APPENDIX F
BIOLOGICAL RESOURCES SUPPLEMENTAL INFORMATION

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Draft

Supplemental Environmental Impact Statement/ Overseas Environmental Impact Statement Atlantic Fleet Training and Testing

TABLE OF CONTENTS

List of Figures

List of Tables

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F BIOLOGICAL RESOURCES SUPPLEMENTAL INFORMATION

F.1 HABITATS

The purpose of this section is to provide basic ecological background information that is not present in the main document of the Supplemental Environmental Impact Statement/Overseas Environmental Impact Statement (EIS/OEIS). With exception to the following topics the remainder of the necessary information can be found in the 2018 Final EIS/OEIS (Section 3.5, Habitats):

- Updates to the distributions of shallow-water coral reefs, live-hard bottoms, and deep-sea coral or sponge habitats have been accessed using the most recent published literature. The most up-to-date mapping of seafloor resources is needed for the most accurate estimates of impact footprint as well as effective mitigation measures that minimize the footprint of sensitive habitats.
- Literature exploration and verification of recent published articles was used to determine the potential effects from the numerous threat sources described in the 2018 Final EIS/OEIS. This information provided vital context for the analysis of cumulative impacts.

Literature exploration and verification of recent published articles was completed to assess the status of the previously discussed threats as well as emerging threats. This information provided vital context for the analysis of cumulative impacts.

F.1.1 GENERAL BACKGROUND

With the exceptions noted below, general background for abiotic habitats in the Study Area, as described in the 2018 Final EIS/OEIS (Section 3.5.2.1, General Background), has not appreciably changed. As such, the information presented in the 2018 Final EIS/OEIS remains valid.

F.1.1.1 Natural Features

F.1.1.1.1 Bottom Habitats

The following information was updated from the 2018 Final EIS/OEIS:

The depth limit for differentiating natural-bottom habitats relevant to biological resources is approximately 2,500 meters (m); the biological resources present on substrate greater than 2,500 m deep is mostly bacteria where chemosynthetic features (e.g., hydrothermal vents) are absent, regardless of substrate type (refer to Section F.3, Invertebrates, for more information on the depth limits). The deeper areas are also devoid of live hard corals (Tittensor et al., 2009), structure-forming sponges (Hourigan et al., 2017), as well as specific fish-habitat associations in the mid-Atlantic region (Ross et al., 2015b). Bottom habitats, including abiotic substrate types, are depicted in Section 3.3 (Habitats) Figure 3.3-1 through Figure 3.3-5 (artificial reefs and bottom habitats figures).

A detailed breakdown of natural-abiotic substrates intersecting specific named training and testing areas is provided in the following paragraphs.

F.1.1.1.1.1 Regions Intersecting the United States Exclusive Economic Zone

For hard substrate/live-hard bottom within regions intersecting the United States (U.S.) Exclusive Economic Zone (EEZ), the highest percent coverages (30 to 72 percent) are mapped in the South Florida Ocean Measurement Facility/Southeast U.S. Continental Shelf, Jacksonville Range Complex/Gulf Stream, Key West Range Complex/Gulf of Mexico, and other Atlantic Fleet Training and Testing (AFTT) inshore waters/Northeast U.S. Continental Shelf. The highest percent coverages for intermediate substrate (30 to 38 percent) are in other training or testing areas/Northeast U.S. Continental Shelf, Lake Borgne/Gulf of Mexico, the Northeast Range Complexes, and the Virginia Capes Range Complex. Soft substrate dominates the abiotic substrate in the vast majority of training and testing locations at depths less than 2,500 m (Table 3.3-1 in Section 3.3, Habitats), with the exception of the South Florida Ocean Measurement Facility. The highest coverage of bottom deeper than 2,500 m and within the U.S. Exclusive Economic Zone include the Sinking Exercise Box/Gulf Stream, other training/testing areas/Gulf Stream, Virginia Capes Range Complex/Gulf Stream, and Northeast Range Complexes/Gulf Stream.

F.1.1.1.1.2 Regions Outside of the United States Exclusive Economic Zone

Other large marine ecosystems and open-ocean areas outside of the U.S. Exclusive Economic Zone are mapped at broad scales by U.S. Department of the Navy (Naval Oceanographic Office). The Canadian Eastern Arctic – West Greenland, Labrador – Newfoundland, and Scotian Shelf ecosystems have relatively little high-quality mapping data that is publicly available for inclusion in the database; these areas also do not have established range complexes and experience very limited amounts of military readiness activities.

F.1.1.1.2 Water Column Habitats

In the Study Area, water column habitats (e.g., floating *Sargassum*) and artificial feature points, including typical current speeds and flow directions, are mapped in Figure 3.3-6 to Figure 3.3-10 (water column figures). The recent satellite-based mapping is more detailed than the generalization drawing of flow directions depicted in the 2018 Final EIS/OEIS's Figure 3.0-7 (Major Currents in the Study Area). For example, the Loop Current is much farther west in the Gulf of Mexico, and the strongest flows of the Gulf Stream are much closer to shore in the U.S. Southeast Atlantic than what Figure 3.0-7 suggests.

F.1.1.2 Artificial Features

The only difference between the subsequent paragraphs and those of the 2018 Final EIS/OEIS is the inclusion of seafloor pipelines as a habitat for biological resources (Schramm et al., 2021).

Artificial reef habitats have been intentionally created with a wide variety of materials ranging from simple concrete blocks to highly engineered structures. They have also been created as a result of structures built for other purposes (e.g., jetties, oil and gas platforms, seafloor pipelines) or unintentional sinking of vessels (i.e., shipwrecks).

Factors such as the materials, structural features, interstitial space, and surface area of the artificial substrate, as well as local environmental conditions influence the variety and abundance of sessile and sedentary organisms that may become established and the relative success of attracting or enhancing local fish populations (Ajemian et al., 2015; Broughton, 2012; Macreadie et al., 2011; Powers et al., 2003; Ross et al., 2016; Schramm et al., 2021).

F.1.1.3 General Threats

Estuarine and ocean habitats worldwide are under pressure from a variety of human activities (Crain et al., 2009) such as coastal development: shoreline stabilization, dredging, flood control, and water diversion; in additional to destructive fishing practices (Auster & Langton, 1999); offshore energy and resource development and extraction (Boehlert & Gill, 2010); global climate change (Hoegh-Guldberg & Bruno, 2010); marine debris (Rochman et al., 2016); and cumulative effects (Clarke et al., 2014; Halpern et al., 2015) all impact these habitats. These types of activities produce a range of physical and chemical stressors on habitats that are only increasing in intensity as economies grow (Danish et al., 2019). Stressors associated with these activities are not distributed randomly across habitat types and ecosystems; most are more prevalent closer to highly developed landscapes (Halpern et al., 2015). Areas where heavy concentrations of human activity co-occur with military readiness activities have the greatest potential for cumulative stress on the marine ecosystem (see Chapter 4, Cumulative Impacts, for more information). Many threats are also related to some degree (e.g., development and pollution are correlated) that can result in either enhancing or diminishing an impact across a substrate. There are also numerous emerging issues facing marine habitats that include both negative and positive effects (e.g., increased toxicity of metal pollution due to ocean acidification, extraction of rare earth metals from deep-sea habitats, colocation of marine activities) (Herbert-Read et al., 2022) that further complicates an understanding of how a single activity or the compilation of activities impacts a specific substrate type.

F.1.1.3.1 Water Quality Degradation

The following information was updated from the 2018 Final EIS/OEIS:

The occurrence of dead zones (areas of low [less than 2.0 milligrams per liter, hypoxic] or no dissolved oxygen [anoxic]) are increasing in coastal waters worldwide, representing a major threat to the health and economy of the United States and elsewhere (Committee on Environment and Natural Resources, 2010). This trend is exemplified most dramatically off the coast of Louisiana and Texas, where the second largest hypoxic zone in the world is associated with the excess nutrient discharge by the Mississippi and Atchafalaya rivers (Figure F.1-1). Hypoxic conditions recur annually in the northern Gulf of Mexico, typically during summer and coinciding with increasing salinity stratification along with higher seasonal riverine input and nutrient loading (May et al., 2012). Most aquatic organisms cannot survive in hypoxic conditions, which form each summer. This area is one of the most-productive recreational and commercial fishing grounds in the conterminous United States. The biological impact of physical disturbance of the bottom during periods of hypoxia is likely minimal due to the relatively low abundance and resilience of surviving organisms (van Denderen et al., 2022). There are no other large hypoxic zones that intersect the Study Area, though there are smaller dead zones associated with large coastal rivers draining into highly developed watersheds (Diaz & Rosenberg, 2008).

F.1.1.3.2 Climate Change

The following information was updated from the 2018 Final EIS/OEIS:

- Sea-level rise, caused by increasing ocean temperatures, is having the greatest impacts on intertidal and coastal ecosystems, mostly due to intertidal and shallow subtidal vegetation trying to migrate with the changing shorelines (discussed more in Section F.2, Vegetation).
- Ocean acidification is starting to inhibit the calcification of some habitat-forming invertebrates (e.g., oysters, hard corals), which in turns affects their growth rate and building of hard substrate (discussed more in Section F.3, Invertebrates).
- Changes in climate are altering ocean circulation, upwelling/downwelling, rainfall, and nutrient distribution patterns (Chen et al., 2019; Gonçalves Neto et al., 2021; Wang et al., 2015). The wet tropical areas and mid-latitude land are experiencing more frequent and extreme precipitation (Allan et al., 2020), which is increasing erosion-related sedimentation and runoff to coastal habitats (Keener et al., 2012). These can alter primary productivity to a system and thus result in food-web-level impacts.

