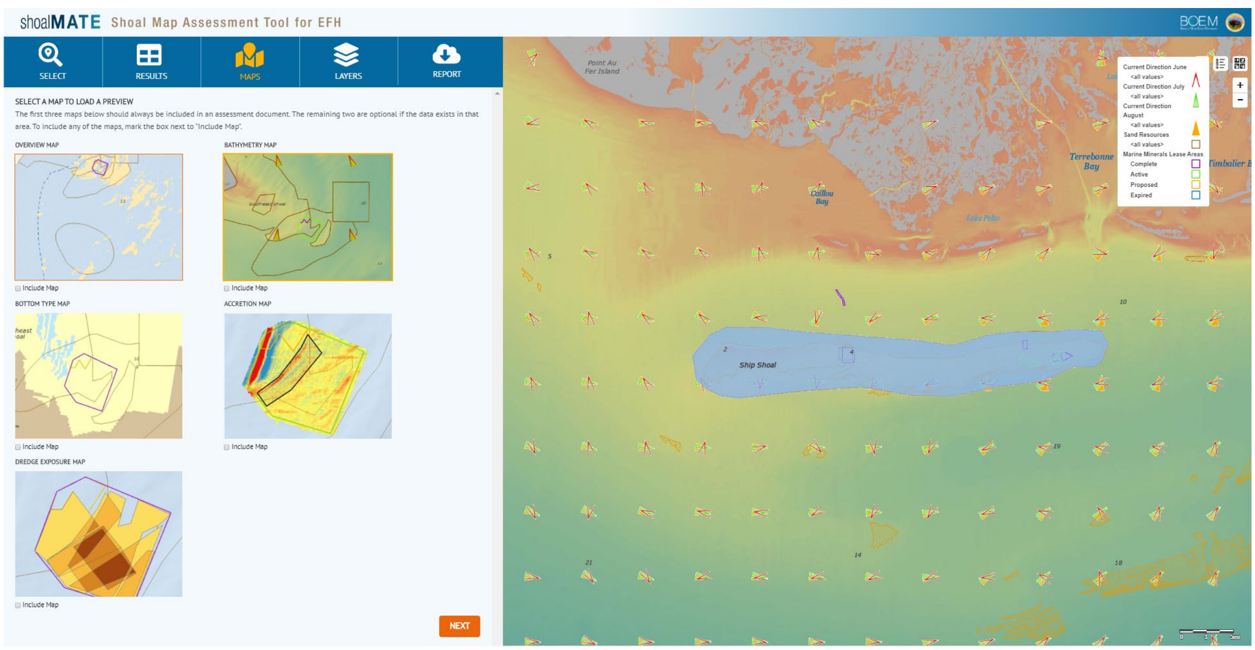

Figure 2-7. Selecting a map to load a preview.
(Left) Thumbnails of available preset maps. At a minimum, the overview, bathymetry, and bottom type maps should always be included. (Right) Bathymetry map with prevailing bottom current directions for summer months overlaid.

2.5 Step 4 – Create Custom Maps

Additional data sources accumulated for this project are also available to generate maps outside of the five default maps to include in the report. A selection of over 100 data layers are available to map. The user can create multiple custom maps (for individual species distributions, for instance) by saving them to the report and then clearing the layers and starting over (**Figure 2-8**).

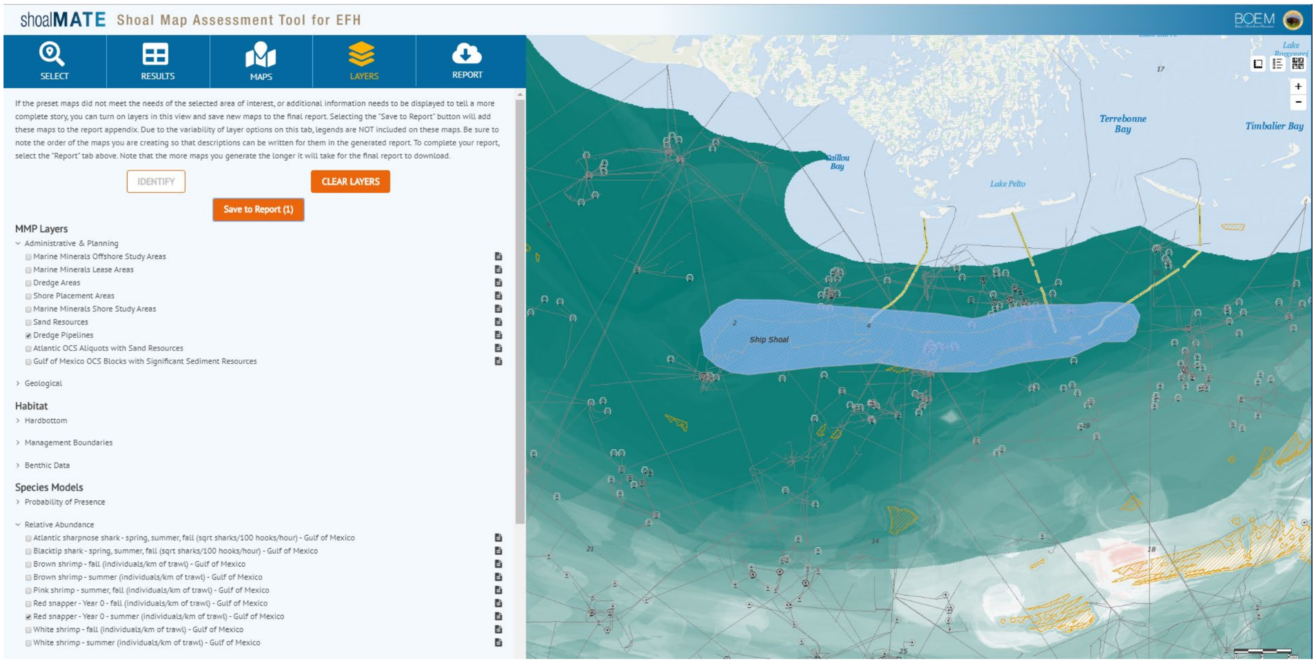

Figure 2-8. A custom map displaying juvenile red snapper relative abundance in relation to OCS drilling platforms and oil and gas pipelines.

2.6 Step 5 – Generate Report

User selections are stored throughout the run of the tool, so a report containing a summation of results can be generated upon completion (**Figure 2-9**). The report is exported as a Word document to be stored locally **(Appendix B: Example Report from ShoalMATE Reporting Tool)**. The generated report is formatted and includes all information gathered from the tool. Within the report are additional prompts that users must manually complete to satisfy the remaining requirements for the EFH Assessment. Having a large portion of this information already identified via the intersection tables gives the user easy access to share it with planning partners.

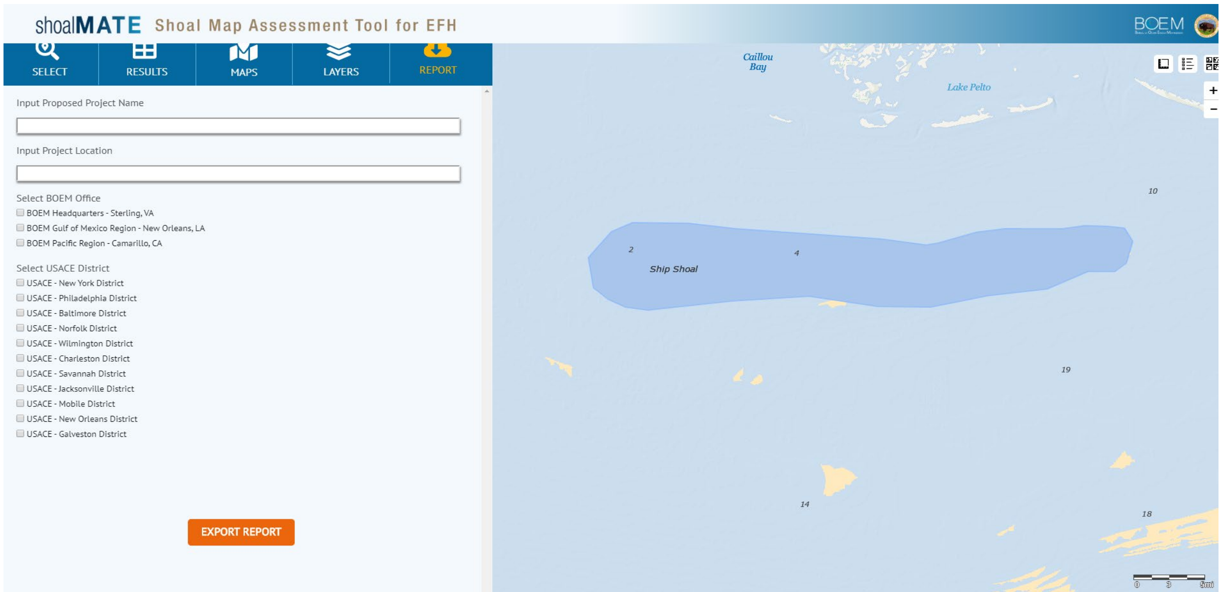

Figure 2-9. Preview of the report export page of the ShoalMATE tool.

2.7 Potential Improvements and Future Work

Through the course of development, new needs were generated that exceeded the scope and/or timeline of this project. The implementation of these needs would result in a more accurate and/or more robust tool and should be considered. A list of key suggestions is provided in **Table 2-1**.

Table 2-1. Suggested improvements to the ShoalMATE tool for future development

ID	Improvements	Logic	Effort
1	Create a user-friendly interface for the back-end data processing	To update the shoal feature class on regular intervals as new MMIS sand resources are identified, the ETL processes need to be re-run to include the new feature in the tool. Currently, the process is run through a series of Python scripts with minimal graphical user interface (GUI). Advanced users only should update the database.	Low
2	Incorporate additional project data	Identify how to incorporate cutter head dredge operations into the exposure raster generation process so that those maps can be completed. Determine workflows for fully developing the accretion rasters to make that information more widely available.	Medium
3	Add additional habitat descriptors to the report	Provide distance to hard bottom features in the study region.	Low

Appendix A: User Manual for ShoalMATE: Shoal Map Assessment Tool for EFH

The following pages contain the complete user manual for ShoalMATE. It is presented in original format to be consistent with the ShoalMATE reporting tool.

BOEM
Bureau of Ocean Energy Management

shoalMATE
Shoal Map Assessment Tool for EFH

Reporting Tool User Manual

Authors: **Emily Sandrowicz**

Revision: **1.0, 9/16/2019**

Office for Coastal Management
National Oceanic and Atmospheric Administration

Quantum Spatial

CSS

NCCOS
National Centers for Coastal Ocean Science

Document Information

General Information

Project Name	shoalMATE: Shoal Map Assessment Tool for EFH		
Prepared By	Emily Sandrowicz	Document Version	1.0
Title	shoalMATE User Manual	Document Version Date	9/16/2019
Reviewed By		Review Date	

History

Ver. No.	Ver. Date	Revised By	Description	Filename
1.0	9/16/2019	Emily Sandrowicz	How-to guide for shoalMATE	shoalMATE_User_Manual.docx

Distribution List

From	Date	Contact Information

To	Action	Due Date	Contact Information

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Table of Contents

1 Summary

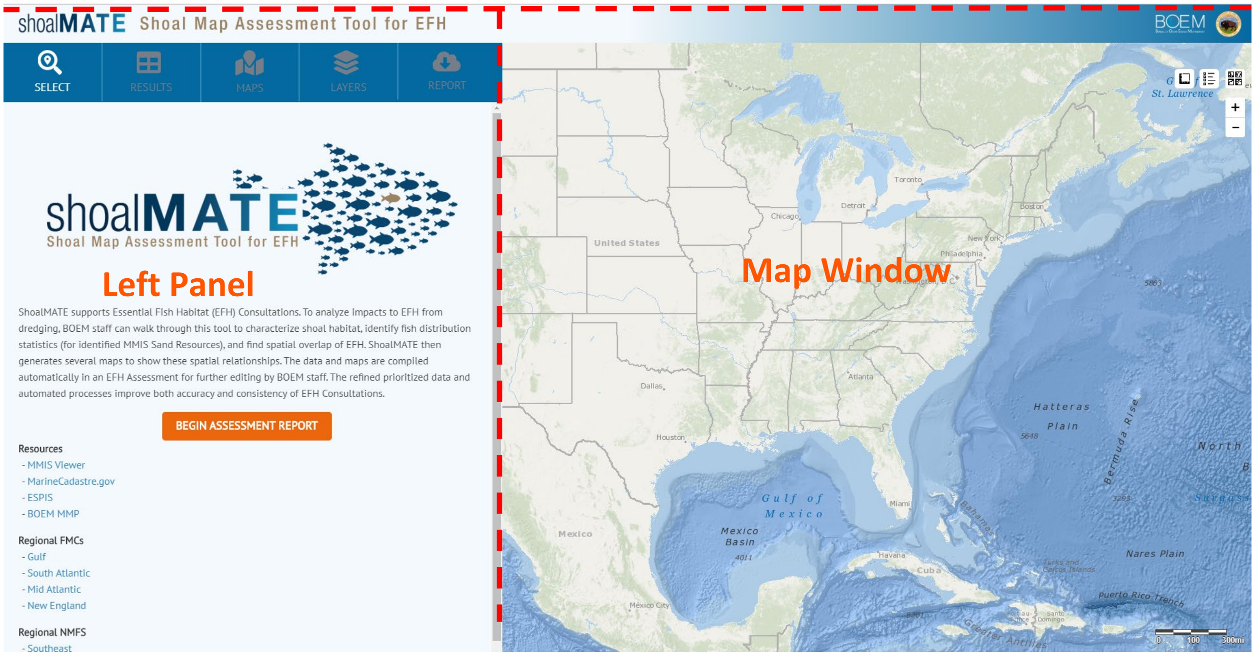
Quantum Spatial, Inc. (QSI) was contracted by the National Oceanic and Atmospheric Administration (NOAA), through an interagency agreement with the Bureau of Ocean Energy Management (BOEM), to develop an assessment tool that integrates multiple data sources within a simple and standardized user interface to support environmental assessments.