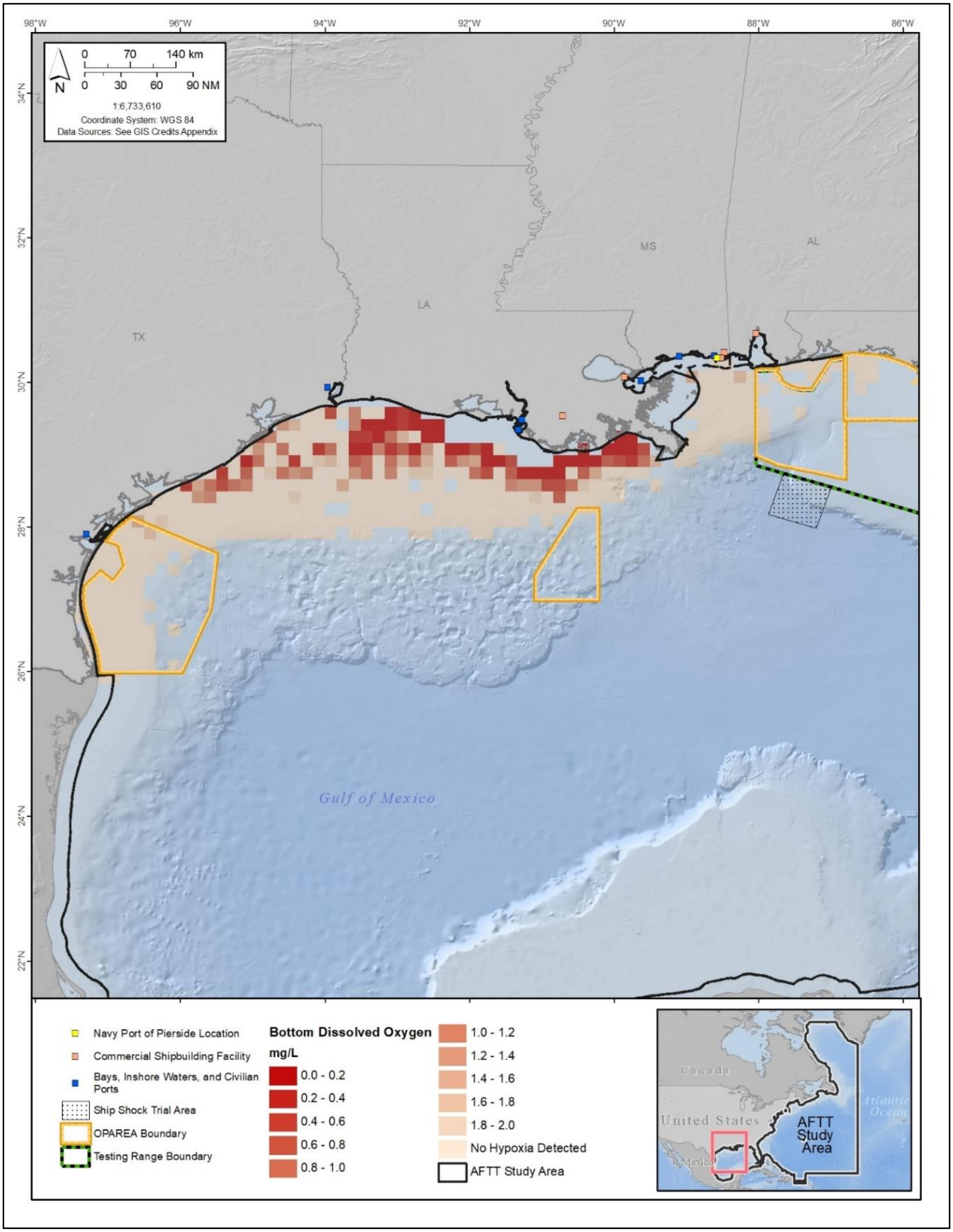

Figure F.1-1: Extent of Seasonal Hypoxia (e.g., Low Dissolved Oxygen) in the Northwestern Gulf of Mexico from 2001 to 2011

F.1.1.3.3 Marine Debris

A multi-year study conducted from 1997 to 2007 along the southeast Atlantic coast concluded that marine debris was mostly general source (land or water-based) or land-based (42 and 38 percent, respectively) followed by ocean based (e.g., items originating from recreational and commercial fishing, shipping, and tourism activities) (20 percent) (Ribic et al., 2010). A similar study conducted along the U.S., Caribbean, and Gulf of Mexico coasts reported similar results (Ribic et al., 2011). An assessment of marine debris collected between 2008 and 2015 in the mid-Atlantic region (Delaware to Virginia) found that the most abundant debris items were plastic, foam, and tobacco-related products (Mid-Atlantic Regional Council On The Ocean, 2015). The majority of marine debris reported in these studies were plastic. U.S. Navy vessels have a zero-plastic discharge policy and return all plastic waste to appropriate disposal or recycling sites onshore, with the exception of military expended materials containing plastic that are not recovered.

Other studies conducted along the northwestern Atlantic shelf break and slope have either observed relatively high concentrations of lighter marine debris (e.g., plastic fragments) in submarine canyons (Jones et al., 2022) or scattered heavier debris (Quattrini et al., 2015). Quattrini et al. (2015) used a remote operated vehicle to survey over 26 kilometer (km) of seafloor and noted 140 items of marine debris. The items were mostly derelict fishing gear and other debris (e.g., soda cans, glass bottles, balloons, rugs, plastic bags). Bauer et al. (2008) conducted a remote operated vehicle study of marine debris in Gray's Reef National Marine Sanctuary and found mostly fishing gear snagged on ledges. Fishing gear was also found along mesophotic depths (30 to 120 m) of the Florida Reef Tract intersecting the Key West Range Complex (Reed et al., 2021). No items of military origin were noted in any of these surveys. In the Mediterranean, lighter debris were more often observed in shallower waters compared to deeper waters where heavier items were also observed (mostly along shipping routes) among generally higher concentrations of debris (Ramirez-Llodra et al., 2013). The dynamic nearshore environment likely resulted in heavier items being buried in the soft substrate (Jenkins & Wever, 2007) while lighter items remained on the surface to be moved greater distances from where they initially settled (Jones et al., 2022).

F.2 VEGETATION

The purpose of this section is to provide basic ecological background information that is not present in the main document of the Supplemental EIS/OEIS. With exception to the following topics the remainder of the necessary information can be found in the 2018 Final EIS/OEIS (Section 3.3, Vegetation):

- Updates to the numbers and population statuses of vegetation species in the Study Area have been accessed using the most recent published literature. This information more accurately characterizes the affected environment.
- Depth limits and distribution of coastal wetlands, seagrass beds, and benthic macroalgae habitats have been updated using the most recent published literature. This information affects the analysis directly because some activities have specific depth ranges in which they occur. If vegetation does not co-occur within the activity's depth range, then the activity is assumed to have no impact on vegetation.
- Literature exploration and verification of recent published articles to determine the potential effects from the numerous threat sources described in the 2018 Final EIS/OEIS. This information provides vital context for analysis of cumulative impacts.

- Literature exploration and verification of recent published articles to assess the status of the previously discussed threats as well as emerging threats. This information provides vital context for analysis of cumulative impacts.

F.2.1 GENERAL BACKGROUND

Broadly speaking, aquatic vegetation is a relatively stationary feature of estuaries and oceans that includes many growth forms and species forming the foundation of the food chain and biotic habitat for animal populations. There are approximately 8,600 marine plant species catalogued in the world's oceans (Mora et al., 2011) and many more that are likely unidentified (Isbell et al., 2023). Estuarine and marine ecosystems depend almost entirely on the energy produced by marine vegetation through photosynthesis (Castro & Huber, 2000), which is the transformation of the sun's energy into chemical energy. In the lighted surface waters of the open ocean and coastal waters, marine algae and flowering plants provide oxygen and habitat for many organisms in addition to forming the base of the food web (Dawes, 1998). Vegetation in the Study Area is comprised of many thousands of plants and plant-like species spanning many taxonomic groups (taxonomy is a method of classifying and naming organisms). The basic taxonomic groupings of vegetation include microalgae (e.g., phytoplankton, surface algae), macroalgae (e.g., seaweed, kelp), and vascular plants.

- Microalgae are either single-celled attached (sometimes filamentous or chain forming) or individuals that can be free-floating (phytoplankton) or substrate-oriented (surface algae).
- Macroalgae are multi-celled algae that grow loosely attached to hard intertidal (e.g., dead man's fingers, *Codium*) or subtidal substrate (e.g., kelp, *Laminaria*) or drift about in floating mats (e.g., *Sargassum*).
- Vascular plants include seagrasses and coastal wetlands:
  - Seagrasses are flowering plants that grow fully submerged and rooted in the shallow and sheltered margins of estuarine or marine ecosystems.
  - Coastal wetland plants (e.g., salt-brackish marsh, mangroves) grow rooted in intertidal flats along the sheltered margins of estuarine or marine ecosystems.

It is estimated that most marine plant species have not yet been described (Mora et al., 2011). The total number of plant species that occur in the Study Area is therefore unknown but is likely to be several hundred at least (Fautin et al., 2010). The results of a research effort to estimate the number of marine plant species in various areas identified 246 species in the Northeast U.S. Continental Shelf Large Marine Ecosystem, 113 in the Southeast U.S. Continental Shelf Large Marine Ecosystem, and 967 species in the Gulf of Mexico Large Marine Ecosystem (Fautin et al., 2010).

F.2.1.1 Habitat Use

F.2.1.1.1 Shoreline Habitats

Emergent wetlands are found along the shallowest margins of estuaries, coastal lagoons, tidal creeks or rivers, wherever the sediment is adequate to support root development (Levinton, 2013a; Mitsch et al., 2009). Saltmarsh cordgrass (*Sporolus* spp., formerly *Spartina* spp.) dominates emergent wetlands along most estuarine (or otherwise sheltered) shorelines of the Study Area. Mangroves dominate in areas that do not regularly have freezing temperatures, including tropical and subtropical waters of Texas, Louisiana, South Florida, and Puerto Rico (Bunting et al., 2018). The most current mapping of emergent wetlands intersecting the Study Area is depicted in Section 3.3 (Habitats) Figure 3.3-1 to Figure 3.3-5; the total area of coastal emergent wetlands mapped in the Study Area is 1,611 square kilometers (km^2).

F.2.1.1.2 Bottom Habitats

The following information was updated from the 2018 Final EIS/OEIS:

- Benthic algae can grow at depths with less than 1 percent light penetration, which reaches approximately 95-m depths in the clearest conditions that occur in the offshore waters of North Carolina (Smith Jr., 1981) depicted in Section 3.3, Habitats (Figure 3.3-1 to Figure 3.3-10). The depth limit for benthic algae in the 2018 Final EIS/OEIS was stated as 200 m without an adequate reference. The maximum depth of a resource matters for analysis. Using the updated parameters, benthic macroalgae could occur on 47,000 km^2 of seafloor in Study Area locations (refer to Table 3.3-1 in Section 3.3, Habitats, for supporting data).
    - The depth limits of algae growth vary in the Study Area, with relatively shallow depths on the U.S. northeast Atlantic coast and greater depths around South Florida and Gulf of Mexico. Kelp (*Laminaria* species; a large, brown seaweed) and other leathery seaweeds occur from the low tide line out to only about 20 to 40 m in the New England region, depending on water clarity (Calvert & McGonigle, 2020; Luning, 1990; Steneck et al., 2002; Vadas & Steneck, 1988).
    - Red macroalgae grows down to 63, 85, and 105 m in the Gulf of Maine (Vadas & Steneck, 1988), northwestern Gulf of Mexico (Clark et al., 2014), and Pulley Ridge (Reed et al., 2019), respectively.
- Seagrasses grow from the intertidal zone to a maximum depth of about 99 feet (ft. [30 m]) in the clear, protected waters off southern Florida (Duarte et al., 2007). The 2018 Final EIS/OEIS reported a maximum depth of 297 ft. (90 m) from seagrass growth, but the references used included Duarte et al. (2007) without determining if it applied to the Study Area.
- The most current mapping of seagrass beds intersecting the Study Area is depicted in Section 3.3 (Habitats) Figure 3.3-1 to Figure 3.3-5. The total area of seagrass beds mapped in the Study Area is 7,821 km^2.

F.2.1.1.3 Water Column Habitats

In general, marine plants inhabiting the ocean water column include various species of phytoplankton and floating seaweed. For a description of phytoplankton taxa that were discounted for analysis in Section 3.4, Vegetation, refer to the 2018 Final EIS/OEIS (Section 3.3.2.3, Species Not Listed Under the Endangered Species Act). Floating *Sargassum* (a brown macroalgae) forms floating mats that are important habitats for marine life (South Atlantic Fishery Management Council, 2002). In the Study Area, floating *Sargassum* occurs in all the offshore ranges (Section 3.4, Table 3.4-1). The seasonal coverage of floating *Sargassum* is depicted in Figure 3.3-6 through Figure 3.3-10 of Section 3.3, Habitats, and described in Section F.2.2, Species Not Listed under the Endangered Species Act. The 2018 Final EIS/OEIS did not describe the seasonal distribution of floating *Sargassum.*

F.2.1.2 General Threats

Human activities and associated effects on marine and estuarine vegetation include excessive nutrient input (such as fertilizers), siltation (the addition of fine particles to the ocean), pollution (oil, sewage), climate change, fishing practices, shading from structures, habitat degradation from construction and dredging, and introduced or invasive species. These activities produce a range of physical and chemical stressors on marine habitats, including vegetation, that are increasing in intensity as economies grow (Danish et al., 2019). Most stressors associated with these activities are close to highly developed landscapes, rather than randomly distributed (Halpern et al., 2015). In addition to the well-known threats to marine vegetation, there are also several emerging threats (e.g., increasing wildfire impacts, coastal darkening) that have yet to be studied thoroughly (Herbert-Read et al., 2022).