BOEM's Marine Minerals Program (MMP) is tasked with managing the use of marine minerals on the Outer Continental Shelf (OCS) in an environmentally responsible way. Through execution of this project, BOEM will develop a tool to help analyze the impact of dredging on Essential Fish Habitat (EFH). Such OCS sediment resource dredging projects are designed to support shore protection and coastal restoration projects along the Atlantic and Gulf of Mexico coasts. The purpose of the tool is to generate reports intended to assist in EFH consultations with the National Marine Fisheries Service (NMFS) by analyzing shoal habitat, identifying fish distribution statistics, and finding resulting overlap of EFH in a user-specified area and season. Information for this tool was gathered from EFH documentation, literature reviews, scientific models, readily available data sources, subject matter experts, and the MMIS.

This document contains information on how to use the Shoal Map Assessment Tool for EFH (shoalMATE) reporting tool. Also included is information on how to interpret the results and the logic that went into getting them.

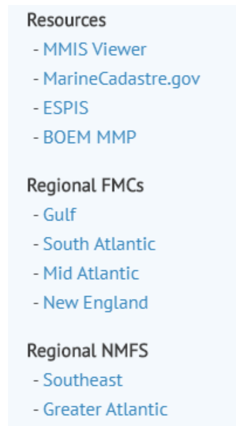
2 Getting Started

1. Open a web browser from within the DOI network and navigate to https://mmisdev.bc.doi.net/shoalMATE. The shoalMATE homepage has three sections, the **Title Bar**, **Left Panel** and **Map Window**.



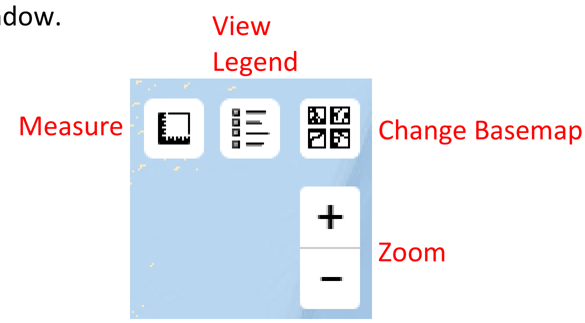
2. The **Title Bar** holds links to various webpages that users may find helpful.
    a. Clicking on the shoalMATE Logo will return users to the shoalMate home screen
    b. Clicking on the BOEM or DOI Logo will open the respective entity's home page

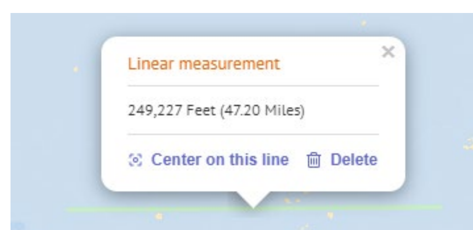
3. There are also links to other resources at the bottom of the **Left Panel**, including additional data portals, regional FMCs, and regional NMFSs.

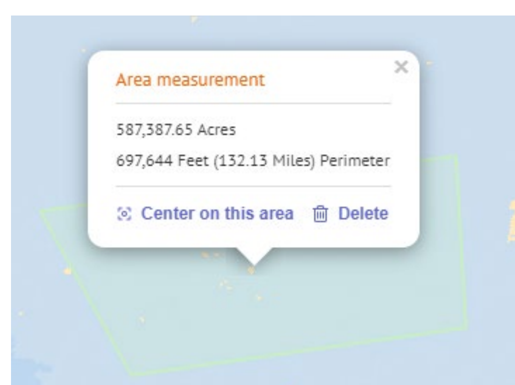


2.1 Map Window

The **Map Window** displays the mappable area and also contains useful tools. Users can measure distances and areas, view the legend, switch the base map and zoom in/out with the tools in the top right corner of the Map Window.







1. To measure, select the white measuring square and click ⊖ Create a new measurement from the popup.
2. Lines are measured by left clicking on the map where the desired measurement is to begin, then double-clicking where the line is to end.
3. Areas are measured in a similar manner, but with additional vertices added via left click in between the starting and ending points. Click again at the next point in the measurement.
4. Measurements are removed from the map by clicking the Delete icon in the measurement dialog box.

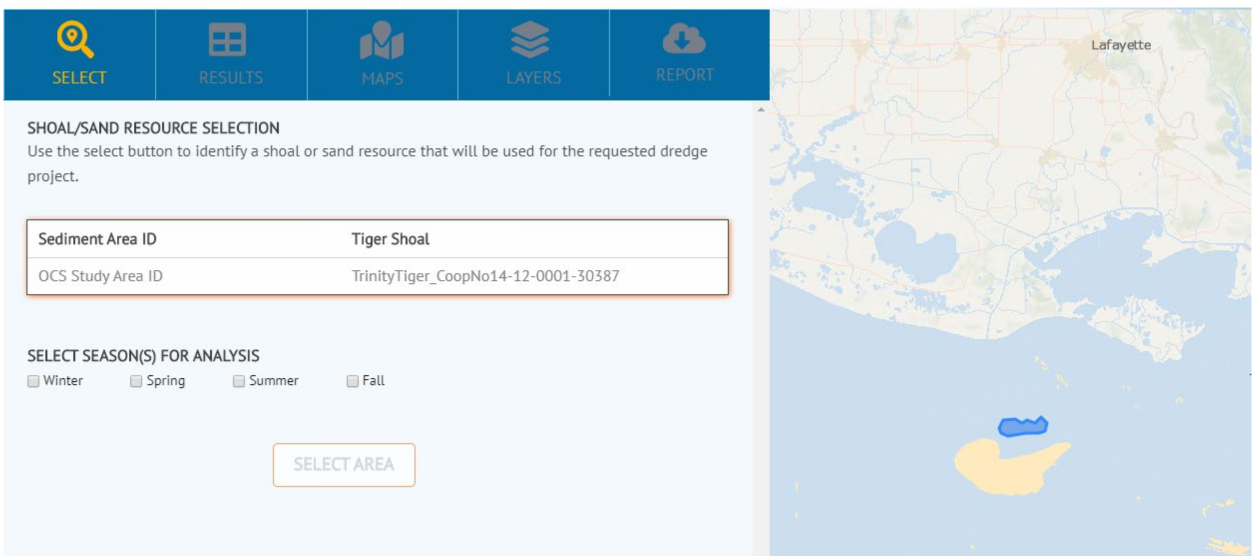
2.2 Left Panel

Users will be guided through the tabs on the **Left Panel** as they select criteria to customize their report.

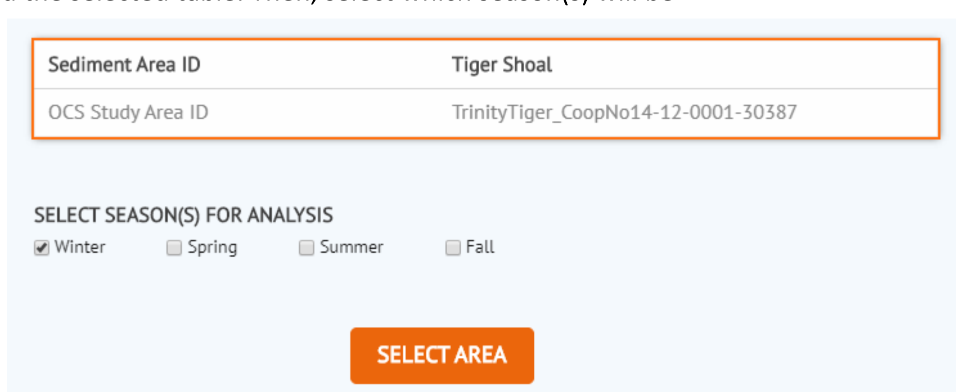
3 Generating Reports

3.1 SELECT Tab

1. Begin by clicking the **BEGIN ASSESSMENT REPORT** button on the **SELECT Tab** homepage.
2. Pan and zoom the map until the desired area is centered in the **Map Window**. Click on a shoal to view its details.

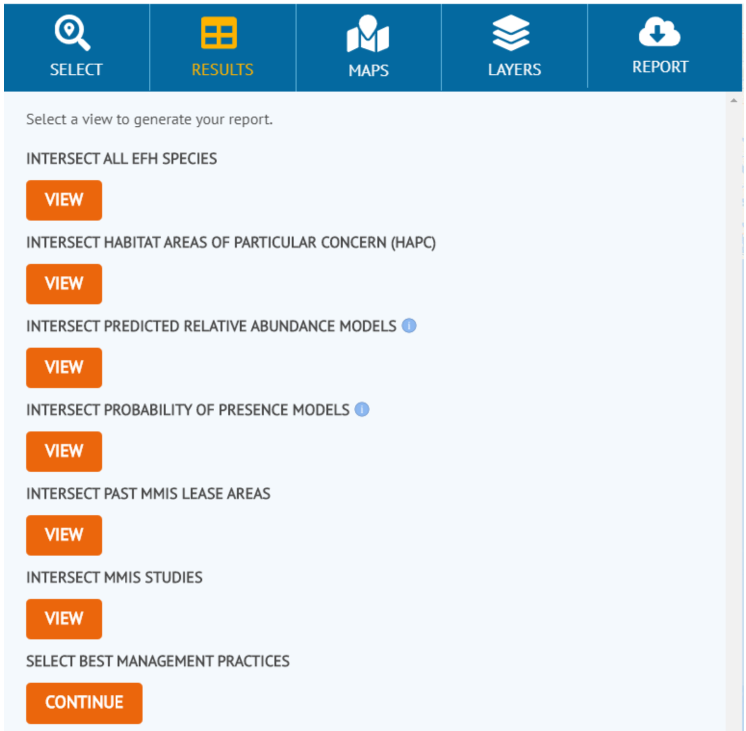


3. Select the shoal by clicking on the table containing the Sediment Area ID and the OCS Study Area ID. An orange border will appear around the selected table. Then, select which season(s) will be included in the analysis. This will affect data visible in the results table and maps. Once the selections are made, click the **SELECT AREA** button to advance to the **RESULTS Tab**. In this example Tiger Shoal and Winter have been selected.

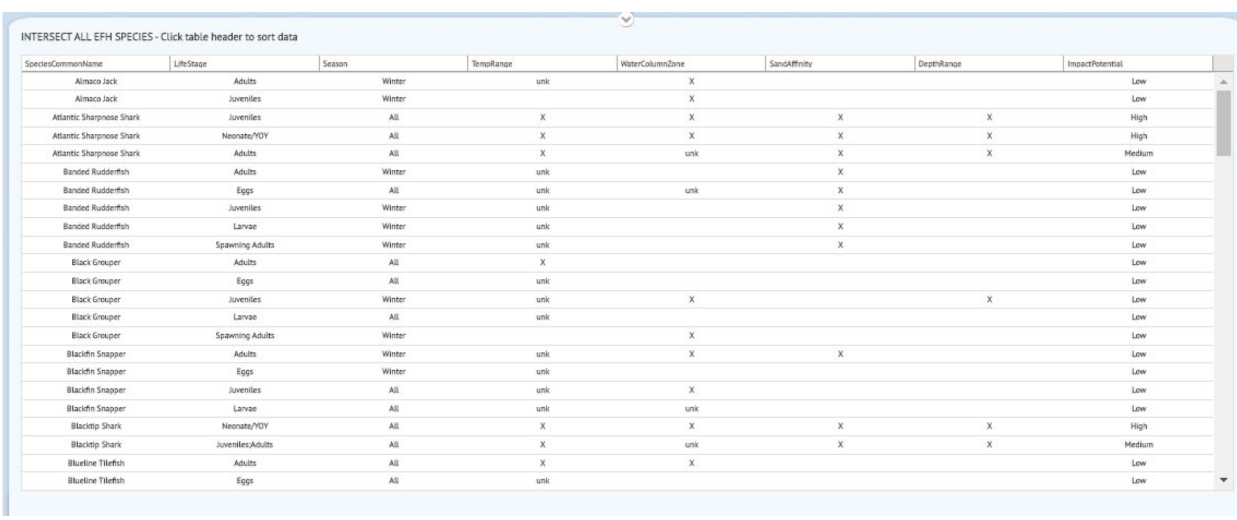


3.2 RESULTS Tab

1. The **RESULTS Tab** provides access to 6 data tables and a list of Best Management Practices.



2. Each of the 6 result reports contains the subset of data from the stated source that intersects the selected shoal and season. For this example, by clicking on the VIEW button below the INTERSECT ALL EFH SPECIES heading, users can view all EFH Species that have the same spatial and/or temporal extent as Tiger Shoal during the winter.


INTERSECT ALL EFH SPECIES - Click table header to sort data

SpeciesCommonName	LifeStage	Season	TempRange	WaterColumnZone	SandAffinity	DepthRange	ImpactPotential
Almaco Jack	Adults	Winter	unk	X			Low
Almaco Jack	Juveniles	Winter		X			Low
Atlantic Sharpnose Shark	Juveniles	All	X	X	X	X	High
Atlantic Sharpnose Shark	Neonate/YOY	All	X	X	X	X	High
Atlantic Sharpnose Shark	Adults	All	X	unk	X	X	Medium
Banded Rudderfish	Adults	Winter	unk		X		Low
Banded Rudderfish	Eggs	All	unk	unk	X		Low
Banded Rudderfish	Juveniles	Winter	unk		X		Low
Banded Rudderfish	Larvae	Winter	unk		X		Low
Banded Rudderfish	Spawning Adults	Winter	unk		X		Low
Black Grouper	Adults	All	X				Low
Black Grouper	Eggs	All	unk				Low
Black Grouper	Juveniles	Winter	unk	X		X	Low
Black Grouper	Larvae	All	unk				Low
Black Grouper	Spawning Adults	Winter		X			Low
Blackfin Snapper	Adults	Winter	unk	X	X		Low
Blackfin Snapper	Eggs	Winter	unk				Low
Blackfin Snapper	Juveniles	All	unk	X			Low
Blackfin Snapper	Larvae	All	unk	unk			Low
Blacktip Shark	Neonate/YOY	All	X	X	X	X	High
Blacktip Shark	Juveniles;Adults	All	X	unk	X	X	Medium
Blueline Tilefish	Adults	All	X	X			Low
Blueline Tilefish	Eggs	All	unk				Low

3. Clicking on the **CONTINUE** button below the SELECT BEST MANAGEMENT PRACTICES heading brings up a list of Best Management Practices (BMP) and Mitigation Measures and their associated reasons for implementation. Users will check the box next to the BMP and Mitigations Measures that are relevant to their project. Each checked box will trigger the inclusion of associated pre-written text about the BMP or Mitigation Measure in the final report. The use of appropriate BMPs and/or Mitigation Measures in the EFH Assessment may avoid the need for Conservation Recommendations (CRs) from NMFS.