The biodiversity of marine plants has experienced dramatic declines worldwide that are predicted to continue (Isbell et al., 2023). The estimated declines in biodiversity are relatively high compared to marine invertebrates (Isbell et al., 2023). The primary indirect threats to marine plants (in order of significance) include human population, production and consumption, governance, technology, and trade. The primary direct threats to marine plants include land and sea-use change, climate change, pollution, and overexploitation. Refer to the threat subsections below for details and citations.

The threats facing marine vegetation in general are further described in the subsections below. Taxa-specific threats are updated in Section F.2.2 (Species Not Listed under the Endangered Species Act).

F.2.1.2.1 Development and Human Activities

The impacts on marine and estuarine vegetation are primarily from inshore and nearshore developments and associated activities described in Section F.1.1.3 (Habitats, General Threats). The nature of the threat from development is taxa-specific, with nearshore taxa (e.g., coastal wetlands, seagrass, kelp) affected more by coastal development than offshore taxa (e.g., floating *Sargassum* and other seaweed). Offshore taxa are more likely to be affected by offshore renewable resource development (e.g., wind turbines), which can cause an increase in hard substrate where there is sufficient light for algae growth (Karlsson et al., 2022; Tsiamis et al., 2020). However, the effects of offshore renewable energy infrastructure on benthic habitats includes some significant unknowns, such as the hydrodynamic changes that could alter primary production (e.g., algae growth) and allow the range expansion of non-native/invasive species (Dannheim et al., 2019). This is important to the extent it applies to military expended materials.

F.2.1.2.2 Water Quality Degradation

The overall impact of pollutants on marine and estuarine habitats from oil spills and polluted runoff/discharge is described in Section F.1.1.3 (Habitats, General Threats). Not only does coastal pollution and agricultural runoff degrade water quality for submerged plant species (e.g., seagrass), it can also cause toxic “red tides” and other harmful algae blooms. Harmful algae blooms are diverse phenomena involving multiple species and classes of algae that occupy a broad range of habitats from lakes to oceans and produce a multiplicity of toxins or biologically active compounds that impact aquatic life (Anderson et al., 2021). Based on 30 years of data for the United States, there has been a significant increase in all harmful algae bloom events caused by the dispersion of species to new areas, the discovery of new harmful algae bloom poisoning effects, and the exacerbating effects of human activities like nutrient pollution, aquaculture expansion, ocean warming, and other activities/effects (Anderson et al., 2021).

F.2.1.2.3 Climate Change

The overall impact of climate change on marine and estuarine habitats is described in Section F.1.1.3 (Habitats, General Threats). The most obvious consequence of climate change on marine and estuarine vegetation is a landward shift that occurs along natural or undisturbed shorelines, where the overall photic zone can move upslope with sea-level rise. As such, sea-level rise may reduce the spatial extent of some biogenic habitats (e.g., wetlands) where it outpaces the accretion rates of habitat-forming species even where upslope migration is unobstructed by vertical-shoreline structures (Schuerch et al., 2018). However, the effect is highly dependent on the slope of the shorelines being inundated and extent of shoreline development. The effect of sea-level rise and increasing storm intensities is most significant along shorelines with artificial-vertical stabilization (e.g., bulkheads, sea walls) or steep elevation gradients that prevent upslope movement of shallow, nearshore habitats (Borchert et al., 2018). Artificial stabilization also deepens the nearshore environment and increases wave energies along adjacent shorelines, which increases the erosion of some shorelines that provides a sediment supply for other habitats (Palinkas et al., 2018).

The forces of erosion and sea-level rise along increasingly armored shorelines will likely continue reducing the area of suitable habitat for vegetation (e.g., marsh grasses, seagrasses, benthic macroalgae). Eight sub-estuaries of the Chesapeake Bay (including the Virginia Capes Inshore training area) are more than 50 percent hardened/armored (e.g., bulkheads, rip-rap), and 23 more are between 30 and 50 percent hardened/armored (Erdle et al., 2006; Patrick et al., 2014). Armoring will probably increase through this century as coastal zone populations increase and as sea levels continue to rise in response to global-climate change (Curtis & Schneider, 2011; Dugan et al., 2008; Small & Nicholls, 2003). The development of small channels connecting expanding shoreline developments (e.g., marinas, community docks) with established navigation channels will likely increase as well. The threats facing migration of shallow-water coastal habitats along the Atlantic coast are further exacerbated by estimates that 60 percent of land below 3.3 ft. (1 m) in elevation is open for development, with only 10 percent set aside for conservation (Titus et al., 2009); with sea-level rise, shallow-coastal habitats need high-elevation wetlands and low-elevation uplands to migrate into.

Increased atmospheric carbon dioxide and sea temperature may lead to several impacts that could affect vegetation (e.g., stratification, acidification). Warmer waters may lead to a greater stratification in the water column, which may support harmful algal blooms (World Ocean Review, 2015). The stratification may also inhibit upwelling, as seen during El Niño events, which would prevent nutrients from circulating to the surface (Lehmköster, 2015; World Ocean Review, 2015), reducing photosynthetic capabilities. Additionally, increased sea temperatures may lead to changes in the composition of marine vegetation communities (Schiel et al., 2004). Ocean acidification with increased dissolved carbon dioxide has also been shown to reduce the ability of seagrasses to deter herbivores (Arnold et al., 2012).

F.2.1.2.4 Other Threats

Other threats to marine and estuarine vegetation include marine debris and invasive species. Marine plastic debris has only speculative effects on algae and seagrass directly, though there may be an indirect effect on plant-eating consumers (Bonanno & Orlando-Bonaca, 2020). Other types of marine debris have not been recognized as a threat to aquatic vegetation, except where it accumulated in very high concentrations (e.g., litter accumulation in wetland areas). In Caribbean seagrass meadows and perhaps elsewhere, the ecosystem services of native species are threatened by a combination of intensive grazing and opportunities invasive seagrass species that leave seabeds more vulnerable to erosion from storms (James et al., 2020).

F.2.2 SPECIES NOT LISTED UNDER THE ENDANGERED SPECIES ACT

For the following discussion, vegetation have been divided into three major taxonomic groups (microalgae, macroalgae, and vascular plants) and eight major phyla that have distinct morphological, biochemical, physiological, and life history traits that reflect their evolutionary history and influence their distributions and ecological relationships.

F.2.2.1 Microalgae

The great diversity of microalgae makes generalization difficult, but overall, algae are resilient and are able to colonize disturbed environments created by stressors (Levinton, 2013b). Trends in the abundance of many microalgae species is unknown, but likely increasing due to excess nutrients reaching surface waters. However, excess nutrients tend to favor an overabundance of harmful algae species (Wurtsbaugh et al., 2019).

F.2.2.2 Macroalgae

The three primary groups of macroalgae include green, brown, and red algae. Among the prominent macroalgae groups mentioned previously, floating *Sargassum* has been increasing in abundance from 2011 to 2018, though the increase is due to massive amounts of the seaweed on Caribbean beaches that originated in the central Atlantic (Gower & King, 2020; Gower et al., 2013; Wang et al., 2019). The causes for the persistent increase have been attributed to nutrient enrichment from coastal rivers and climate variation. However, kelp forests in New England have been declining due to climate change and invasive species (Witman & Lamb, 2018); a decline that is most dramatic in shallow, nearshore habitats. There has also been an increase in benthic macroalgae at the expense of hard corals in the western Atlantic over time (Tebbett et al., 2023).

F.2.2.2.1 Brown Algae (Phylum Phaeophyta)

The following information on kelp (*Laminaria* species) was updated from the 2018 Final EIS/OEIS:

- Kelp species are abundant on Cashes Ledge in the central Gulf of Maine (Witman & Lamb, 2018) where leathery macroalgae make up 70 percent of the benthic community on rocky habitats less than 40 m deep (Calvert & McGonigle, 2020). This habitat also coincides with the Northeast Range Complex. The density of kelp on Cashes Ledge is much higher than rocky habitats closer to shore (Witman & Lamb, 2018).
- The primary threat to kelp beds globally is considered to be coastal development followed closely by water pollution and overharvesting (Morris et al., 2020). In the New England area, commercial farming/harvesting of kelp and other “rockweeds” (attached seaweed species) has raised concerns about the ecological effects on the associated marine animals that depend on kelp beds as habitat (Kim et al., 2019). Maine has recently developed a rockweed fishery management plan aimed at ensuring the sustainable use of this resource (Maine Department of Marine Resources, 2014). Fishing can also impact kelp forests via the ecological effect of removing kelp consumers (e.g., sea urchins) or the predators of those consumers (e.g., large, benthic carnivores (Steneck et al., 2002).

The following information on floating *Sargassum* was updated from the 2018 Final EIS/OEIS:

- In the western North Atlantic, *Sargassum* can double in size every 15 to 28 days, depending on the level of nutrient enrichment (Lapointe, 1986). The growth rate of vegetation determines how quickly it can recover from physical disturbance and strike.
- Given the size of the Study Area (approximately 2.6 million square nautical miles [NM^2]), the relative coverage of *Sargassum* ranges from less than 1 percent to 5 percent of the sea surface (2018 Final EIS/OEIS). However, this estimate does not account for the massive infusion of *Sargassum* from the Central Atlantic that has been occurring in recent years (Wang et al., 2019). However, the central Atlantic is not in the Study Area and the quantity of floating *Sargassum* flowing into the Study Area from this region is unknown.

F.2.2.3 Vascular Plants (Phylum Tracheophyta)

F.2.2.3.1 Seagrasses

The following information was updated from the 2018 Final EIS/OEIS:

- The largest area of seagrass in the Study Area occurs in the inshore locations associated with the Northeast, Virginia Capes, Key West, and Gulf of Mexico range complexes (see Section 3.3, Habitats, Figure 3.3-1 to Figure 3.3-5). A review of seagrass from 1879 to 2006 found that global seagrass coverage decreased by 75 percent (Waycott et al., 2009). However, the study documented much less severe declines or slight increases within inshore training or testing areas where more recent trend analysis were available. The nuanced picture for seagrass trends in training and testing locations was not mentioned in the 2018 Final EIS/OEIS.
- There is some evidence suggesting seagrass growth is stimulated with ocean acidification, which will help to offset the stress of increasing water temperatures (Zimmerman, 2021).

F.2.2.3.2 Coastal Wetlands

Areal coverage of coastal wetlands (mostly saltmarsh cordgrass, but also mangroves) on the U.S. Atlantic and Gulf of Mexico coasts have decreased dramatically during the 20th century, with additional losses of 0.4 and 5.2 percent on the Atlantic and Gulf coasts, respectively, from 2004 to 2009 (Dahl & Stedman, 2013). In the Gulf of Mexico, the primary causes for the decline were erosion and/or inundation due to climate change and oil/gas development. Based on satellite imagery from 2011 and 2016 for the continental United States, Homer et al. (2020) estimated a continued decline of wetlands in most coastal draining watersheds adjoining the Study Area. Boat wakes in sheltered inshore waters can also erode shorelines and fringing wetlands that would otherwise be relatively stable (Fonseca & Malhotra, 2012; Parnell et al., 2007).

F.2.2.3.2.1 Saltmarsh Cordgrass

The following information was updated from the 2018 Final EIS/OEIS:

- The most common plant species of salt and brackish marshes in the Study Area is known as smooth or salt marsh cordgrass (*Sporobolus alterniflorus)* (Peterson et al., 2014). The species scientific name was *Spartina alterniflora* in the 2018 Final EIS/OEIS.

F.2.2.3.2.2 Mangroves

The following information was updated from the 2018 Final EIS/OEIS:

- Mangroves occur along the shores of Florida, Louisiana, Texas, and Puerto Rico in the Study Area (Bunting et al., 2018). Mangroves are now documented to occur in more states than was reported in the 2018 Final EIS/OEIS.
- The area of mangroves in the Study Area is included in the estimate reported for coastal emergent wetlands.

F.3 INVERTEBRATES

The purpose of this section is to provide basic ecological background information that is not present in the main document of the Supplemental EIS/OEIS. With exception to the following topics the remainder of the necessary information can be found in the 2018 Final EIS/OEIS (Section 3.4.2.1, General Background):

- Updates to the numbers and population statuses of invertebrate species in the Study Area have been accessed using the most recent published literature. This information more accurately characterizes the affected environment.
- Literature exploration and verification of recent published articles to determine the soft-bottom habitats that dominate the Study Area seafloor and the depth distribution, size range, and density of benthic invertebrates offshore. This information strengthens the reasoning presented

in the 2018 Final EIS/OEIS for the diminished impact of physical disturbances on the most resilient habitats and deepest regions of the Study Area.

- Literature exploration and verification of recently published articles to assess the distribution of shallow-water coral reefs and shallow-water invertebrate beds. Up-to-date mapping of seafloor resources is needed for the most accurate estimates of impact footprint as well as effective mitigation measures that minimize the impact footprint on sensitive habitats.
- Literature exploration and verification of daily vertical migrations of many pelagic invertebrates and the distribution of aerial insects in the Study Area. This information is directly used in the analysis to address impacts to surface waters that occur mostly during the day.
- Literature exploration and verification of recent published articles to determine the potential effects from the numerous threat sources described in the 2018 Final EIS/OEIS. This information provided vital context for the analysis of cumulative impacts.
- Literature exploration and verification of recent published articles to assess the status of the previously discussed threats as well as emerging threats. This information provides vital context for analysis of cumulative impacts.

The following topics on Endangered Species Act (ESA)-listed invertebrate species were updated from the 2018 Final EIS/OEIS (Section 3.4.2.2):

- Proposed listing and reclassification of ESA-listed marine invertebrate species and designation of critical habitats.
- Population statuses and distribution of ESA-listed and ESA-proposed invertebrate species and current threats based on latest 5-year review by the National Marine Fisheries Service (NMFS). This information provides vital information for the intersection of stressors and affected environments and vital context for cumulative impacts analysis.

The following topics on specific invertebrate taxa were updated from the 2018 Final EIS (Section 3.4.2.3):

- Typical percent coverage of living shallow- and deep-water hard coral species on live hard bottoms. The information provides important qualifiers for the estimated footprint of physical disturbance on the habitat for these species.

F.3.1 GENERAL BACKGROUND

Invertebrates, which are animals without backbones, are the most abundant life form on Earth, with marine invertebrates representing a large, diverse group with 246,352 species accepted worldwide to date (World Register of Marine Species Editorial Board, 2024) with many more that are likely unidentified (Mora et al., 2011). The results of a research effort to estimate the number of marine invertebrate species in various areas identified over 3,000 species in the Northeast U.S. Continental Shelf Large Marine Ecosystem and over 10,000 species in the Gulf of Mexico (Fautin et al., 2010). Of the known species, a large portion are lacking the data necessary to estimate their population statuses (Collier et al., 2016).

Aerial invertebrates (i.e., flying insects) may also occur along the Study Area coastline at low altitudes (e.g., butterflies) as well as high over the ocean during wind-assisted migrations. However, relatively few flying insect species occur over large expanses of open water (see Section F.3.1.2, Movement and Behavior, for supporting details).

F.3.1.1 Habitat Use

F.3.1.1.1 Bottom Habitats

As a general rule, benthic invertebrate size and biomass decreases exponentially with increasing depth in the ocean; with greater declines for larger species (Rex et al., 2006). Benthic macroinvertebrates dominate over bacterial biomass in depths of 0 to 4,000 m (continental shelf to bathyal zone), and bacterial biomass dominates the abyssal zone (2018 Final EIS/OEIS's Figure 3.0-2, Three-Dimensional Representation of the Intertidal Zone, Continental Margin, Abyssal Zone, and Water Column Zones). The only areas of the bathyal and abyssal zone known to be densely populated by macroinvertebrates are around hydrothermal vents and cold seeps.

Soft bottom constitutes one of the largest habitat types on earth, covering roughly 80 percent of Earth's seafloor (Byers & Grabowski, 2013) and 84 percent in the mapped portion of the Study Area (Table 3.3-2, Shipwrecks and Designated Artificial Reefs in Training and Testing Locations of the Study Area). The invertebrate species present depend on the sediment grain size and associated water conditions (e.g., salinity, depth, dissolved oxygen, flow/turbulence). Long-lived suspension feeders (e.g., filter-feeding bivalves crustaceans, soft corals) commonly occur in higher-energy/sandy sediment and deposit feeders (e.g., snail, worms, sea cucumbers) commonly occur in lower-energy/muddy sediment. Based on the predictive relationships developed in Rex et al. (2006), the biomass of benthic macroinvertebrates at 200 to 4,000 m depths is approximately 3.84 to 0.07 tons (7,680 and 150 pounds [lb.], respectively) per square mile. At depths of less than 200 m on the U.S. Northeast Continental Shelf, the average biomass of benthic macroinvertebrates ranged from 127 to 267 tons per square mile (mi^2) (Theroux & Wigley, 1998). Whereas soft bottom habitats are relatively unstructured, there are features present in the form of depressions, ripples, tubes, and burrows that take time to recover after physical disturbance (van Denderen et al., 2015). Such features also recover more quickly than layered structures forming on hard substrate (e.g., oysters). There are also small patches of soft bottom that receive a net benefit from proximity to hard/intermediate bottom for various reasons (Byers & Grabowski, 2013). The distribution of soft and intermediate bottom in the Study Area is depicted in Section 3.3 (Habitats) Figure 3.3-1 to Figure 3.3-10.

Other, less common, benthic habitats in the Study Area include submerged aquatic vegetation (e.g., kelp forests, seaweed beds, seagrass beds, coastal wetlands), shallow-water coral reefs, and various types of live-hard bottom (e.g., seaweed beds, worm reefs, shallow-water invertebrate beds, deep-water invertebrate beds). The distribution of sessile invertebrates associated with live-hard bottoms (e.g., sponges, anemones, hard corals, barnacles, oysters, mussels) is approximated by the location of artificial, hard, and (to a lesser extent) intermediate substrate depicted in Section 3.3 (Habitats) Figure 3.3-1 to Figure 3.3-10. The bottom types associated with live-hard bottoms cover 16 percent of substrate types in the mapped portion of the Study Area (Table 3.3-1 of Section 3.3, Habitats).

Shallow-Water Coral Reefs

The hard consolidated structures characteristic of shallow-water coral reefs are created by symbiotic algae that are generally limited to shallow depths (less than 99 ft. [30 m] in the tropics), high salinities, and annual-minimum temperatures greater than 18 degrees Celsius (˚C) (Kleypas et al., 1999). However, the depth limit referenced in Kleypas et al. (1999) was based on what could be mapped from the air; there are also shallow-water coral reef forms (i.e., low relief plates) with symbiotic Coralline algae that grow down to 495 ft. (150 m) (Rocha et al., 2018), though they have only been documented in the Study Area down to 52 and 105 m in the Flower Garden Banks/northwest Gulf of Mexico (Clark et al., 2014) and Pulley Ridge (Reed et al., 2019). Shallow-water coral reefs in water deeper than 30 m are referred to as "mesophotic" reefs. Shallow-water coral reefs are not expected in estuarine/inshore waters of the Study Area due to unsuitable salinities and/or temperatures. Shallow-water coral reefs are also not expected on artificial structures overlapping the high-salinity zone of Port Canaveral because this area is substantially north of the known shallow water coral reefs. Shallow coral reefs only overlap the AFTT training and testing ranges of Key West (e.g., Pulley Ridge, Florida Reef Tract) and the South Florida Ocean Measurement Facility, respectively. The remaining coral reefs in the Study Area (e.g., Flower Garden Banks, reefs off Mexico, Bermuda, and Puerto Rico) do not overlap any named training or testing area (e.g., Key West Range Complex). The total estimated area of shallow-water coral reefs in the Study Area is 4,724 km^2 as depicted in Section 3.3 (Habitats) Figure 3.3-1 to Figure 3.3-5, although the mapping resolution and percent coverage in living hard coral colonies varies greatly between reef areas. The defining characteristic of a shallow-water coral reef is not necessarily the presence of living, reef-building corals, but rather the structural presence of coral skeletons with some amount of coverage in living coral colonies.

Shallow-Water Invertebrate Beds

Shallow-water invertebrate beds are a type of live-hard bottom that occurs on shallow, hard-bottom areas too cold for shallow-water coral reefs. These habitats provide attachment sites for benthic macroalgae, oysters, barnacles, bryozoans, limpets, sea anemones, sea fans, sponges, and tunicates, among others. Other invertebrates move about or shelter in crevices, including crustaceans (e.g., crabs, lobsters), echinoderms (e.g., brittle stars, sea cucumbers, sea urchins, sea stars), and molluscs (e.g., snails, nudibranchs, sea hares, octopus). The maximum depth of this habitat varies from region to region, depending on the maximum depth of benthic algae growth (refer to Section F.2, Vegetation, for details). Relatively shallow areas are dominated by vegetation (e.g., seaweed, seagrass) and relatively deep areas are dominated by sessile invertebrates. On the U.S. South Atlantic Continental Shelf, the coverage of sessile organisms on shallow, hard bottom present in Gray's Reef National Marine Sanctuary (20 to 30 m depth) varies according to topography, with 1 to 20 percent coverage on low-relief areas, and nearly 100 percent coverage on high-relief areas (Kendall et al., 2005). The areas of low- and high-relief hard bottom cover 25 and less than 1 percent of Gray's Reef bottom features, respectively.

F.3.1.2 Movement and Behavior

Planktonic animals commonly undergo daily migrations to surface waters at dusk and return to deeper waters at dawn (Brierley, 2014). This includes small, microscopic zooplankton and larvae, larger crustaceans (e.g., shrimp), and jellyfish. Many zooplankton, mysid shrimp and squid species that occur in the water column at night, migrate to deeper waters during the day. At any time of the day, jellyfish may occur in various portions of the water column, including near the surface. Large scale analysis using acoustic backscatter data indicates that vertical migrants leave the ocean surface at approximately 20 minutes before sunrise and return to the depths approximately 17 minutes after sunset (Bianchi & Mislan, 2016). The depth of migration varies from 200 to 650 m, depending on the depth of the oxygen-minimum layer (Bianchi et al., 2013).