SELECT BEST MANAGEMENT PRACTICES

Please choose from the pre-defined list of Best Management Practices (BMPs) those that are relevant to the proposed project area. The use of appropriate BMPs in the EFH Assessment may avoid the need for Conservation Recomendations (CRs) from NMFS, thereby avoiding the 30 day time limit for a response that would take effect according to Section 305(b)(4)(B) of the Magnuson-Stevens Fisheries Conservation & Management Act.

BMP	Reason
☐ Activities will be consistent with those evaluated in all applicable NEPA documents and project permits.	To avoid unexpected impacts associated with activities not considered.
☐ The project complies with all applicable environmental laws.	To ensure BOEM activities comply with applicable laws.
☐ Pipelines directly convey sediment from the borrow area to the placement site.	To minimize turbidity plumes or water quality changes.
☐ The dredge and any bottom-disturbing equipment has an onboard global positioning system (GPS). All appropriate Dredging Quality Management (DQM) and Automatic Identification System (AIS) (if applicable) data will be submitted to BOEM.	To make sure all activities are within the analyzed footprint and not located somewhere that has not been cleared of natural or cultural resources.
☐ Dredging will occur preferentially in naturally accreting areas of the shoal complex, avoiding erosional areas of the shoal to the extent possible, and will avoid creating deep depressions or pits.	To avoid interrupting natural shoal migration and potentially reduce the time required for site refilling, as well as minimize anoxic areas.
☐ Dredge operators (and any other contractor[s]) will prepare and implement a Marine Pollution Control and Contingency Plan.	To minimize the risk of accidental spills or discharges, as well as methods of response.
☐ Pre- and post-dredging bathymetric surveys of the Borrow Area will be submitted to BOEM. Additional monitoring one and three years after dredging is recommended.	To monitor physical changes (including accretion and erosion).

Mitigation Measures	Reason
☐ Coordinate with NMFS to create a management plan.	To encourage sustainable, long-term use of sediment resource.
☐ Screen dredge draghead.	To minimize potential entrainment impacts on late juvenile and early adult lifestages of fishes.
☐ Turn off draghead suction when lifted off the bottom.	To prevent or minimize entrainment of species.
☐ Maintain shoal morphology and follow existing bottom contours to the maximum extent practicable.	To minimize disruption to hydrodynamics and prevent anoxic conditions.
☐ Leave undisturbed sections of benthic habitat within dredge area.	To facilitate benthic recolonization and recovery, which also may decrease effects on other fish, predators, and habitat. This may also lead to a more uniform infilling.
☐ Target beach-compatible sediments (i.e., those with less silt and clay).	To minimize turbidity plumes size and duration.
☐ Implement rotational dredging.	To provide recovery time between dredge events.
☐ Plant native species on shore.	To decrease the amount of sand needed in future beach nourishment projects.
☐ Minimize the amount of sand dredged.	To decrease the footprint of impact, as well as the time spent dredging and at-sea.
☐ Dredge over large, shallow areas rather than in deep pits.	To decrease the potential for anoxic areas.
☐ Dredge shoal crests and higher areas of shoals.	To facilitate more rapid sediment reworking and site infilling.
☐ Dredge shoals in <30 m depths.	To facilitate regrowing after dredging.
☐ Dredge shoals with Relative Shoal Height (defined as Height/Base Depth) of more than 0.5.	To facilitate regrowing after dredging.

Users also have the option to compose their own BMP for inclusion in the report at the bottom of the page. Scroll down to access this section. Click the **SAVE** button when complete.

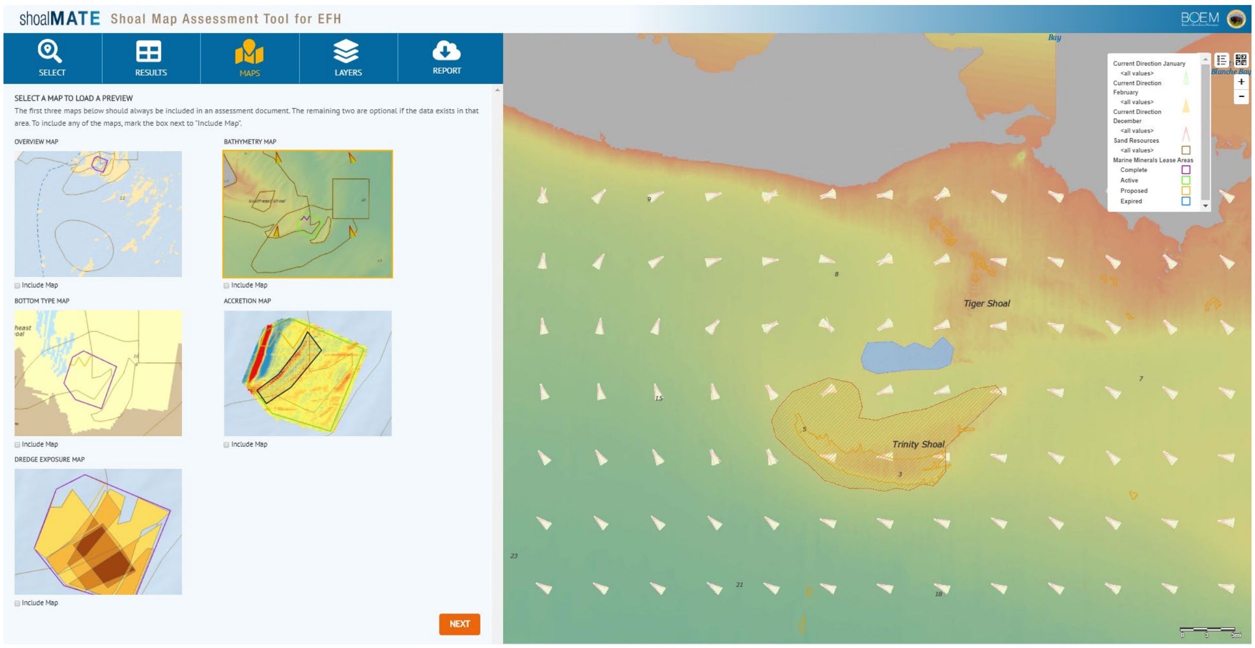
Add New Best Management Practice

SAVE

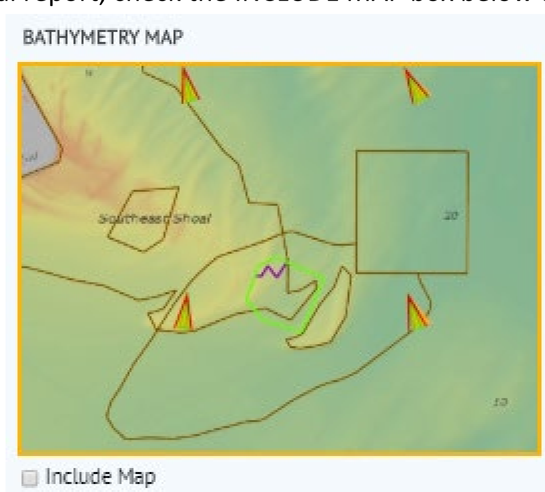
3.3 MAPS Tab

The **MAPS Tab** allows users to view 5 pre-defined map products and select which, if any, will be included in their report.

1. Clicking on a map's thumbnail image displays a preview of the map in the Map Window. When activated, an orange border will appear around the thumbnail.



2. To include a map in the final report, check the INCLUDE MAP box below the appropriate thumbnail.



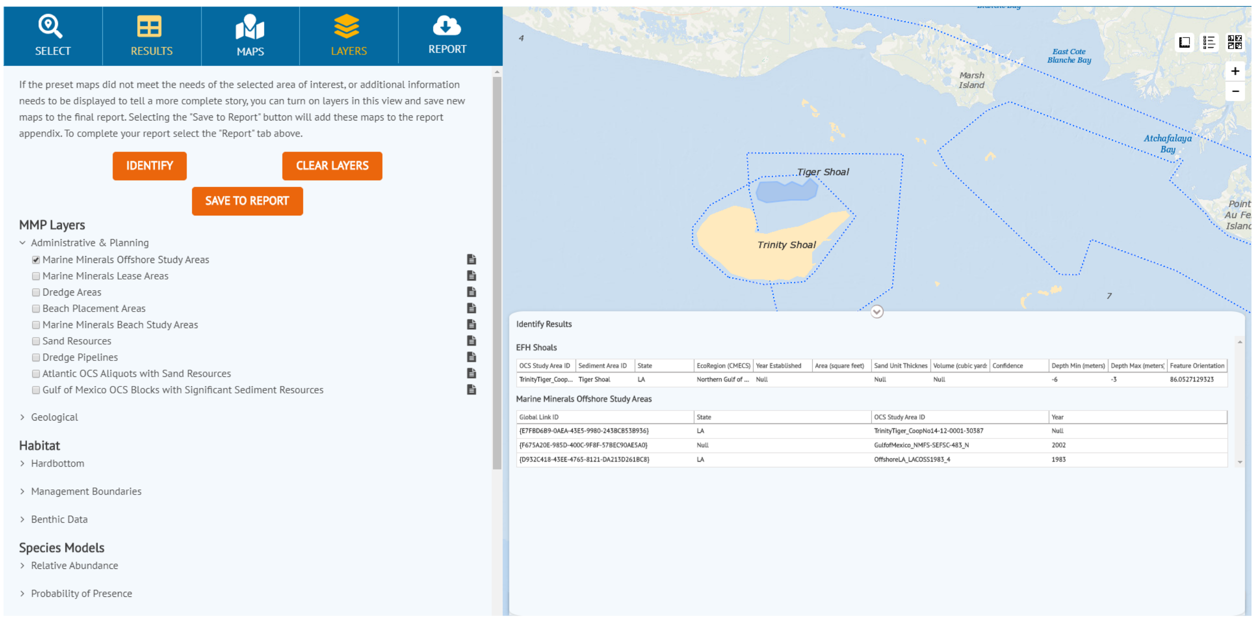
3. Currently the Accretion Map and Dredge Exposure Map are still under development and will not yield any data for display.

3.4 LAYERS Tab

The **LAYERS Tab** enables users to create their own cartographic products for inclusion in their final report. This functionality should be used when the preset maps available on the MAPS Tab do not meet all of the needs of the selected area of interest, or if additional information would be helpful in telling a more complete story.

The **IDENTIFY** button provides additional information on the features in the available layers.

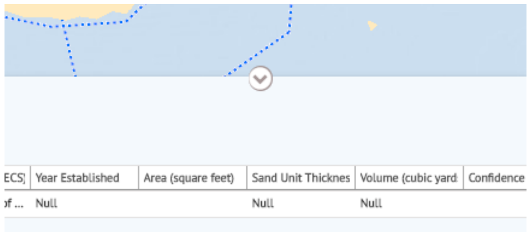
1. Expand the Administrative and Planning dataset by clicking on the arrow next to the heading. Then, check the box next to Marine Minerals Offshore Study Areas.
2. Activate the tool by clicking on the **IDENTIFY** button.
3. Click on Tiger Shoal in the Map Window. Information on the shoal and the overlapping Marine Minerals Offshore Study Areas will appear in a Results window.



4. **Metadata** for each layer can be accessed by clicking on the metadta icon next to the layer name when available. Some layers were provided by individual communication and do not contain complete FGDC metadata.

Users can create multiple maps for inclusion in the report, each showing one or more layers, with the use of the **SAVE TO REPORT** and **CLEAR LAYERS** buttons.

5. Minimize the Identify Results by clicking on the down arrow in the top center of the window.



6. Turn on the Gulf of Mexico OCS Blocks with Significant Sediment Resources by checking the box next to the layer.
7. Click the **SAVE TO REPORT** button to export a map with the selected layers to the final report.

8. Click the **CLEAR LAYERS** button to deactivate all selected layers and return to a blank map.

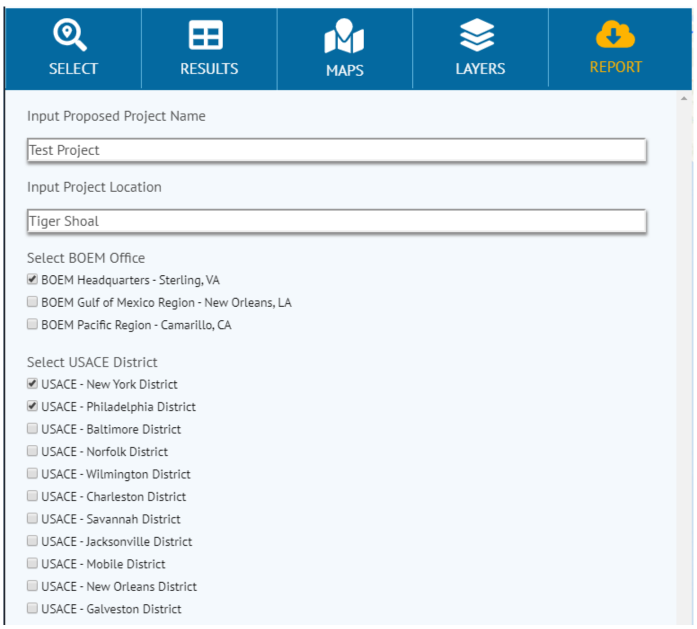
9. Continue creating and exporting additional maps as desired in this fashion. A count of custom maps is noted on the **SAVE TO REPORT** button. Importantly, the user should keep a chronological record of the layers included in each custom map. Due to the variability of layer combinations a legend is not included in the exported map and the user will have to describe the map contents in the caption.

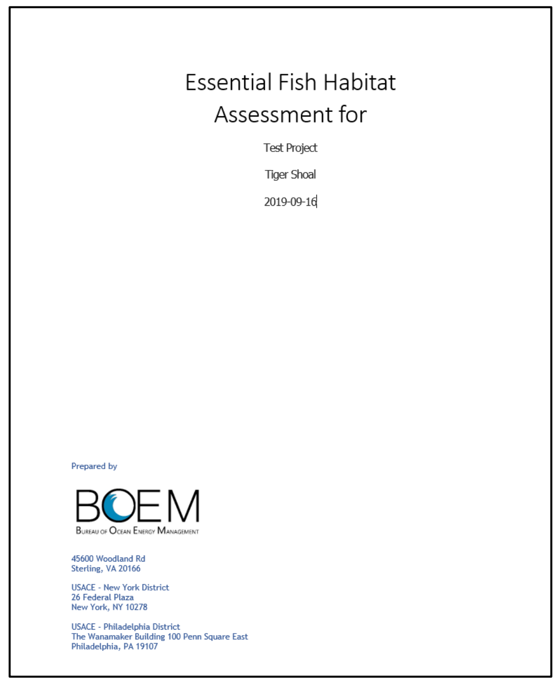
NOTE: The more custom maps that are saved to the report the longer the final export of the report will take.