Aerial insects have a predisposition to resist being carried over the sea unless they are habitual transoceanic migrants (literature review in Becciu et al., 2019). Habitual transoceanic migrants include some night-flying moths and day-flying butterflies, among an array of insects considered mostly pests (Chapman et al., 2010; Drake & Reynolds, 2012). Other insects collected on offshore oil/gas platforms (mostly true bugs and flies) in the Gulf of Mexico may have been attracted to the lights during storms that swept them out to sea (Keaster et al., 1996). Of particular interest in the Study Area is the monarch butterfly *Danaus plexippus* that is a candidate for proposed listing under the ESA (85 *Federal Register* 81813). In the Study Area, the eastern sub-population of monarch butterflies migrate mostly along the Gulf of Mexico and western Atlantic coasts (Brower, 1996; Tracy et al., 2019b), with speculative, high-altitude migration across the open water from the northeastern Gulf of Mexico to the mountains of central Mexico (Brower, 1996). However, more recent research modeling the migration routes of monarch butterflies did not include any routes over the Gulf of Mexico or Atlantic Ocean (Tracy et al., 2019b).

F.3.1.3 Sound Sensing and Production

The background information for sound sensing and production of invertebrates in the Study Area as described in the 2018 Final EIS/OEIS (Section 3.4.2.1.3) has not appreciably changed. As such, the information presented in the 2018 Final EIS/OEIS remains valid.

F.3.1.4 General Threats

General threats to marine invertebrates include overexploitation and destructive fishing practices, habitat degradation resulting from pollution, disease, invasive species, oil spills, global climate change and ocean acidification, marine debris, microplastic ingestion, and possibly human-generated noise. There are also emerging issues facing marine invertebrates causing both negative and positive effects (e.g., increasing wildfire impacts, coastal darkening, colocation of marine activities) (Herbert-Read et al., 2022). These issues produce a range of physical and chemical stressors on marine habitats that are only increasing in intensity as economies grow (Danish et al., 2019). Stressors associated with these activities are not distributed randomly across habitat types and ecosystems; most are more prevalent closer to highly developed landscapes (Halpern et al., 2015). And the threats are often correlated, to some degree (e.g., pollution and disease), which can result in either an additive or diminutive effect on the resource.

The biodiversity of marine benthic communities, including estuarine invertebrates, has experienced dramatic declines worldwide that are predicted to continue (Collier et al., 2016; Worm et al., 2006). The loss of species has consisted mostly of uncommon or specialist varieties. Though still alarming, the estimated declines in biodiversity are relatively low compared to other biological resources (Isbell et al., 2023), although the population status of most marine invertebrates is unknown (Collier et al., 2016). For marine invertebrates, the primary indirect threats (in order of significance) include human population, production and consumption, governance, trade, and technology. The primary direct threats are considered climate change, land and sea-use change, overexploitation, pollution, and invasive species. For estuarine invertebrates, the decline has been documented in numerous studies and primarily attributed to pollutants, eutrophication, and hypoxia (i.e., nutrient enrichment and low dissolved oxygen), physical habitat destruction, and invasive species (Collier et al., 2016).

Insect biodiversity (included aerial invertebrates) has been in a dramatic decline that could lead to 40 percent of species going extinct over the next few decades, worldwide (Sánchez-Bayo & Wyckhuys, 2019). As less common/specialist species decline, the abundance of some common/generalist species, including many pest species, has actually increased. The main drivers of decline in insect species (in order of importance) include habitat loss and conversion to intensive agriculture and development, pollution (e.g., synthetic pesticides and fertilizers), biological factors (e.g., pathogens, introduced species), and climate change (Sánchez-Bayo & Wyckhuys, 2019).

F.3.1.4.1 Development and Human Activities

The following information was updated from the 2018 Final EIS/OEIS:

The development of offshore-renewable resources (e.g., wind turbines) may only impact marine invertebrates to the extent hard substrate is increased. However, the effects of offshore-renewable energy infrastructure on benthic habitats includes some significant unknowns, such as the hydrodynamic changes that could alter primary production (e.g., algae growth) and allow the range expansion of non-native/invasive species (Dannheim et al., 2019).

F.3.1.4.2 Other Threats

Other significant threats to marine invertebrates include diseases, invasive species, and oil spills. The review of literature in Ward and Lafferty (2004) documented an increase in marine diseases affecting corals, crustaceans, and echinoderms despite a continuing lack of baseline data for most invertebrate species. Collier et al. (2016) listed invasive species among the key agents of biodiversity decline in aquatic ecosystems, including invertebrates in the marine environment. Geburzi and McCarthy (2018) highlighted trends among successful invasive invertebrate species, from methods of transport (e.g., shipping and ballast water discharge) to ecological flexibility (e.g., attachment to artificial substrate). Oil spills continue to be a stressor on marine ecosystems despite a transition to more renewable resources (Adzigbli & Yuewen, 2018). Some microbes consume small quantities of oil from naturally occurring seeps, but large quantities released in spills would not be able to be consumed. The effects of excess oil include the destruction of food webs and toxicity (both acute and chronic) that vary by species and life stage, as evidenced by the monitoring conducted after the Deepwater Horizon oil spill (Buskey et al., 2016). However, oil spills were not listed among the key agents of biodiversity decline in aquatic ecosystems in Collier et al. (2016), though they are associated with development and human activities that are key agents.

F.3.2 ENDANGERED SPECIES ACT-LISTED SPECIES

There are eight species of invertebrates listed or proposed as threatened under the ESA and known to be present in the Study Area. Seven listed coral species are discussed in Section F.3.2.1 (Elkhorn Coral [*Acropora palmata*]) through Section F.3.2.7 (Rough Cactus Coral [*Mycetophyllia ferox*]). The one non-coral species (queen conch [*Alger gigas*]) is also ESA threatened and discussed in Section F.3.2.8.

The following subsections reference Figure F.3-1 in describing the habitat of ESA-listed invertebrate species. The only shallow-water coral reef areas fully outside of both the Florida Keys National Marine Sanctuary and the 12-NM boundary from shore are Pulley Ridge and the shelf break connection to the Florida Reef Tract (Figure F.3-1). There have been no ESA-listed coral species observed in these areas to date despite multiple surveys (Halley et al., 2005; National Marine Fisheries Service, 2022j; Reed et al., 2019). There are some mesophotic reef sites along the Florida Reef Tract (at approximately 30 to 116 m depths) that have been surveyed recently (Reed et al., 2021), though there were no ESA-listed coral species observed in survey locations intersecting the Key West or Gulf of Mexico Range Complexes (National Marine Fisheries Service, 2022j).

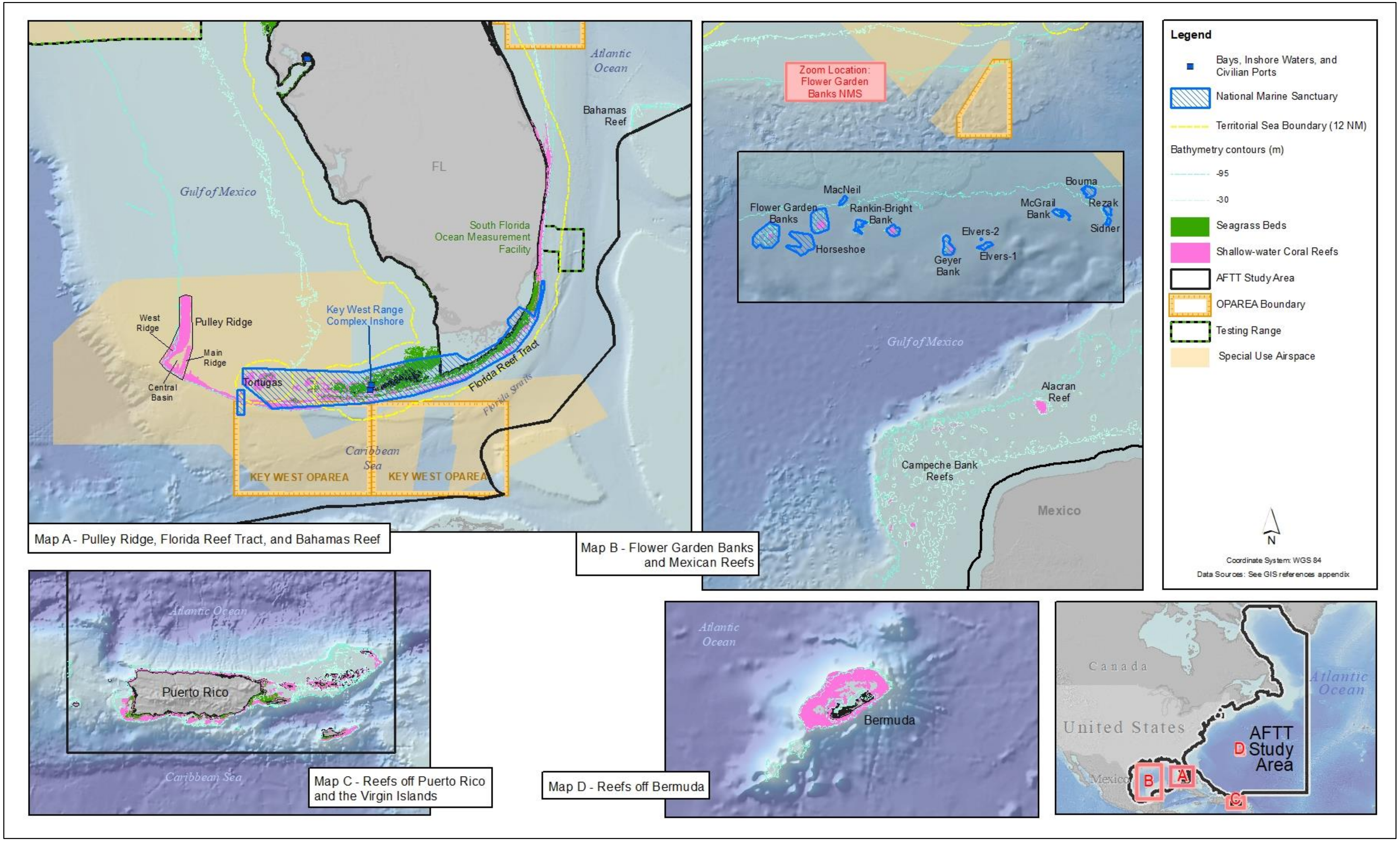

Figure F.3-1: Shallow-Water Habitats for ESA-Listed Coral Species and Queen Conch in the Study Area

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F.3.2.1 Elkhorn Coral (*Acropora palmata*)

F.3.2.1.1 Status and Management

Elkhorn coral is listed as a threatened species under the ESA, and critical habitat has been designated. The critical habitat designation identifies the physical or biological features essential to the species' conservation as, "substrate of suitable quality and availability to support larval settlement and recruitment by sexual reproduction, and reattachment and recruitment of asexual fragments," (74 *Federal Register* 72210). For purposes of this definition, "substrate of suitable quality and availability," means natural consolidated hard substrate or dead coral skeleton that is free from fleshy or turf macroalgae cover and sediment cover (Endangered and Threatened Species; Critical Habitat for Threatened Elkhorn and Staghorn Corals, 73 *Federal Register* 72210, November 26, 2008). This definition applies to depths from mean low water to 30 m. No other essential features were sufficiently definable. The critical habitat designation for elkhorn coral applies to staghorn coral as well (see Section F.3.2.2, Staghorn Coral (*Acropora cervicornis*)). While most shallow-water coral habitat in the Study Area falls within the definition of critical habitat for elkhorn and staghorn coral, the United States contains only about 10 percent of all potential critical habitat in the Caribbean (Bryant et al., 1998). Exemptions from critical habitat designations include a small zone around Naval Air Station Key West and a small area within the South Florida Ocean Measurement Facility Testing Range. The exemption for Naval Air Station Key West was granted in accordance with a provision of the National Defense Authorization Act that allows such exemptions for installations with approved Integrated Natural Resources Management Plans. The exemption for the South Florida Ocean Measurement Facility was granted for national security reasons (73 *Federal Register* 72210). However, ESA protection is not limited to critical habitat designations; the species and where it might occur are also protected via regulatory consultation requirements.