3.5 REPORT Tab

Additional formatting can be added on the **REPORT Tab**.

1. Type the appropriate information in the Input Proposed Project Name and Input Project Location fields. The tool will automatically enter these values on the title page and at other predetermined locations in the final report.
2. Select the BOEM Office(s) and, as applicable, USACE District(s) involved in the project from the available lists. The tool will add the appropriate office addresses to the title page of the report based on these selections.





3. Click the **EXPORT REPORT** button to create the final report in .doc format.

4. The report will be populated with all the information selected on the previous tabs. However, there are some sections that will need to be populated manually after the report is created, such as the opening paragraph of the Introduction Section shown here.

I. Introduction

[The following information should be input manually:

- *Description of why project is proposed/why they need sand on the beach.*
- *Brief description of past projects, if any. This section is expanded on in Section 3.*
- *Who prepared this assessment and why (1 paragraph)*
- *Description of the physical location of the project and coastal features that it is most adjacent to. This section is expanded on in Section 3.]*

See Maps 1-3 for more information on the proposed borrow area and its surrounding environment including bathymetry, bottom currents, and seafloor substrate.

Additional information regarding the proximity of the proposed project to features of interest not covered in this report can be obtained through BOEM and NOAA's Ocean Reporting Tool (NOAA 2018b).

5. Do a document search for brackets ([ ]) to ensure all manual portions of the report have been addressed.
6. Remove any default map captions that were not selected for inclusion in the report.
7. Appendix A of the report contains the varibles utilized in the species models associated with the region of the selected feature
8. Appendix B includes all custom maps.

Appendix B: Example Report from ShoalMATE Reporting Tool

The following pages contain an example EFH assessment report from ShoalMATE. It is presented in original format to be consistent with the ShoalMATE reporting tool.

Essential Fish Habitat Assessment for

Sand Dredging Test1

2019-12-19

Prepared by

BOEM
Bureau of Ocean Energy Management

45600 Woodland Rd
Sterling, VA 20166

USACE - Wilmington District
69 Darlington Ave
Wilmington, NC 28403

Table of Contents

[Table of Contents may need to be updated after export and editing of this report.]

List of Abbreviations and Acronyms

BOEM	Bureau of Ocean Energy Management
CFR	Code of Federal Regulations
cm	centimeter(s)
CMECS	Coastal and Marine Ecological Classification Standard
cy	cubic yards
EA	Environmental Assessment
EEZ	Exclusive Economic Zone
EFH	Essential Fish Habitat
EIS	Environmental Impact Statement
FMC	Fisheries Management Council
FMP	Fisheries Management Plan
ft	foot/feet
GOM	Gulf of Mexico
in	inch(es)
km	kilometer
m	meter(s)
m^3	cubic meters
mm	millimeter(s)
MMP	Marine Minerals Program
MSFCMA	Magnuson-Stevens Fisheries Conservation & Management Act
NEPA	National Environmental Policy Act
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
ppt	parts per thousand
TSS	total suspended sediments
SS	suspended sediments
unk	unknown
USACE	U.S. Army Corps of Engineers

I. Introduction

[The following information should be input manually:

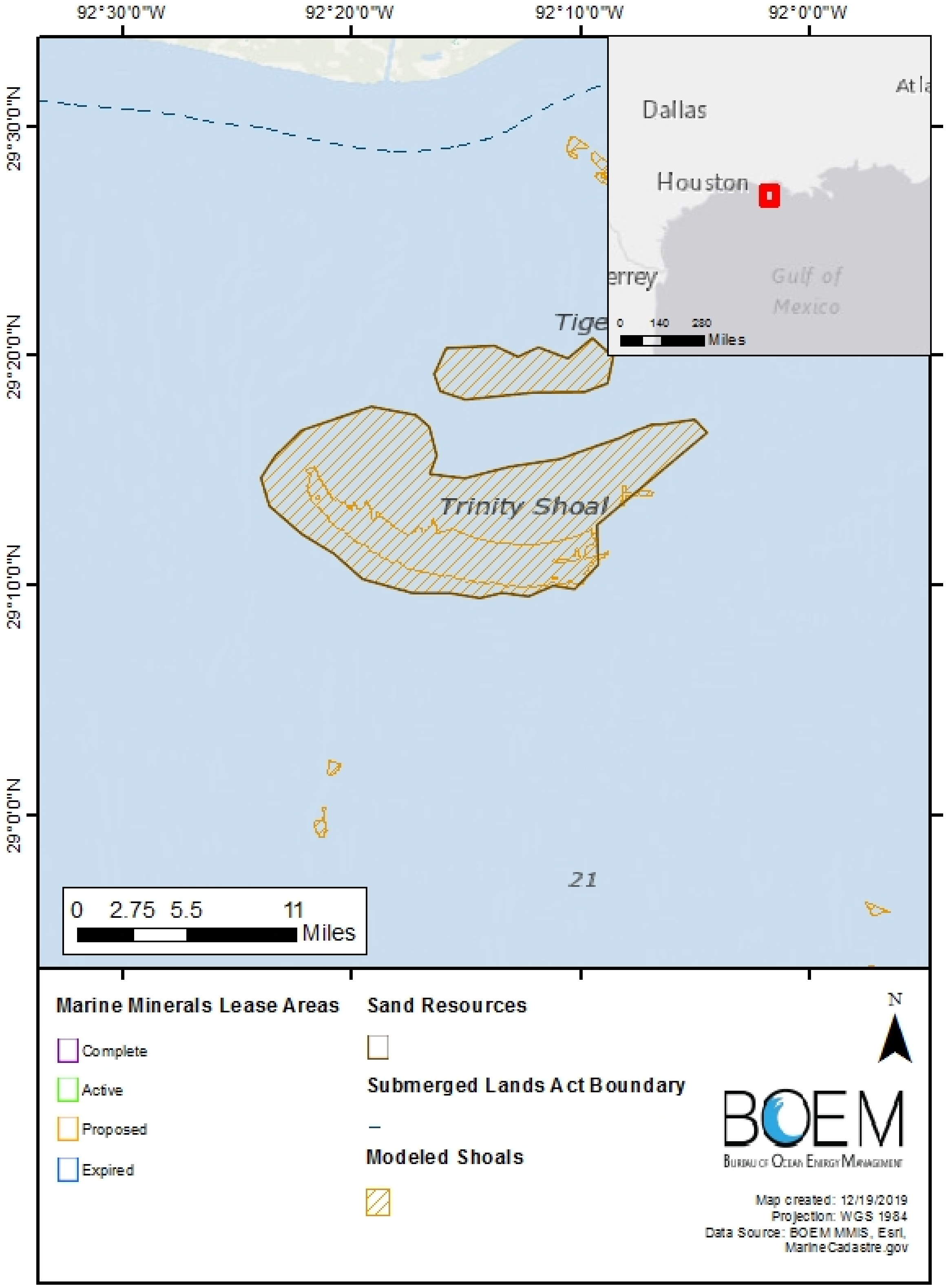
- *Description of why project is proposed/why they need sediment on the beach.*
- *Brief description of past projects, if any. This section is expanded on in Section 3.*
- *Who prepared this assessment and why (1 paragraph)*
- *Description of the physical location of the project and coastal features that it is most adjacent to. This section is expanded on in Section 3.]*

See Maps 1-3 for more information on the proposed borrow area and its surrounding environment including bathymetry, bottom currents, and seafloor substrate.

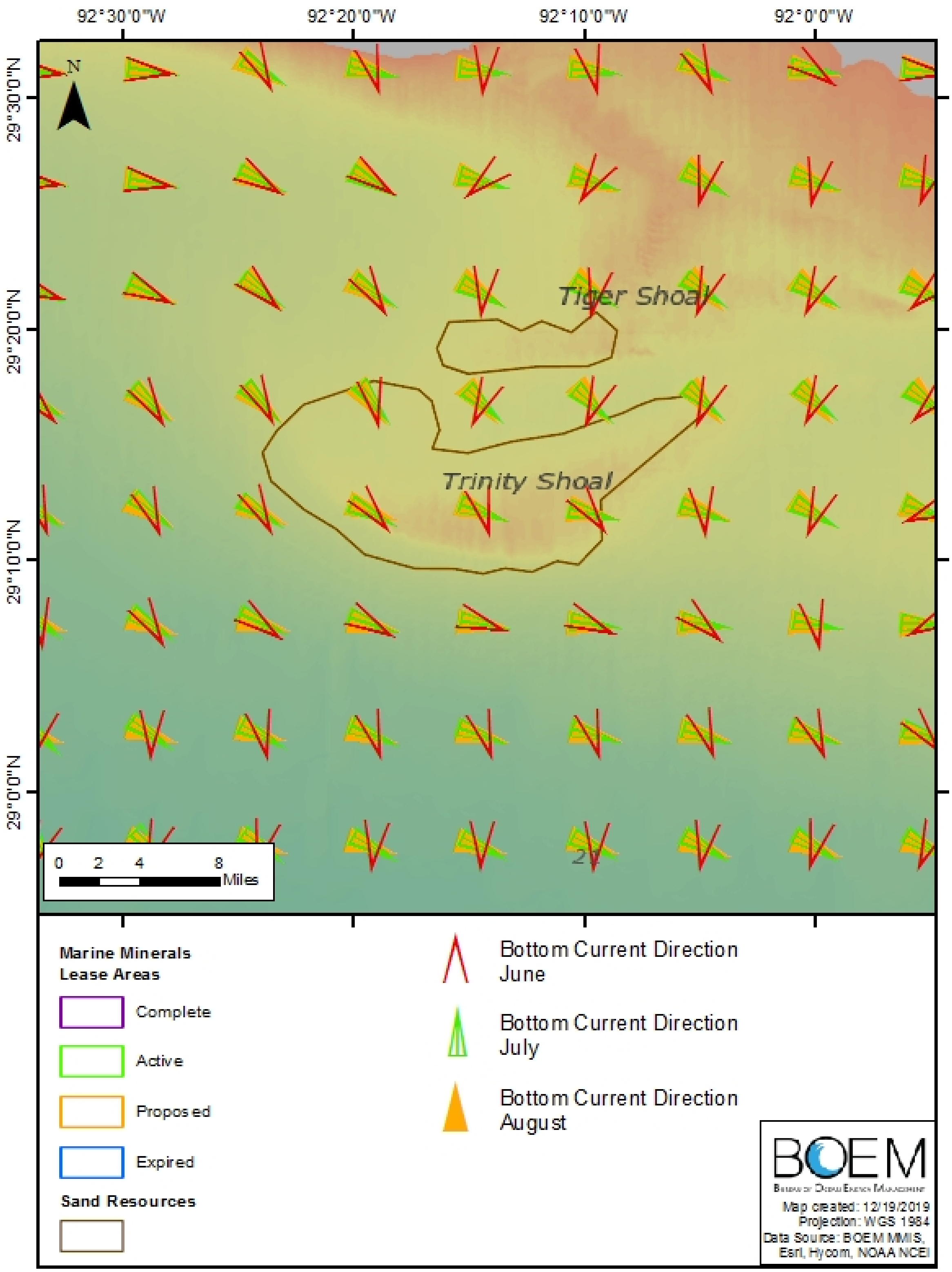
Additional information regarding the proximity of the proposed project to features of interest not covered in this report can be obtained through BOEM and NOAA's Ocean Reporting Tool (NOAA 2018b).

[If Maps 1-3 do not all exist, edit the above reference and the map headers below as applicable.]

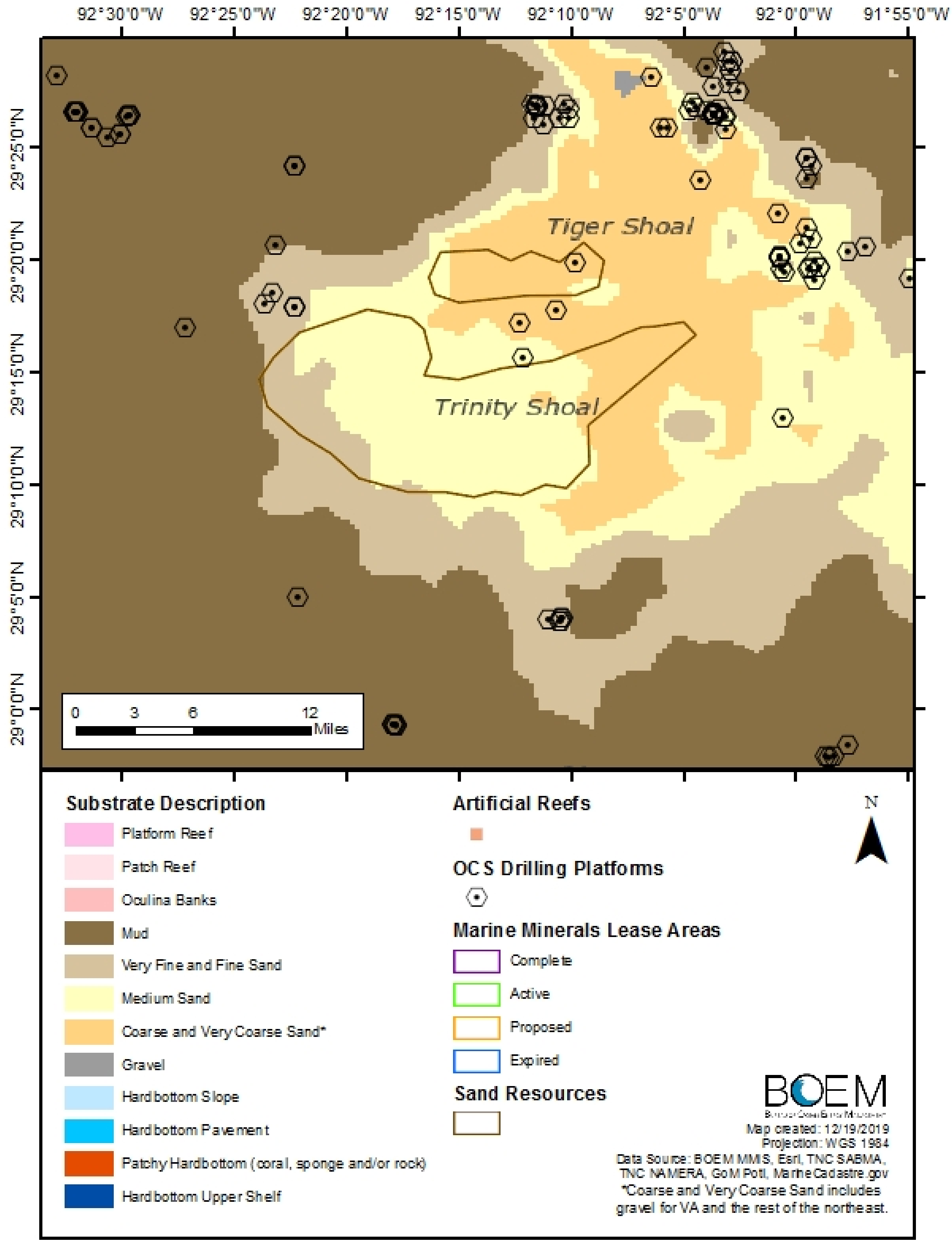
Map 1: Proposed Project Area



Map 2: Bathymetry and Bottom Currents



Map 3: Proposed Borrow Area and Surrounding Benthic Substrate



II. Purpose

Provisions of the MSFCMA (16 USC 1801) require that EFH areas be identified for each species managed under a fishery management plan, and that all Federal agencies consult with the NMFS on all Federal actions that may adversely affect EFH. EFH is defined as "those waters and substrate necessary to fish for spawning, breeding, feeding or growth to maturity." This EFH assessment is being prepared pursuant to Section 305(b)(2) of the MSFCMA and includes the following required parts: 1) identification of species of concern; 2) a description of the proposed action; 3) an analysis of the effects of the proposed action; 4) proposed mitigation; and 5) the Federal agency's views regarding the effects of the proposed action. The purpose of this consultation process is to address specific federal actions that may adversely affect EFH, but do not have the potential to cause substantial adverse impact.

III. Proposed Project

[The following information should be input manually:

- *How many cubic yards (cy) of sediment are to be removed.*
- *Location of removal on the shoal/sediment resource. 'Leeward' vs. 'windward' side; cardinal direction; relation of removal location to other prominent features on the landscape.*
- *What equipment will be used. Type of dredge. Buoys/pump-out stations used? Pipelines for pump outs on shore used? Bulldozers and/or graders used on the shore?*
- *What type of sediment is going to be mined. CMECS description.*
- *When sediment is proposed to be mined. Months or season.*
- *If applicable, Alternative A*
- *If applicable, Alternative B*
- *If applicable, more Alternatives (C-Z)]*

The selected borrow area, which has been allocated for sediment extraction for this project, is in the NGM ecoregion as defined by CMECS. This sediment feature ranges in depth from 3.0m (9.84ft) to approximately 10.0m (32.8ft). It is classified under CMECS as Geoform Coponent (GC) Origin None. The predominant CMECS classification for the material contained within this feature is Substrate Component (SC) Origin Geologic, SC Class Unconsolidated Mineral, SC Subclass Fine Unconsolidated. For additional CMECS variables that define this resource please see Table 1.

Table 1: Classification and values associated with the proposed borrow area (modified from CMECS)

Attribute	Value	Unit	Classification
Magnitude of Bottom Current - June	0.0614551168677	m/s	
Magnitude of Bottom Current - July	0.0683867815434	m/s	
Magnitude of Bottom Current - August	0.0630514614566	m/s	
Rugosity	1.0		
Slope Range	0.0 - 0.5	Degrees	
Substrate Descriptor			unk
Surface Pattern			
Orientation	272.2838978287	Degrees	
Shelf Position			unk
Accretion Status			unk
Bathymetric Position Index (BPI)	1.04		
Temporal Persistence			unk
Disturbance Regime			unk
Dissolved Oxygen Minimum	4.3911190033	mg/L	
Temperature Range	15.6353683472 - 30.4577026367	Degrees C	
Anthropogenic Impact			unk

[More information about the resource, if available. Eg. Description of surrounding area, prevalent underwater features nearby, anthropogenic features nearby, results of video surveys, accretion studies, or other types of studies potentially derived from studies provided in Table 2.] Additional information relevant to this sediment resource may be available from past studies (see Table 2 for further details).

[Any information on species known to use the resource, with focus on species that need the seafloor habitat(s) for one or more life stages. E.g. Number of species (fish, marine mammals, sea turtles).]

Table 2: MMIS Studies overlapping the proposed borrow area

Study ID	Report Link
OffshoreLA_LACOSS1983_4	Geophysics Log; LACOSS IV
GulfofMexico_NMFS-SEFSC-483_N	Compilation of Data Sets Relevant to the Identification of Essential Fish Habitat on the Gulf of Mexico Continental Shelf and for the Estimation of the Effects of Shrimp Trawling Gear on Habitat
TrinityTiger_CoopNo14-12-0001-30387	Assessment of Sand Resources in the Trinity Shoal Area

Previous dredging in this area has occurred 0 times between 10000 and 0. Over that time, 0 cy (0.0 m^3) of material has been removed for beach nourishment projects. See Table 3 for further information, including links to associated BOEM documents. Map 4 shows the amount of accretion that has occurred between the previous two dredge events. The raster was calculated by subtracting the pre-dredge survey from the most recent dredge event from the post-dredge survey of the second most recent event. The result is the change in elevation, in meters, between the two projects which may be used as an indicator of the recovery potential of the sediment resource after construction.

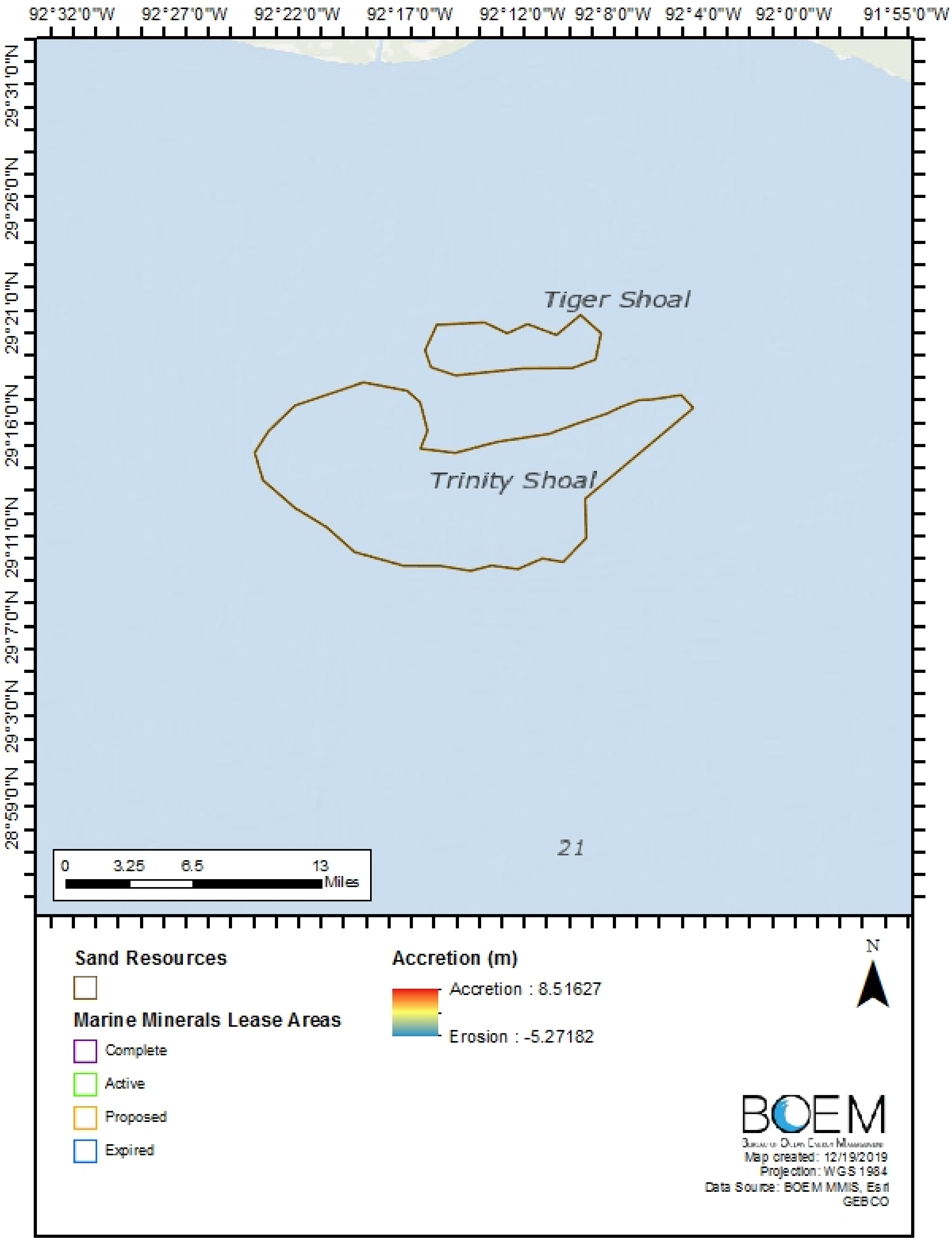
Sediment resources within a 3 km radius of the proposed project cover <area of sand resources in a 3 km radius>. Of that area, <area dredged> have previously been mined. This accounts for <x>% of the sediment resources within a 3 km radius of the proposed burrow area.

[If Map 4 does not exist, edit the text above.]

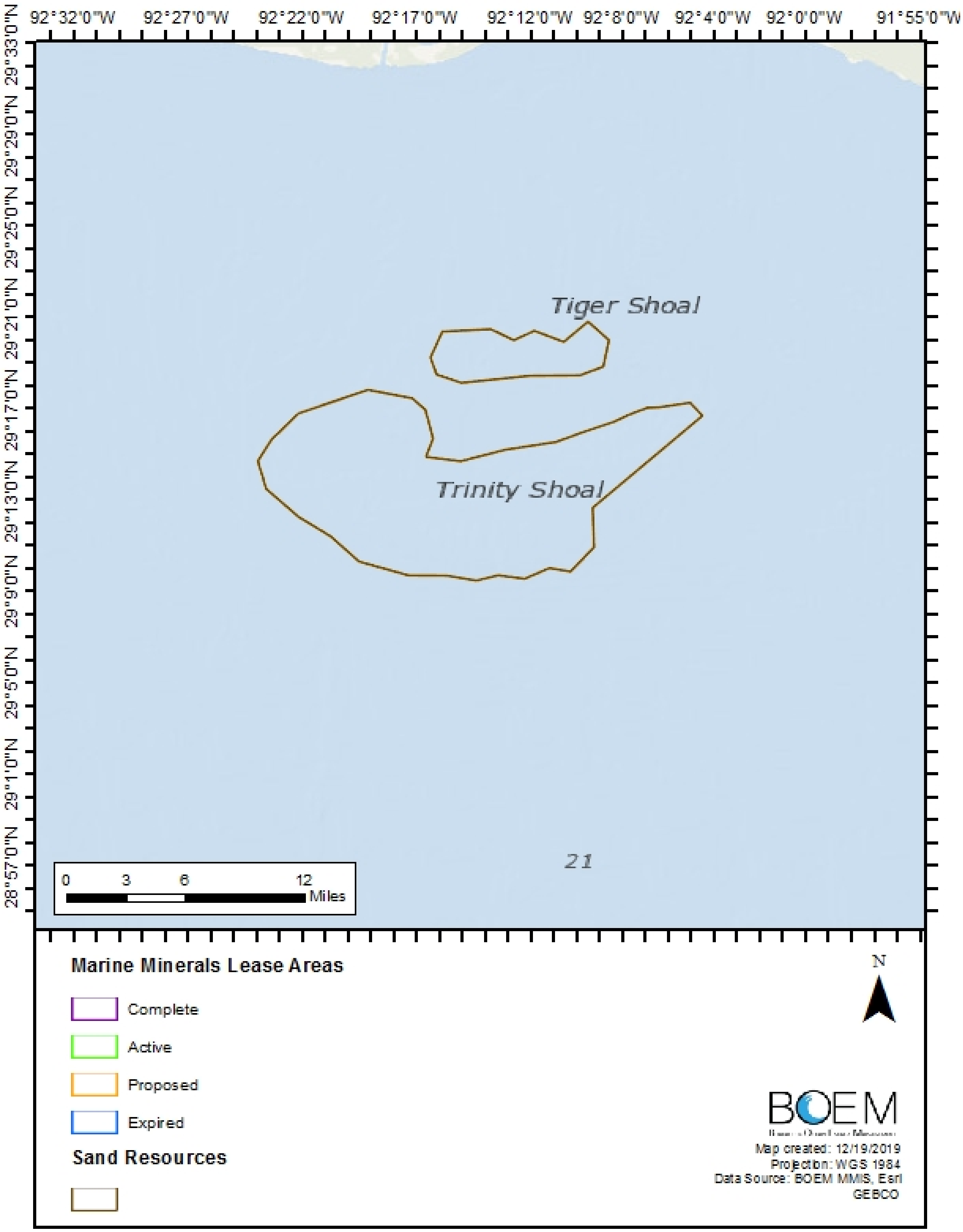
Table 3: Past Dredging Projects- No Results Found

Lease #	Volume allocated (cy)	Volume dredged (cy)	Fiscal Year	BOEM Web Link	EA	FONSI	EIS	ROD	NEPA

Map 4: Sediment Accretion in Proposed Borrow Area, <Fiscal Year - 2>-<Fiscal Year - 1>



Map 5: Past Dredge Events in Proposed Borrow Area



[Delete the Map headings above if not applicable.

Results of On-Site Inspection, if applicable.

Views of recognized experts on the habitat or species that may be affected, if applicable.]