Critical habitat for the species is designated in multiple areas near Florida (1,329 mi^2), Puerto Rico (1,383 mi^2), St. John and St. Thomas (121 mi^2), and St. Croix (126 mi^2) (see Figure 3.5-1, Critical Habitat for Elkhorn and Staghorn Coral and Five ESA-Listed Coral Species in the Study Area). The Florida areas contain the only critical habitats that intersect a range complex or testing range in the Study Area. Areas adjacent to the Naval Air Station Key West and areas within the boundaries of the South Florida Ocean Measurement Facility Testing Range meet the definition of elkhorn critical habitat. However, areas within 50 yards of the shore of Naval Air Station Key West and a small portion of the nearshore footprint of the South Florida Ocean Measurement Facility Testing Range (combined total of 5.5 mi^2) have been exempted from the critical habitat designation (73 *Federal Register* 72210).

A 5-year review was completed by NMFS for this species in January 2022 (National Marine Fisheries Service, 2022j).

F.3.2.1.2 Habitat and Geographic Range

Elkhorn coral is typically found on outer reef crests and slopes with exposure to wave action at depths of 1 to 20 m (3 to 66 ft.), although it has been reported as deep as 30 m (98 ft.) (Aronson et al., 2008b; Boulon et al., 2005). The optimal water temperature range for elkhorn coral is 77 to 84 degrees Fahrenheit (°F), and it requires a salinity range of 34 to 37 parts per thousand (Aronson et al., 2008b; Boulon et al., 2005; Goreau & Wells, 1967). Elkhorn coral inhabits shallow waters with high oxygen content and low nutrient levels (Spalding et al., 2001). Clear, shallow water allows the coral sufficient sunlight exposure to support zooxanthellae (symbiotic photosynthetic organisms; analogous to plants living inside the animals). Elkhorn coral primarily inhabit the seaward margins of reefs where appropriate conditions are more likely to occur (Ginsburg & Shinn, 1964).

Elkhorn corals are typically found in the southeastern part of the Gulf of Mexico Large Marine Ecosystem, the northern part of the Caribbean Sea Large Marine Ecosystem, and the southern part of the Southeast

U.S. Continental Shelf Large Marine Ecosystem. Elkhorn coral distribution in the Study Area extends from southeastern Florida through the Florida Keys and surrounds Puerto Rico and the U.S. Virgin Islands (Aronson et al., 2008b). Elkhorn coral is known to occur in portions of the South Florida Ocean Measurement Facility Testing Range (Gilliam & Walker, 2011) and the Key West Range Complex. Two colonies of elkhorn coral occur in the Flower Garden Banks National Marine Sanctuary in the Gulf of Mexico, but this area is not included in designated elkhorn critical habitat (73 *Federal Register* 72210). Although the Flower Garden Banks National Marine Sanctuary is located in the Gulf of Mexico, it does not intersect a training or testing range. Potential habitat for the species is depicted in Figure F.3-1.

F.3.2.1.3 Population Trends

Elkhorn coral was once a dominant coral on Caribbean reefs and was so abundant that an entire reef zone is named for it. A review of quantitative data of Acroporidae in the wider Caribbean area, including the Florida Keys and Dry Tortugas, indicates a greater than 97 percent reduction of Acroporidae coverage since the 1970s with peak declines in the 1980s (Boulon et al., 2005; National Marine Fisheries Service, 2015b). Multiple stressors, including disease, increased water temperature, decreased breeding population, loss of recruitment habitat, and sedimentation, may be affecting the recovery of this species. The current range of Acroporidae is considered to be the same as the historical range, despite the substantial decline in abundance (Bruckner, 2003; Rothenberger et al., 2008).

Research on the population status of elkhorn coral, in particular, indicates a drastic decline. Surveys of Carysfort Reef (1974 to 1982) and Molasses Reef (1981 and 1986) revealed slight declines or stable colonies (Jaap et al., 1988). It was not until the observation of a 93 percent decrease of coral in Looe Key (1983 to 2000) that the elkhorn coral populations mirrored the substantial decline of other coral species such as staghorn coral (Miller et al., 2002). Continued long-term monitoring in the Florida Keys and the U.S. Virgin Islands has found that elkhorn coral remains at less than 1 percent of all corals on reefs (Rothenberger et al., 2008), and the species' continued decline since 2004 is attributed principally to fragmentation, disease, and predation (Williams & Miller, 2011). Notwithstanding, the additional focus provided by the 2006 decision to list elkhorn coral as threatened, the population has continued to decline by 50 percent or more, recruitment failure has been observed, and genetic studies have shown that approximately half of all colonies are clones, which reduces the number of genetically distinguishable individuals. The latest 5-year review reported further declines in the Florida Keys population based on survey results from 2014 and 2015 (National Marine Fisheries Service, 2022j).

Elkhorn coral can reproduce sexually by spawning (once each year in August or September) (Boulon et al., 2005), or asexually by fragmentation (National Marine Fisheries Service, 2023b). Although fragmentation of adult colonies helps maintain high growth rates (from 4 to 11 centimeters [cm] [approximately 2 to 4 inches (in.)] per year), fragmentation reduces the reproductive potential of elkhorn coral by delaying the production of eggs and sperm for 4 years after the damage occurs (Lirman, 2000). Furthermore, large intact colonies produce proportionally more gametes than small colonies (such as new colonies started from fragmentation) because tissue at growing portions of the base and branch tips is not fertile (National Marine Fisheries Service, 2015b). During sexual reproduction, eggs and sperm immediately float to the sea surface where multiple embryos can develop from the fragmentation of a single embryo. Developing larvae travel at or near the sea surface for up to several weeks (Boulon et al., 2005) before actively seeking specific micro-habitats suitable for growth. Maturity is reached between 3 and 8 years, the average generation time is 10 years, and longevity is likely longer than 10 years based on average growth rates and size (Wallace, 1999). Combined with a severely reduced population, these factors restrict the species' capacity for recovery.

F.3.2.1.4 Predator and Prey Interactions

Predators of corals include sea stars, snails, and fishes (e.g., parrotfish and damselfish) (Boulon et al., 2005; Roff et al., 2011). The marine snail, *Coralliophila abbreviata*, and the bearded fireworm (*Hermodice carunculata*), are the primary predators on elkhorn coral (Boulon et al., 2005).

Corals feed on zooplankton, which are small organisms that inhabit the ocean water column. Corals capture prey with tentacles armed with stinging cells that surround the mouth or by employing a mucus-net to catch suspended prey. In addition to capturing prey, these corals also acquire nutrients through their symbiotic relationship with zooxanthellae. The coral host provides nitrogen in the form of waste to the zooxanthellae, and the zooxanthellae provide organic compounds produced by photosynthesis (the process by which sunlight is used to produce food) to the host (Brusca & Brusca, 2003; Schuhmacher & Zibrowius, 1985). Zooxanthellae also provide corals with their characteristic color.

F.3.2.1.5 Species-Specific Threats

The greatest threat to elkhorn coral is ocean warming. Non-ideal conditions (like increased water temperature) will cause stress to corals, resulting in the release the zooxanthellae (symbiotic algae) that live in their tissue (i.e., and provide them food), usually causing death (National Marine Fisheries Service, 2023b). Other threats to elkhorn coral are ocean acidification (decrease in water pH caused by increased carbon dioxide in the atmosphere) that makes it harder for them to build their skeleton, unsustainable fishing practices that deplete the herbivores (animals that feed on plants) that keep the reef clean, and land-based sources of pollution that impacts the clear, low nutrient waters in which they thrive. Elkhorn coral is also more susceptible to disease than many other Caribbean corals (Aronson & Precht, 2001; Pandolfi et al., 2003; Patterson et al., 2002; Porter et al., 2001). In particular, elkhorn coral is susceptible to “white pox” and “white band” disease. A human fecal bacterium found in several other coral species is known to cause white pox disease in only elkhorn coral (Sutherland et al., 2011). Discharge of sewage from all oceangoing vessels therefore has the potential to expose elkhorn coral to this bacterium. Whiteband disease is associated with naturally occurring microbes that infects elkhorn coral stressed by human activity (Lesser et al., 2007; Muller & van Woesik, 2012) and climate change (Randall & van Woesik, 2015).

NMFS evaluated the population’s demographic, spatial structure, and vulnerability factors and determined that the species was likely to have an, “…extremely high risk of extinction with little chance for recovery…,” by 2100 (Brainard et al., 2011). The following elements contributed to elkhorn coral’s threatened listing: (1) high vulnerability to ocean warming, ocean acidification and disease; (2) high vulnerability to sedimentation and elevated nutrient levels; (3) uncommon abundance; (4) decreasing trend in abundance; (5) low relative recruitment rate; (6) restricted geographic range; (7) concentrated in the Caribbean; and (8) inadequacy of regulatory mechanisms. The latest 5-year review continued to support these conclusions (National Marine Fisheries Service, 2022j).

F.3.2.2 Staghorn Coral (*Acropora cervicornis*)

F.3.2.2.1 Status and Management

Staghorn coral is designated as a threatened species under the ESA. Staghorn coral shares the four areas of designated critical habitat with elkhorn coral, as well as the two exemptions at Navy facilities (refer to Section F.3.2.1.1, Status and Management, for information on critical habitat for these two species). Exemptions from critical habitat designations include a small zone around Naval Air Station Key West and a small area within the South Florida Ocean Measurement Facility Testing Range. The exemption for Naval Air Station Key West was granted in accordance with a provision of the National Defense

Authorization Act that allows such exemptions for installations with approved Integrated Natural Resources Management Plans. The exemption for the South Florida Ocean Measurement Facility was granted for national security reasons (73 *Federal Register* 72210).

A 5-year review was completed by NMFS for this species in January 2022 (National Marine Fisheries Service, 2022j).

F.3.2.2.2 Habitat and Geographic Range

Staghorn coral is commonly found in lagoons and the upper to mid-reef slopes, at depths of 1 to 20 m, and requires a salinity range of 34 to 37 parts per thousand (ppt) (Aronson et al., 2008a; Boulon et al., 2005) (refer to Section F.3.2.1.2, Habitat and Geographic Range, as habitat information provided for elkhorn coral applies to staghorn coral as well).

In the Study Area, staghorn distribution extends south from Palm Beach, Florida and along the east coast to the Florida Keys and Dry Tortugas (Jaap, 1984), in the southern part of the Gulf of Mexico Large Marine Ecosystem, the northern part of the Caribbean Sea Large Marine Ecosystem, and the southern part of the Southeast U.S. Continental Shelf Large Marine Ecosystem. Staghorn coral is known to occur in portions of the Key West Range Complex (77 *Federal Register* 73219). Potential habitat for the species is depicted in Figure F.3-1.