IV. Identification of Managed Species

Table 4: Essential Fish Habitat species and life stages that overlap the proposed borrow area. *Information in this table was gathered from official EFH documentation when available or other well recognized studies of sand affinity (noted in the shoalMATE study report). X's indicate that the proposed area matches the habitat criteria for the species/life stage combination to determine the possibility that a species/life stage with an overlapping EFH polygon may utilize the proposed area. The use of "unk" indicates that the habitat parameter was not defined for that species/lifestage combination in the documentation and is treated as a match to indicate that particular care should be taken in researching the impacts on these species. An "X" in the Water Column Zone field indicates the species is known to be demersal for some portion of that lifestage (as opposed to pelagic). The impact potential is a qualitative assessment based on the combination of results for the four parameters (defined in the shoalMATE study report).*

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Atlantic Sharpnose Shark	Neonate/YOY	All	X	X	X	X	High
Atlantic Sharpnose Shark	Mating/Birthing	Summer	unk	unk	X	unk	High
Atlantic Sharpnose Shark	Juveniles	All	X	X	X	X	High
Atlantic Sharpnose Shark	Adults	All	X	unk	X	X	High
Banded Rudderfish	Eggs	All	unk	unk	X	X	High
Blacktip Shark	Neonate/YOY	All	X	X	X	X	High
Blacktip Shark	Juveniles;Adults	All	X	unk	X	X	High
Bonnethead Shark	Neonate/YOY	Summer	X	unk	X	X	High

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Bonnethead Shark	Juveniles	Summer	X	unk	X	X	High
Bonnethead Shark	Adults	Summer	X	unk	X	X	High
Bull Shark	Neonate/YOY	All	X	X	X	X	High
Bull Shark	Juveniles;Adults	All	X	unk	X	X	High
Gag	Juveniles	Summer	X	X	X	X	High
Gray Snapper	Spawning Adults	Summer	unk	X	X	X	High
Gray Snapper	Juveniles	All	X	X	X	X	High
Gray Snapper	Adults	All	X	X	X	X	High
Gray Triggerfish	Spawning Adults	All	unk	X	X	unk	High
Gray Triggerfish	Juveniles	All	unk	X	X	X	High
Gray Triggerfish	Eggs	Summer	unk	X	X	X	High
Gray Triggerfish	Adults	All	unk	X	X	X	High
Lane Snapper	Larvae	Summer	X	X	X	X	High
Lane Snapper	Juveniles	Summer	X	X	X	X	High
Lane Snapper	Adults	All	X	unk	X	X	High
Red Drum	Spawning Adults	Summer	X	unk	X	unk	High
Red Drum	Larvae	Summer	X	X	X	unk	High
Red Drum	Adults	Summer	X	unk	X	X	High

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Red Grouper	Juveniles	All	X	X	X	X	High
Red Grouper	Adults	All	X	X	X	X	High
Red Snapper	Adults	Summer	X	X	X	X	High
Spinner Shark	Spawning Adults	Summer	unk	unk	X	unk	High
Spinner Shark	Neonate/YOY	All	X	unk	X	unk	High
Spinner Shark	Juveniles	All	X	unk	X	X	High
Spinner Shark	Adults	All	unk	unk	X	X	High
Almaco Jack	Spawning Adults	Summer	unk	unk		unk	Low
Almaco Jack	Juveniles	Summer	X	X		X	Low
Almaco Jack	Eggs	Summer	unk			unk	Low
Almaco Jack	Adults	Summer	unk	X			Low
Black Grouper	Larvae	All	unk			X	Low
Black Grouper	Juveniles	Summer	unk	X		X	Low
Black Grouper	Eggs	All	unk				Low
Black Grouper	Adults	All	X			X	Low
Blackfin Snapper	Spawning Adults	Summer	unk	X			Low
Blackfin Snapper	Larvae	All	unk	unk			Low
Blackfin Snapper	Juveniles	All	unk	X		X	Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Blackfin Snapper	Eggs	Summer	unk				Low
Blackfin Snapper	Adults	Summer	unk	X	X		Low
Blueline Tilefish	Spawning Adults	Summer		unk			Low
Blueline Tilefish	Larvae	All	unk				Low
Blueline Tilefish	Juveniles	All	unk	unk			Low
Blueline Tilefish	Eggs	All	unk				Low
Blueline Tilefish	Adults	All	X	X			Low
Brown Shrimp	Sub-adults	Summer		X		X	Low
Brown Shrimp	Spawning Adults	Summer	unk	X			Low
Brown Shrimp	Late Postlarvae;Juveniles	Summer	X	X			Low
Brown Shrimp	Larvae;Pre-settlement Postlarvae	Summer	X			X	Low
Brown Shrimp	Adults	Summer	X	X			Low
Cobia	Spawning Adults	Summer		unk	X	unk	Low
Cobia	Juveniles	Summer			X	X	Low
Cobia	Adults	Summer			X	X	Low
Cubera Snapper	Spawning Adults	Summer	X	X		X	Low
Cubera Snapper	Larvae	All	unk	unk		X	Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Cubera Snapper	Juveniles	All	X	X		X	Low
Cubera Snapper	Eggs	Summer	unk			X	Low
Cubera Snapper	Adults	All	unk	unk		X	Low
Gag	Adults	Summer		X	X		Low
Goldface Tilefish	Adults	All	unk	unk		unk	Low
Goliath Grouper	Spawning Adults	Summer	unk	X			Low
Goliath Grouper	Larvae	Summer	unk				Low
Goliath Grouper	Juveniles	Summer	unk	X		X	Low
Goliath Grouper	Eggs	Summer	unk				Low
Goliath Grouper	Adults	All	X	X		X	Low
Gray Snapper	Larvae	Summer		X	X	X	Low
Greater Amberjack	Larvae	Summer	unk			unk	Low
Greater Amberjack	Juveniles	Summer	unk	X		unk	Low
Greater Amberjack	Adults	Summer	unk	X		X	Low
Hogfish	Spawning Adults	Summer	unk	X		X	Low
Hogfish	Larvae	All	unk			unk	Low
Hogfish	Eggs	Summer	unk			unk	Low
Hogfish	Adults	Summer	X	X		X	Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
King Mackerel	Spawning Adults	Summer	X	unk	X		Low
King Mackerel	Larvae	Summer	X		X		Low
King Mackerel	Eggs	Summer	unk		X		Low
Lane Snapper	Spawning Adults	Summer	unk	X	X		Low
Lesser Amberjack	Juveniles	Summer	unk	X			Low
Lesser Amberjack	Adults	Summer	unk	X			Low
Mutton Snapper	Spawning Adults	Summer	unk	X			Low
Mutton Snapper	Larvae	Summer	unk			unk	Low
Mutton Snapper	Juveniles	Summer	unk	X		unk	Low
Mutton Snapper	Eggs	Summer	unk			unk	Low
Mutton Snapper	Adults	Summer	unk	X		unk	Low
Pink Shrimp	Sub-adults	Summer	X	X		X	Low
Pink Shrimp	Spawning Adults	Summer	X	X		X	Low
Pink Shrimp	Late Postlarvae;Juveniles	Summer	X	X		X	Low
Pink Shrimp	Larvae;Pre-settlement Postlarvae	Summer	X			X	Low
Pink Shrimp	Fertilized Eggs	Summer	X	unk		X	Low
Pink Shrimp	Adults	Summer	X	X		X	Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Queen Snapper	Spawning Adults	Summer	unk	unk			Low
Queen Snapper	Juveniles	All	unk				Low
Queen Snapper	Eggs	All	unk				Low
Queen Snapper	Adults	All	X	X			Low
Red Grouper	Spawning Adults	Summer		X	X		Low
Red Grouper	Larvae	Summer			X		Low
Red Snapper	Spawning Adults	Summer		X	X		Low
Red Snapper	Larvae	Summer	X		X		Low
Red Snapper	Juveniles	Summer	X	X	X		Low
Red Snapper	Eggs	All	unk		X		Low
Royal Red Shrimp	Spawning Adults	Summer	unk	unk			Low
Royal Red Shrimp	Larvae	All	unk	unk			Low
Royal Red Shrimp	Juveniles	All	unk	unk			Low
Royal Red Shrimp	Eggs	Summer		unk			Low
Royal Red Shrimp	Adults	Summer		X			Low
Scamp	Spawning Adults	Summer	X	X			Low
Scamp	Juveniles	All	unk	X			Low
Scamp	Adults	All	X	X			Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Silk Snapper	Spawning Adults	Summer	unk	unk			Low
Silk Snapper	Larvae	Summer	unk	unk			Low
Silk Snapper	Juveniles	Summer	unk	unk			Low
Silk Snapper	Eggs	Summer	unk	unk			Low
Silk Snapper	Adults	Summer		unk			Low
Snowy Grouper	Spawning Adults	Summer	unk	X			Low
Snowy Grouper	Larvae	Summer	unk				Low
Snowy Grouper	Juveniles	All	X	X		X	Low
Snowy Grouper	Eggs	All	unk				Low
Snowy Grouper	Adults	All	X	X			Low
Speckled Hind	Spawning Adults	Summer	unk	X			Low
Speckled Hind	Larvae	All	unk				Low
Speckled Hind	Juveniles	All	unk	unk			Low
Speckled Hind	Eggs	All	unk				Low
Speckled Hind	Adults	All	unk	X			Low
Tilefish	Spawning Adults	Summer	unk		X		Low
Tilefish	Spawning Adults	Summer	unk	unk	X		Low
Tilefish	Larvae	Summer	unk		X		Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Tilefish	Juveniles	All	unk	X	X		Low
Tilefish	Eggs	Summer	unk		X		Low
Tilefish	Adults	All		X	X		Low
Vermilion Snapper	Spawning Adults	Summer	unk	unk		unk	Low
Vermilion Snapper	Larvae	Summer	unk				Low
Vermilion Snapper	Juveniles	All	unk	X			Low
Vermilion Snapper	Eggs	All	unk				Low
Vermilion Snapper	Adults	Summer		X			Low
Warsaw Grouper	Spawning Adults	Summer	unk	X			Low
Warsaw Grouper	Larvae	All	unk				Low
Warsaw Grouper	Juveniles	All	unk	X			Low
Warsaw Grouper	Eggs	All	unk				Low
Warsaw Grouper	Adults	All	X	X			Low
Wenchman	Spawning Adults	Summer	unk	X			Low
Wenchman	Larvae	Summer	unk				Low
Wenchman	Juveniles	All	unk	unk			Low
Wenchman	Eggs	Summer	unk				Low
Wenchman	Adults	Summer		X			Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
White Shrimp	Sub-adults	Summer	X	X		X	Low
White Shrimp	Spawning Adults	Summer	unk	unk		X	Low
White Shrimp	Late Postlarvae;Juveniles	Summer	X	X			Low
White Shrimp	Larvae;Pre-settlement Postlarvae	Summer		unk		X	Low
White Shrimp	Fertilized Eggs	Summer	unk	unk		X	Low
White Shrimp	Adults	Summer	X	X		X	Low
Yellowedge Grouper	Spawning Adults	Summer	unk	X			Low
Yellowedge Grouper	Larvae	Summer	unk				Low
Yellowedge Grouper	Juveniles	All	unk	X		X	Low
Yellowedge Grouper	Eggs	All	unk				Low
Yellowedge Grouper	Adults	All	X	X			Low
Yellowfin Grouper	Spawning Adults	Summer	unk	X			Low
Yellowfin Grouper	Larvae	All	unk	unk			Low
Yellowfin Grouper	Juveniles	All	unk	X		X	Low
Yellowfin Grouper	Eggs	All	unk	unk			Low
Yellowfin Grouper	Adults	All	X	X		X	Low
Yellowmouth Grouper	Spawning Adults	All	unk	unk			Low

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
Yellowmouth Grouper	Larvae	All	unk				Low
Yellowmouth Grouper	Juveniles	All	unk	unk			Low
Yellowmouth Grouper	Eggs	All	unk				Low
Yellowmouth Grouper	Adults	All	X	X			Low
Yellowtail Snapper	Spawning Adults	Summer	unk	unk		unk	Low
Yellowtail Snapper	Larvae	All	unk			X	Low
Yellowtail Snapper	Juveniles	All	X	X		X	Low
Yellowtail Snapper	Eggs	Summer	unk			X	Low
Yellowtail Snapper	Adults	All	X	X		X	Low
Banded Rudderfish	Spawning Adults	Summer	unk		X	X	Medium
Banded Rudderfish	Larvae	Summer	unk		X	X	Medium
Banded Rudderfish	Juveniles	Summer	unk		X	X	Medium
Banded Rudderfish	Adults	Summer	unk		X	X	Medium
Cobia	Larvae	Summer	X		X	X	Medium
Cobia	Eggs	Summer	X		X	unk	Medium
Gray Snapper	Eggs	Summer	unk		X	X	Medium
Gray Triggerfish	Larvae	All	unk		X	unk	Medium
King Mackerel	Juveniles	Summer	unk		X	X	Medium

	Life Stage	Season	Temp	Water Column Zone	Sand Affinity	Depth Range	Impact Potential
King Mackerel	Adults	All	X		X	X	Medium
Lane Snapper	Eggs	Summer	unk		X	X	Medium
Red Drum	Eggs	Summer	X		X	unk	Medium
Spanish Mackerel	Spawning Adults	Summer	X		X	X	Medium
Spanish Mackerel	Larvae	Summer	X		X	X	Medium
Spanish Mackerel	Juveniles	Summer	X		X	X	Medium
Spanish Mackerel	Eggs	Summer	unk		X	X	Medium
Spanish Mackerel	Adults	All	X		X	X	Medium

V. Evaluation of Impacts on EFH Species

Fish species' presence within waters of the project impact area is highly variable, both spatially and temporally. Presence can vary for highly migratory species, among life stages, and seasonally.

The short-term impacts of dredging on fish include entrainment, physiological or behavioral changes due to human-made sounds, loss of prey/food web effects, loss of bottom substrate, and effects due to suspended and resuspended sediment plumes, sedimentation of the seafloor, and the potential release of contaminants (Kim et al. 2008; Suedel et al. 2008; Wenger et al. 2017). Hopper and cutterhead dredges use hydraulic suction fields to obtain and transport unconsolidated sediments from aquatic ecosystems. These actions may result in the *entrainment* of fish and shellfish, as defined as the direct uptake of organisms due to the hydraulic suction field generated by a draghead or cutterhead dredge (Reine et al. 1998).

Sounds from dredging operations are produced from vessels in transit to/from the dredging location, supporting vessels, and the dredging operation itself (see Reine et al. 2014a; Reine et al. 2014b; Robinson et al. 2012; Pickens and Taylor 2020). Underwater sounds emitted from dredging operations are of the amplitude to affect the behavior of fish at a considerable distance from the dredge operation (~400-1,200 m). However, the maximum sound levels emitted by dredge activities are restricted to approximately 0-300 m from the source of the vessel. These sounds are not at a level that would result in mortality or severe injury. At the closest proximities, effects may include permanent or temporary hearing impairment. 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. 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 (Pickens and Taylor 2020).

Regarding suspended sediments, the rotation of the cutterhead itself (for cutterhead dredges) produces substantial sediment resuspension in the lower part of the water column; plume concentrations at the surface of the water column may be half of the concentration at the bottom (Havis 1988). Overflow from hopper dredges can be extremely turbid in close proximity to the dredge, as fine-grained TSS may reach >750 mg/L (Havis 1988). Additionally, undesirable fine sediments may be discarded in the sorting and screening process (Michel et al. 2013; Sutton et al. 2009). Havis (1988) compared trailing suction hopper dredges (THSD) and cutterhead dredges, and showed TSS concentrations were much greater for TSHD (with overflowl allowed), particularly at greater depths. Potential responses of fish to SS 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). Fish eggs and larvae are particularly susceptible to sedimentation and SS; this may be because of their lack of mobility, relatively high oxygen demand, and/or anatomy (Appleby and Scarratt 1989; Wilber and Clarke 2001). 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). Negative impacts to fish habitat may also include sedimentation of hard bottom or damage/mortality of corals from sedimentation or SS (Erftemeijer et al. 2012; Linderman and Snyder 1999; Pickens and Taylor 2020).

Long-term impacts to fish from offshore dredging operations include loss of physical habitat and suspended/resuspended sediment plumes. Although most studies measure turbidity over hours to a few days following dredging, Fisher (2015) showed turbidity fluxes over 1 ½ years after dredging; turbidity fluxes were not observed >2 km from the initial dredge site. Overall, the pattern has emerged that extremely high turbidity occurs for a relatively short duration (10-15 minutes) during and immediately following dredging. The area most affected by high TSS and sedimentation is generally 300-600 m from the dredge site, but some effects are expected to 3 km. Under certain oceanographic conditions, sediments plumes may extend up to 20 km from the dredge site. Recommendations for best practices for dredging near corals, and coral reefs, are further provided by PIANC (2010). All species listed in Table 4 may have long-term impacts due to dredging operations.

Some species/life stages classified as 'low' in the 'Impact Potential' column in Table 4 may lack a depth of information regarding the environmental conditions at which they have been observed and/or they lack information on their temporal presence within the proposed borrow area as specified in Fisheries Management Plans. Further review of the existing body of scientific literature may reveal information which can be used to fill in these knowledge gaps. Another important note regarding this report is that distribution and/or abundance information specifically for important forage species for EFH species was not considered but may exist as part of species models or as part of the data that was used in the creation of EFH GIS shapes.

1 EFH Species with High Potential for Impacts

The species listed below are those with some combination of variables that indicate potential use of the proposed borrow area from Table 4. Details for each species include links to official EFH descriptions and relevant background information. Some species are lumped into groups for EFH purposes and therefore will have identical EFH descriptions.

1.1 Atlantic Sharpnose Shark

https://www.fisheries.noaa.gov/webdam/download/69616917

Neonate/YOY

https://www.fisheries.noaa.gov/webdam/download/69616917

Mating/Birthing

https://www.fisheries.noaa.gov/webdam/download/69616917

Juveniles

https://www.fisheries.noaa.gov/webdam/download/69616917

Adults

https://www.fisheries.noaa.gov/webdam/download/69616917

1.1.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.2 Banded Rudderfish

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Eggs

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.2.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.3 Blacktip Shark

https://www.fisheries.noaa.gov/webdam/download/69616917

Neonate/YOY

https://www.fisheries.noaa.gov/webdam/download/69616917

Juveniles;Adults

https://www.fisheries.noaa.gov/webdam/download/69616917

1.3.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.4 Bonnethead Shark

https://www.fisheries.noaa.gov/webdam/download/69616917

Neonate/YOY

https://www.fisheries.noaa.gov/webdam/download/69616917

Juveniles

https://www.fisheries.noaa.gov/webdam/download/69616917

Adults
https://www.fisheries.noaa.gov/webdam/download/69616917

1.4.1 [Potential Project Impacts]
[Insert further applicable information manually if available. Delete if this section is empty.]

1.5 Bull Shark
https://www.fisheries.noaa.gov/webdam/download/69616917

Neonate/YOY
https://www.fisheries.noaa.gov/webdam/download/69616917

Juveniles;Adults
https://www.fisheries.noaa.gov/webdam/download/69616917

1.5.1 [Potential Project Impacts]
[Insert further applicable information manually if available. Delete if this section is empty.]

1.6 Gag
http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Juveniles
http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.6.1 [Potential Project Impacts]
[Insert further applicable information manually if available. Delete if this section is empty.]

1.7 Gray Snapper
http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Spawning Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Juveniles

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.7.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.8 Gray Triggerfish

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Spawning Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Juveniles

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Eggs

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.8.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.9 Lane Snapper

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Larvae

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Juveniles

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.9.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.10 Red Drum

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Spawning Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Larvae

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.10.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.11 Red Grouper

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Juveniles

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.11.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.12 Red Snapper

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

Adults

http://gulfcouncil.org/wp-content/uploads/EFH-5-Year-Revew-plus-App-A-and-B_Final_12-2016.pdf

1.12.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

1.13 Spinner Shark

https://www.fisheries.noaa.gov/webdam/download/69616917

Spawning Adults

https://www.fisheries.noaa.gov/webdam/download/69616917

Neonate/YOY

https://www.fisheries.noaa.gov/webdam/download/69616917

Juveniles

https://www.fisheries.noaa.gov/webdam/download/69616917

Adults

https://www.fisheries.noaa.gov/webdam/download/69616917

1.13.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

2 EFH Species with Medium Potential for Impacts

The species listed in Table 4 with a value of Medium in the 'Impact Potential' column have EFH GIS shapes which spatially overlap the project boundaries, have an observed affinity for sand/sediment resources (Rutecki, et al. 2014), and have observed depth, temporal, and temperature ranges which also overlap the project area. However, these species and life stages are observed to be within the water column, somewhere between a few feet above the seafloor and the surface. Due to their presence in the water column instead of bottom habitats, these species and life stages may experience fewer dredge-related impacts than demersal species.

1.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

3 EFH Species with Low Potential for Impacts

The species and life stages listed in Table 4 with a value of Low in the 'Impact Potential' column have EFH GIS shapes which spatially overlap the project boundaries, however, data from fishery management plans and scientific research (Rutecki, et al. 2014) indicate that it is unlikely that those species and life stages will be found within the project area. This determination was made due to one or more of the following factors: they have not been observed to have affinity for using sand/sediment resources (Rutecki, et al. 2014), they have not been observed within the depth range of the project, they have not been observed within the project area during the season and/or month of the project, or they have not been observed within the anticipated water temperature range of the project. Because these important characteristics do not overlap, these species have the lowest potential of those categorized to be impacted during dredging.

Another group of species with a value of Low in the 'Impact Potential' column of Table 4 are those that are lacking information in fishery management plan documentation with regards to observed depth ranges, seasonality, temperature ranges, or whether the species or life stage is found in the water column or on, near, or within the seafloor substrate. A review of the existing body of scientific literature may reveal more data than what exists in the fishery management plans reviewed in preparation of this document.

1.1 [Potential Project Impacts]

[Insert further applicable information manually if available. Delete if this section is empty.]

4 Predicted Relative Abundance or Probability of Presence of Selected Species

Species distribution models are state-of-the-art statistical models that predict the distribution of species based on species-habitat relationships (Guisan and Zimmermann 2000; Robinson et al. 2011). Distribution models were developed based on fishery-independent survey data from 2003-2017 combined with remote sensing data on oceanographic conditions, substrate, geography, and the surrounding ecosystems of wetlands and estuaries (see Pickens and Taylor 2020 for details). Prey species' distributions were also included as predictor variables. Predictive models of shrimp and fish were assessed with independent validation data, and all models presented explained 30-45% of the deviance (equivalent to an r^2 for count data) in validation data. Species distribution models predicting the probability of presence were >80% accurate as measured by Area Under the Curve (AUC) statistics; this shows very good predictive ability (Manel et al. 2001). Data and results for shrimp and snapper species represent summer and fall seasons, while shark distribution models represent spring, summer, and fall seasons. We selected species to model based on their socio-economic value, use of shoals, data availability, representation of key trophic levels (e.g., prey and apex predators) and guilds (e.g., demersal, juveniles of species that use hard bottom habitats as adults). Species modeled include: brown shrimp (adults), pink shrimp (adults), white shrimp (adults), lane snapper (age-0 and 1), red snapper (age-0 and 1), Atlantic sharpnose shark (juveniles and adults), blacktip shark (juveniles and adults), and spinner shark (juveniles and adults). The relative abundance or probability of presence for the selected species that overlap with this area are listed in Tables 5a - 5b and further indicate the relative importance of the proposed area to the species compared to surrounding habitats. Appendix A lists the most influential variables that went into each model. Models represent the relative abundance or probability of presence for species' life stages sampled in federal waters, and do not extend to state waters or estuaries.

Brown shrimp (***Penaeus aztecus***)**, pink shrimp (*Penaeus duorarum*), and white shrimp (*Penaeus setiferus*):**

All three shrimp species are demersal, depend on estuaries in early life stages, and inhabit offshore waters as adults. Brown-, pink-, and white shrimp have high economic value in the Gulf of Mexico. In 2016, commercial fisheries landed were brown shrimp ($157 million), white shrimp ($206 million), and pink shrimp ($24.4 million) (NOAA NMFS Office of Science and Technology 2019). In particular, brown shrimp have been documented as an integral part of the Gulf of Mexico food web, as they are described as prey of small pelagic fish, small demersal fish, flatfish, king mackerel, Spanish mackerel, benthic feeding sharks, several snapper and grouper species, black drum, red drum, and others (Tarnecki et al. 2016).

Red snapper (*Lutjanus campechanus*):

Red snapper use shoals and sand/mud substrates as demersal juveniles before inhabiting natural and artificial reefs as adults (Gallaway et al. 2009). Red snapper are particularly important to commercial and recreational fisheries; in 2016, commercial landings totaled $26.5 million for the Gulf of Mexico (NOAA NMFS Office of Science and Technology 2019).

Lane snapper (*Lutjanus synagris*):

Lane snapper is a subtropical, reef-associated species that have a demersal juvenile stage well-documented to inhabit shallow waters with sand/mud bottoms, including shoal habitats (Mikulas and Rooker 2008; Wells et al. 2009). Commercial landings of lane snapper are modest and occur primarily in Florida where landings totaled $86,219 in 2016 (NOAA NMFS Office of Science and Technology 2019). They are a regular recreational catch offshore of Florida as well.

Blacktip shark (*Carcharhinus limbatus*):

Blacktip shark is a large coastal shark species that primarily preys on teleost fishes (Cortés 1999) and is listed as ‘vulnerable’ globally by the IUCN (Burgess and Branstetter 2000). Blacktip sharks are one of most valuable sharks to commercial fisheries of the Atlantic Ocean (Castro 1996). In the western Gulf of Mexico (west of -88º longitude), 207 metric tons of blacktip shark were harvested in 2017, whereas 32 metric tons were harvested in the eastern Gulf of Mexico. In the Gulf of Mexico, juvenile blacktip shark feed mostly on Clupeidae, particularly Gulf menhaden *Brevoortia patronus*, followed by croaker (Bethea et al. 2004).

Atlantic sharpnose shark (*Rhizoprionodon terraenovae*):

Atlantic sharpnose shark are relatively small, demersal sharks that feed on crustaceans and teleost fishes (Cortés 1999). In the Gulf of Mexico, Atlantic sharpnose are regularly caught in recreational and commercial fisheries.

Spinner shark (*Carcharhinus brevipinna*):

Spinner shark is listed as 'near threatened' globally by the IUCN (Burgess 2009) and is a common target by commercial fisheries. Spinner shark primarily prey on teleost fishes (Cortés 1999), particularly Gulf menhaden *Brevoortia patronus*, which are an important prey species in the Gulf of Mexico (Geers et al. 2016).

Table 5a: Predicted relative abundance of selected EFH species in the project area and the surrounding marine environment. All items are all mean values. According to their lengths, the vast majority of sharks were juvenile and adults. There were a small fraction of young-of-the-year (0.002%-2% of individuals). Modeled shrimp species are also predominantly adults.

Species	Age group(s)	Season	Unit	Within Shoal/ Borrow Area	Within 20km	Within Species' Geographic Range Within the Region
Atlantic sharpnose shark	All detected in surveys	Spring;Summer;Fall	sharks/100 hooks/hour	11.96	11.07	8.29
Brown shrimp	All detected in surveys	Summer	individuals/km of trawl	77.96	50.51	77.42
Red snapper	Year 0	Summer	individuals/km of trawl	0.45	0.21	0.32
White shrimp	All detected in surveys	Summer	individuals/km of trawl	11.31	9.02	2.46

Table 5b: Probability of presence for selected EFH species in the project area and the surrounding marine environment. All items reported are mean values.

Species	Age group(s)	Season	Within Shoal/ Borrow Area	Within 20km	Within Species' Geographic Range within the Region
Lane snapper	Year 1	All	0.02	0.01	0.31
Red snapper	Year 1	All	0.03	0.07	0.16
Spinner shark	All detected in surveys	Spring;Summer;Fall	0.42	0.73	0.13
Blacktip shark	All detected in surveys	Spring;Summer;Fall	0.86	0.85	0.38
Lane snapper	Year 0	Summer	0.07	0.05	0.23
Pink shrimp	All detected in surveys	Summer;Fall	0.07	0.14	0.51

5 Habitat Areas of Particular Concern (HAPC)

Habitat Areas of Potential Concern (HAPC) are subsets of EFH that have been identified for special consideration during planning due to the rarity of the environment, stressors from development, importance to federally managed species, or vulnerability to anthropogenic degradation (BOEM; NOAA 2018a). HAPCs that overlap the proposed area are listed in Table 6 and have been considered within this assessment.

Table 6: List of HAPCs that overlap the project area.