F.3.2.2.3 Population Trends

Most population monitoring of shallow-water corals is focused on the Florida Keys, which straddle three large marine ecosystems: Southeast U.S. Continental Shelf, Caribbean Sea, and Gulf of Mexico. Because the Florida Keys comprise their own ecological subregion, most reports categorize coral data as Floridian versus Caribbean rather than distinguishing populations on one side of these artificial boundaries. Research on the population status of staghorn coral indicates a drastic decline throughout the Caribbean that peaked in the 1980s. At four monitored reefs in the Florida Keys, staghorn coral cover decreased as follows:

- 18 percent on Carysfort Reef (1974 to 1982) (Dustan & Halas, 1987)
- 96 percent on Molasses Reef (1981 to 1986) (Jaap et al., 1988)
- 80 to 98 percent in the Dry Tortugas (Davis, 1982)
- 98 percent on Looe Key (1983 to 2000) (Causey et al., 2002)

Continued long-term monitoring in the Florida Keys and the U.S. Virgin Islands has found that staghorn coral remains at 2 percent or less of all corals on reefs, a fraction of its former abundance (Boulon et al., 2005; Rothenberger et al., 2008). Staghorn coral grown in “nurseries” to assist recovery programs had substantially higher survival rates after a catastrophic cold-water bleaching event in 2010, suggesting that restoration projects have potential for success (Schopmeyer et al., 2011). This same 2010 cold-water event killed an average of 15 percent of staghorn colonies at monitored reefs in the Florida Keys, a substantial decline in this remnant population (Lirman et al., 2011; National Marine Fisheries Service, 2012). Since the 2006 decision to list staghorn coral as threatened, some populations have continued to decline by 50 percent or more, and reliance on asexual fragmentation as a source of new colonies is not considered sufficient to prevent extinction (77 *Federal Register* 73219). The latest 5-year review reported further declines in the Florida Keys population based on survey results from 2014 to 2020 (National Marine Fisheries Service, 2022j).

Growth rates for this species range from approximately 1 to 5 in. per year (Boulon et al., 2005). Reproductive strategies and characteristics are not materially different from elkhorn coral (Section F.3.2.1.3, Population Trends).

F.3.2.2.4 Predator and Prey Interactions

Predators of corals include sea stars, snails, and fishes (e.g., parrotfish and damselfish) (Boulon et al., 2005; Roff et al., 2011). The marine snail, *Coralliophila abbreviata* (Grober-Dunsmore et al., 2006), and the bearded fireworm, are the primary predators on staghorn coral. Staghorn coral feeding strategies and symbioses are not materially different than those described for elkhorn coral (Section F.3.2.1.4, Predator and Prey Interactions).

F.3.2.2.5 Species-Specific Threats

Staghorn coral has no species-specific threats. It is susceptible to the same suite of stressors that generally threaten corals (Section F.3.2.1.5, Species-Specific Threats). However, it is more susceptible to disease such as white band disease (Patterson et al., 2002; Porter et al., 2001), even though other diseases can also impact staghorn coral survival (National Marine Fisheries Service, 2015b). A white band type II disease that is linked with the bacterial infection, *Vibrio carchariae*, also referred to as *V. charchariae* or *V. harveyi* (Gil-Agudelo et al., 2006), has also been described. A transmissible disease that caused rapid tissue loss in staghorn corals in the Florida Keys was described in 2003 (Williams & Miller, 2005). Similar to white pox in *A. palmata*, the disease manifested with irregular multifocal tissue lesions with apparently healthy tissue remaining in between. Ciliate infections have also been documented at several locations in the Caribbean (Croquer et al., 2006).

NMFS evaluated the population’s demographic, spatial structure, and vulnerability factors to determine whether the species was likely to have an “…extremely high risk of extinction with little chance for recovery…” by 2100 (Brainard et al., 2011). Elements that contribute to staghorn coral’s threatened status include high vulnerability to ocean warming, ocean acidification and disease, sedimentation and elevated nutrient levels, low abundance, decreasing trend in abundance, low relative recruitment rate, restricted geographic range, and inadequacy of regulatory mechanisms. The latest 5-year review continued to support these conclusions (National Marine Fisheries Service, 2022j).

F.3.2.3 Lobed Star Coral (*Orbicella annularis*)

F.3.2.3.1 Status and Management

Lobed star coral (*Orbicella* [formerly *Montastraea*] *annularis*) is listed as threatened under the ESA. *Orbicella annularis*, boulder star coral (*Orbicella franksi*) and mountainous star coral (*Orbicella faveolata*) have partially overlapping morphological characteristics, particularly in northern sections of their range, making identification less certain than for most other Caribbean corals. While there now is reasonable acceptance that these are three separate and valid species, decades of taxonomic uncertainty and difficult field identification have led many to consider these a single species complex. Consequently, many long-term monitoring data sets and previous ecological studies did not distinguish among the three species, instead pooling them together as “*M. annularis* complex” or “*M. annularis* sensu lato” (Brainard et al., 2011; Jaap et al., 2002; National Marine Fisheries Service, 2012; Somerfield et al., 2008).

There is now critical habitat designated for this species (88 *Federal Register* 54026, August 9, 2023) that expands on the critical habitat depicted for elkhorn coral as shown in Figure 3.5-1 (Critical Habitat for Elkhorn and Staghorn Coral and Five ESA-Listed Coral Species in the Study Area).

A 5-year review was completed by NMFS for this species in January 2022 (National Marine Fisheries Service, 2022j).

F.3.2.3.2 Habitat and Geographic Range

Lobed star coral has been reported from depths of 0.5 to 20 m (2 to 66 ft.) (Brainard et al., 2011; National Marine Fisheries Service, 2012). *Orbicella* species, including lobed star coral, occur in most reef habitat types, although less commonly on the reef flat and in the shallow zones formerly dominated by elkhorn coral (Brainard et al., 2011; Goreau, 1959; National Marine Fisheries Service, 2012). *Orbicella* species are key reef-builders. They are known throughout the Caribbean, Bahamas, and the Flower Garden Banks, but are uncommon or possibly absent from Bermuda.

In the Study Area, lobed star coral is typically found in the southern and southeastern parts of the Gulf of Mexico Large Marine Ecosystem, the northern part of the Caribbean Sea Large Marine Ecosystem, and the southern part of the Southeast U.S. Continental Shelf Large Marine Ecosystem. Lobed star coral range includes most portions of the Study Area where shallow-water coral reefs occur. The principal areas of coincidence between lobed star coral habitat and the Study Area are near Puerto Rico and south Florida. Lobed star coral is known to occur in the South Florida Ocean Measurement Facility Testing Range, adjacent to the Naval Air Station Key West, and the Key West Range Complex. However, some of this geographic range information is based on ecological studies that identified the *O. annularis* complex rather than specifying *O. annularis* in particular. Potential habitat for the species is depicted in Figure F.3-1.

F.3.2.3.3 Population Trends

Lobed star coral in the U.S. Virgin Islands declined 72 percent during the years from 1988 to 1999 (Edmunds & Elahi, 2007). Declines between 40 and 60 percent were recorded in Puerto Rico, and 80 to 95 percent declines were observed in Florida between the late 1970s and 2003 (Aronson et al., 2008c; Brainard et al., 2011). However, because many studies in Puerto Rico and Florida did not reliably distinguish between the three species, these changes in abundance should be assumed to apply generally to the *O. annularis* species complex (Brainard et al., 2011). In addition to these declines, the remnant population of *O. annularis* in the Florida Keys was decimated by the 2010 cold-water bleaching event that killed about 56 percent of all *O. annularis* colonies at monitored reefs (Lirman et al., 2011). The recent 5-year review (National Marine Fisheries Service, 2022j) reported on trends in *O. annularis* abundance at permanent monitoring sites in southeast Florida, the Florida Keys+, and the Dry Tortugas. The results indicated a substantial decline between 2014 and 2015 followed by a gradual decline until 2019. In terms of density in these locations, *O. annularis* decreased from approximately 0.04 colonies per square meter in 2014 to about 0.02 colonies per square meter in 2019. The mean colony size was reported as between 25 and 50 cm (10 to 20 in.).

All three of the *O. annularis* complex species are hermaphroditic, spawning over six to eight nights following the new moon in late summer (late August to early October) (Brainard et al., 2011). Buoyant gametes are fertilized at the surface. Fertilization success is low and recruitment rates are considered to be extremely low. For example, one study found only a single *O. annularis* recruit over 16 years of observation of 12 square meters of reef in Discovery Bay, Jamaica (Hughes & Tanner, 2000). Asexual reproduction by fragmentation is occasionally successful, but in general, reproduction rates of this species are extremely low (Aronson et al., 2008c; Brainard et al., 2011). Genetic studies of boulder star coral found that populations in the eastern and western Caribbean are relatively genetically distinct, suggesting that regional differences in population trends or regulations for corals may influence their populations' genetic diversity (Foster et al., 2012).

Growth rates are approximately 1 cm per year for colonies at depths of less than 12 m and growth rates decrease sharply as depth increases (Brainard et al., 2011). Slow growth coupled with low recruitment rates contribute to the three *O. annularis* complex species' vulnerability to extinction (Brainard et al., 2011).

F.3.2.3.4 Predator and Prey Interactions

Lobed star coral is much less susceptible to predation by snails than the *Acropora* species, and although preyed upon by parrotfish, the species is not targeted (Brainard et al., 2011; Roff et al., 2011). Lobed star coral, as well as other species of *Orbicella*, is susceptible to yellow band disease (Closek et al., 2014). Yellow band disease progresses slowly but can cause large die-offs over the course of several seasons. The disease is known to affect several other types of coral and is pervasive in the Caribbean (Closek et al., 2014). Lobed star coral feeding strategies and symbioses are not materially different than those described for elkhorn coral (Section F.3.2.1.4, Predator and Prey Interactions).

F.3.2.3.5 Species-Specific Threats

All three species of the *O. annularis* complex are highly susceptible to thermal bleaching, both warm and cool extremes (Brainard et al., 2011; National Marine Fisheries Service, 2012). Recently, lobed star coral and mountainous star coral (*O. faveolata*) were found to have higher susceptibility to coral bleaching than many other species (van Hooidonk et al., 2012). Among the 25 coral species assessed after a 2010 cold-water bleaching event in Florida, *O. annularis* was the most susceptible to mortality by a factor of almost two (Lirman et al., 2011). Otherwise, this coral has no species-specific threats, and is susceptible to the same suite of stressors that generally threaten corals (Section F.3.1.4, General Threats). Disease and pollution (e.g., nutrients, herbicides, and pesticides) are the most damaging of the general threats (Brainard et al., 2011; Hughes et al., 2003; Pandolfi et al., 2005).

NMFS evaluated the population’s demographic, spatial structure, and vulnerability factors to determine whether the species was likely to have an, “…extremely high risk of extinction with little chance for recovery…,” by 2100 (Brainard et al., 2011). Elements that contribute to lobed star coral’s threatened status are: susceptibility to ocean temperature shifts, disease, sedimentation, elevated nutrient levels, and ocean acidification; susceptibility to trophic effects of fishing; inadequate existing regulatory mechanisms to address global threats; threats by human impacts; decreasing trend in abundance; low relative recruitment rate; narrow overall distribution (based on narrow geographic distribution and moderate depth distribution); the concentration of the species in the Caribbean; and shifts to small size classes via fission and partial mortality of older, larger colonies (79 *Federal Register* 53852). The latest 5-year review continued to support these conclusions (National Marine Fisheries Service, 2022j).