No Results Found

Site Name	Link

6 Forage Species for EFH Species

Certain forage species may be important indicators for the presence of EFH species However, these forage species themselves may not be listed as EFH. *[If, after a manual review of source literature below and any additional literature, information is revealed which may be important to this assessment, fill in here. For example, are there a list of important forage species for an EFH species that ShoalMATE has indicated as having high impact potential for this project? Or, are there non-EFH listed forage species that are known to be in the area of the project and are important for a variety of EFH species?]*

For further information on forage species for EFH, see Tarnecki et al. 2016.*[Add any additional sources identified during manual review]*.

7 Areas Closed to Fishing

[The author of this report should also evaluate the are of interest for possible fishery closures. Fisheries may be closed by season, by gear type, by species, or by other metrics. Federal, state, and local closures should be included here.]

VI. Proposed Mitigation

Measures to minimize or avoid effects on EFH and managed species will be implemented based on consultation with federal agencies. Overarching measures to mitigate impacts are as follows: 1) implementation of best management and engineering practices, 2) completion of hydrographic surveys pre- and post- dredging; and 3) coordination with the NMFS to create a management plan to guide future replenishment so that mining of the sediment resource remains sustainable.

[*Additional mitigations specific to this project not available in the tool.*]

VII. Conclusion and Agency Review

The severity of the impact to EFH and supported species is dictated by: 1) the spatial extent of the impact and 2) the chronic or long-term nature of the impact. A review of international literature has shown heightened levels of turbidity regularly occur within 3 km (or 1.86 miles) of dredging sites; turbidity, as a direct result of dredging, often settles within minutes to hours, but long-term monitoring of dredge sites has also shown resuspension of sediments occurs up to 1 ½ years after the dredging event (Pickens and Taylor 2020). Mortality of fish from turbidity is unlikely, but avoidance of the area by fish is a strong possibility (Pickens and Taylor 2020). Underwater sounds and fish entrainment are more local effects that occur over short time periods during the dredging event itself.

The areas that have been designated as EFH in the project area have been given this classification because they are believed to be "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" (16 U. S. C. 1802). HAPC, a separate designation within EFH, is based on one or more of the following considerations: 1) the importance of the ecological function, 2) extent to which the habitat is sensitive to human-induced degradation, 3) whether and to what extent development activities are stressing the habitat type, or 4) rarity of habitat type [50 CFR 600.815(a)(8)].

As discussed and evaluated in this assessment, offshore dredging, dredge transit, and placement along the shoreline are not expected to impact "those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity" to any appreciable extent over a significantly large area or over any significant period of time. Impacts would be limited and short-term. From a finfish perspective, demersal species, early life stages (i.e., eggs, larvae), dormant life stages, spawning individuals, and habitats that are important for species' migration are predicted to most impacted (Pickens and Taylor 2020). Other pelagic species and life stages are predicted to be minimally impacted. Given the relatively small-size of the impacted area relative to the large geographic ranges of transitory fishes, the proposed activities are likely to have only minor impacts on the populations of finfish evaluated in this analysis.

Accordingly, it has been determined that the proposed project may have adverse effects on EFH for Federally managed species, but adverse effects on EFH species, due to construction, will largely be temporary and localized within the dredged footprints and beach nourishment areas in the surf zone. In conclusion, the project is not anticipated to significantly impact EFH species or habitat that may be in the project area.

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[*Additional information to be filled in manually based on the items cited in manually updated sections above.*]

<Additional references from the tool output's table of cited sources>

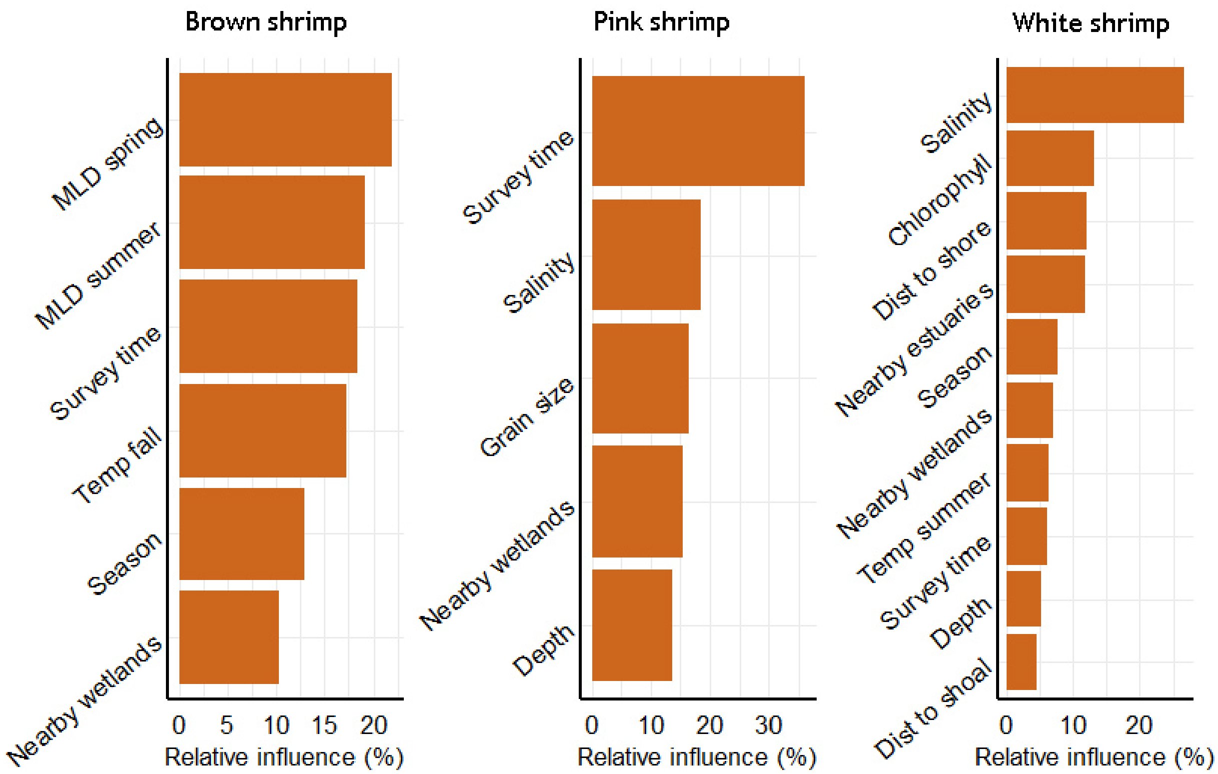
<Possible additional references input here via linking to Endnote document>

Appendix A: Variables in Predictive Models

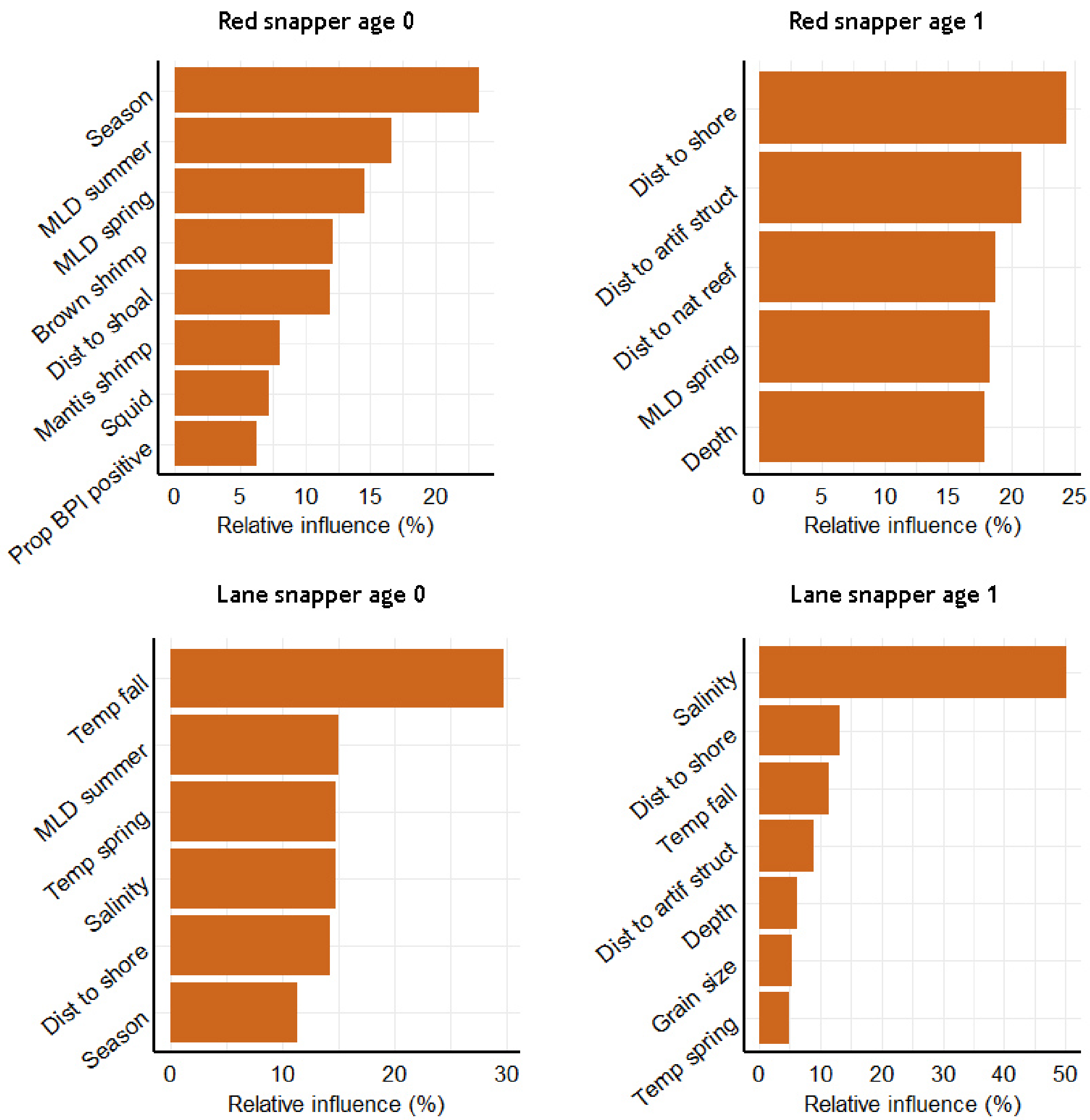
Variable	Description
Bott temp	Bottom temperature
Brown shrimp	Prey species distribution modeled
Chlorophyll	Concentration of chlorophyll in surface waters (mg m^{-3})
Chlorophyll sum	Concentration of chlorophyll in the surface waters during summer (mg m^{-3})
Croaker	(*Micropogonias undulatus*) prey species distribution
Current-U sum	Velocity of west to east currents
Dist to shoal	Distance to shoal (km)
Dist to shore/shoreline	Distance to shoreline (km)
Dist to artif reef	Distance to artificial reef (km) (includes oil platforms/other artificial reefs)
Dist to Gulf Stream	Distance to shoreline
Dist to reef	Distance to natural reef
DOY	Day of year
Nearby estuaries	km^2 of estuarine waters within 160 km of location
Grain size	Sediment grain size (mm)
Hypoxia	Probability of hypoxia
Mantis shrimp	(species in Order Stomatopoda) prey species distribution
MLD	Mixed layer depth (m)
Pink shrimp	(*Pandalus borealis*) prey species distribution modeled
Salinity	Minimum annual bottom salinity
Season	Summer or Fall
Squid	Prey species distribution
Survey time	Time that the survey was conducted (00:00)
Temperature/Temp	Bottom water temperatures (°C)
Wetlands/Nearby wetlands	km^2 of estuarine wetlands within 160 km of location

GULF OF MEXICO MODELS

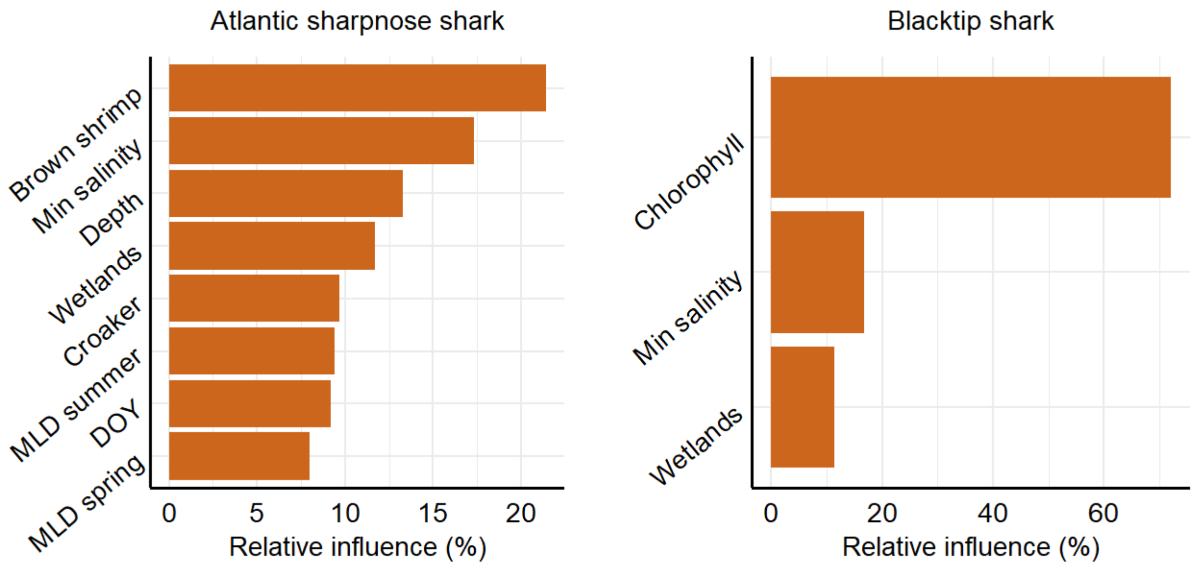
Shrimp Model Variables



Snapper Model Variables



Shark Model Variables



Department of the Interior (DOI)

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 Environmental Studies Program is to provide the information needed to predict, assess, and manage impacts from offshore energy and marine mineral exploration, development, and production activities on human, marine, and coastal environments. The proposal, selection, research, review, collaboration, production, and dissemination of each of BOEM's Environmental Studies follows the DOI Code of Scientific and Scholarly Conduct, in support of a culture of scientific and professional integrity, as set out in the DOI Departmental Manual (305 DM 3).