F.3.2.4 Boulder Star Coral (*Orbicella franksi*)

F.3.2.4.1 Status and Management

Boulder star coral is designated as a threatened species under the ESA. This species, previously identified as *Montastraea franksi*, is part of the *O. annularis* complex, which also includes lobed star coral and mountainous star coral.

There is now critical habitat designated for this species (88 *Federal Register* 54026) that expands on the critical habitat depicted for elkhorn coral as shown in Figure 3.5-1 (Critical Habitat for Elkhorn and Staghorn Coral and Five ESA-Listed Coral Species in the Study Area).

A 5-year review was completed by NMFS for this species in January 2022 (National Marine Fisheries Service, 2022j).

F.3.2.4.2 Habitat and Geographic Range

Boulder star coral is found at least as deep as 50 m (164 ft.) (Brainard et al., 2011), and is found in most reef environments. The *O. annularis* complex has been reported to at least 70 to 90 m (230 to 295 ft.), though only *O. faveolata* and *O. franksi* are likely to occur at these depths.

Boulder star corals are typically found in the southern part of the Southeast U.S. Continental Shelf Large Marine Ecosystem (southeast Florida from Lake Worth Inlet in Palm Beach County to the Dry Tortugas; Figure F.3-1, Map A), the southeastern and northwestern regions of the Gulf of Mexico Large Marine Ecosystem (e.g., Flower Garden Banks; Figure F.3-1, Map B), and the northern part of the Caribbean Sea Large Marine Ecosystem (e.g., Puerto Rico, U.S. Virgin Islands; Figure F.3-1, Map C). Boulder star coral is known to occur in the South Florida Ocean Measurement Facility Testing Range, adjacent to Naval Air Station Key West, and the Key West and Gulf of Mexico Range Complexes. The latest 5-year review reported that this species has not been observed on the Florida Reef Tract at depths greater than 40 m (National Marine Fisheries Service, 2022j). However, the species has been observed on more banks in the Flower Garden Banks National Marine Sanctuary (Figure F.3-1). The species is also found in Bermuda but otherwise its geographic range is not materially different from *O. annularis*. However, some of this geographic range information is based on ecological studies that identified the *O. annularis* complex rather than specifying *O. franksi* in particular.

F.3.2.4.3 Population Trends

This species information is assumed not to be materially different from lobed star coral; however, differences may be masked since many ecological studies collected data at the *O. annularis* complex level rather than specifying *O. franksi* in particular.

F.3.2.4.4 Predator and Prey Interactions

This species information is assumed not to be materially different from lobed star coral; however, differences may be masked since many ecological studies collected data at the *O. annularis* complex level rather than specifying *O. franksi* in particular.

F.3.2.4.5 Species-Specific Threats

Boulder star coral was less susceptible to mortality after a 2010 cold-water bleaching event in Florida than any of its congeners (different species of the same genus) by at least a factor of three (Lirman et al., 2011). Otherwise, susceptibility to threats is not assumed to be materially different from lobed star coral. However, differences may be masked because many ecological studies identified the *O. annularis* complex rather than specifying *O. franksi* in particular.

NMFS evaluated the population's demographic, spatial structure, and vulnerability factors to determine whether the species was likely to have an "…extremely high risk of extinction with little chance for recovery…" by 2100 (Brainard et al., 2011). Elements that contribute to boulder star coral's threatened status are: high susceptibility to ocean warming, disease, elevated nutrient levels, ocean acidification, and sedimentation, susceptibility to trophic effects of fishing, inadequate existing regulatory mechanisms to address global threats, decreasing trend in abundance, slow growth rate, low relative recruitment rate, moderate overall distribution (based on narrow geographic distribution and wide depth distribution), restriction to the Caribbean, and shifts to small size classes via fission and partial mortality of older, larger colonies (79 *Federal Register* 53852). The latest 5-year review continued to support these conclusions (National Marine Fisheries Service, 2022j).

F.3.2.5 Mountainous Star Coral (*Orbicella faveolata*)

F.3.2.5.1 Status and Management

Mountainous star coral is designated as a threatened species under the ESA. The species was previously identified as *Montastraea faveolata*. Mountainous star coral is part of the *O. annularis* complex, which also includes lobed star coral and boulder star coral.

There is now critical habitat designated for this species (88 *Federal Register* 54026, August 9, 2023) that expands on the critical habitat depicted for elkhorn coral as shown in Figure 3.5-1 (Critical Habitat for Elkhorn and Staghorn Coral and Five ESA-Listed Coral Species in the Study Area).

A 5-year review was completed by NMFS for this species in January 2022 (National Marine Fisheries Service, 2022j).

F.3.2.5.2 Habitat and Geographic Range

Mountainous star coral occurs within depths from 0.5 m (2 ft.) to at least 40 m (131 ft.) (Brainard et al., 2011), and like *O. annularis* it is more commonly found in the shallower portions of this depth range. The *O. annularis* complex has been reported to at least 70 to 90 m (230 to 295 ft.), though only *O. faveolata* and *O. franksi* are likely to occur at these depths. The latest 5-year review reported that this species has not been observed on the Florida Reef Tract on seafloor greater than 44 m deep (National Marine Fisheries Service, 2022j). However, the species has been observed on more banks in the Flower Garden Banks National Marine Sanctuary (Figure F.3-1). This species is also found in Bermuda but otherwise its geographic range is not materially different from *O. annularis*.

Mountainous star coral is known to occur in the South Florida Ocean Measurement Facility Testing Range and the Key West Range Complex (both offshore and inshore locations). Three colonies were found growing on the seawalls in Truman Harbor (HDR Environmental Operations and Construction Inc., 2013), which is an inshore location of the Key West Range Complex. However, some of this geographic range information is based on ecological studies that identified the *O. annularis* complex rather than specifying *O. faveolata* in particular. Potential habitat for the species is depicted in Figure F.3-1.

F.3.2.5.3 Population Trends

This species information is assumed not to be materially different from lobed star coral; however, differences may be masked since many ecological studies collected data at the *O. annularis* complex level rather than specifying *O. faveolata* in particular.

F.3.2.5.4 Predator and Prey Interactions

This species information is assumed not to be materially different from lobed star coral; however, differences may be masked since many ecological studies collected data at the *O. annularis* complex level rather than specifying *O. faveolata* in particular.

F.3.2.5.5 Species-Specific Threats

This species information is assumed not to be materially different from lobed star coral; however, differences may be masked since many ecological studies collected data at the *O. annularis* complex level rather than specifying *O. faveolata* in particular.

NMFS evaluated the population’s demographic, spatial structure, and vulnerability factors to determine whether the species was likely to have an “…extremely high risk of extinction with little chance for recovery…” by 2100 (Brainard et al., 2011). Elements that contribute to mountainous star coral’s threatened status are: high susceptibility to ocean warming, disease, sedimentation and elevated nutrient levels, susceptibility to trophic effects of fishing, inadequate existing regulatory mechanisms to address global threats, decreasing trend in abundance, low relative recruitment rate, late reproductive maturity, moderate overall distribution with concentration in areas of high human impact, and shifts to small size classes via fission and partial mortality of older, larger colonies (79 *Federal Register* 53852). The latest 5-year review continued to support these conclusions (National Marine Fisheries Service, 2022j).

F.3.2.6 Pillar Coral (*Dendrogyra cylindrus*)

F.3.2.6.1 Status and Management

Pillar coral is currently designated as a threatened species under the ESA. However, reclassification of pillar coral (*Dendrogyra cylindrus*) from threatened to endangered was proposed on August 29, 2023 (88 *Federal Register* 59494). The action was based on population declines and susceptibility to a recently emerged coral disease.

There is now critical habitat designated for this species (88 *Federal Register* 54026, August 9, 2023) that expands on the critical habitat depicted for elkhorn coral as shown in Figure 3.5-1 (Critical Habitat for Elkhorn and Staghorn Coral and Five ESA-Listed Coral Species in the Study Area).

A 5-year review was completed by NMFS for this species in January 2022 (National Marine Fisheries Service, 2022j).

F.3.2.6.2 Habitat and Geographic Range

Pillar coral most frequently occurs at depths of 3 to 8 m (10 to 26 ft.) but has been documented at depths of 1 to 25 m (3 to 82 ft.) (Brainard et al., 2011; National Marine Fisheries Service, 2012). It is found on rocky outcrops in areas of high wave activity (Marhaver et al., 2015). It is known to occur in south Florida as far north as Broward County and from one colony in Bermuda, but is not known to occur at the Flower Garden Banks or elsewhere in the northern or western Gulf of Mexico.

In the Study Area, pillar coral is typically found in the southern and southeastern parts of the Gulf of Mexico Large Marine Ecosystem, the northern part of the Caribbean Sea Large Marine Ecosystem, and the southern part of the Southeast U.S. Continental Shelf Large Marine Ecosystem. Pillar coral range includes most portions of the Study Area where shallow-water coral reefs occur. The principal areas of coincidence between pillar coral habitat and the Study Area are near Puerto Rico and south Florida. Pillar coral is known to occur in portions of the South Florida Ocean Measurement Facility Testing Range, adjacent to the Naval Air Station Key West, and in the Key West Range Complex. Potential habitat for the species is depicted in Figure F.3-1.

F.3.2.6.3 Population Trends

Pillar coral is both rare and conspicuous (due to its growth form). It has a limited habitat preference and colonies are often dispersed and isolated throughout the habitat range (79 *Federal Register* 53852). Because pillar coral colonies have been killed by warm- and cold-water bleaching, disease, and physical damage, it has been assumed that this rare species is in decline. In general, pillar coral is too rare for meaningful trends in abundance to be detected by typical reef-monitoring programs (Brainard et al., 2011). However, recent studies on reproductive strategies and life history have shown low sexual recruitment rates and slow growth, adding further population and genetic diversity concerns for the species (Marhaver et al., 2015).

Growth rates for this species are typically 8 millimeters (mm) (0.3 in.) per year, though rates up to 20 mm (0.8 in.) per year have been reported (Brainard et al., 2011). Piller coral were first recorded spawning in August 2012, they are known to spawn 3 to 4 days after a full moon. The rate of sexual reproduction is likely to be low because the species is so rare and colonies are gonochoric (i.e., a colony is either male or female); male and female colonies are unlikely to be in close enough proximity for reliable fertilization. For this reason, no juveniles of pillar coral have been observed in the past several decades, and fragmentation seems to be the only successful mode of reproduction for this species

(National Marine Fisheries Service, 2012). The latest 5-year review reported further declines in the Florida Keys population based on recent survey results (National Marine Fisheries Service, 2022j).

F.3.2.6.4 Predator and Prey Interactions

Predators of this species seem to be few, and though the corallivorous fireworm (*Hermodice carunculata*) feeds on diseased pillar coral, it does not seem to be a major predator (Brainard et al., 2011). A species of sea urchin (*Diadema antillarum*) has been known to cause partial mortality at the base of pillar coral colonies (79 *Federal Register* 53852). Pillar coral is distinctive among Caribbean corals because its tentacles are extended for feeding on zooplankton during the day, while most other corals' tentacles are retracted during the day (Boulon et al., 2005; Brainard et al., 2011). Pillar coral feeding strategies and symbioses are not materially different than those described for elkhorn coral (Section F.3.2.1.4, Predator and Prey Interactions).

F.3.2.6.5 Species-Specific Threats

Pillar coral has no species-specific threats. It is susceptible to the same suite of stressors that generally threaten corals (Section F.3.1.4, General Threats); however, it was historically more susceptible to exploitation by the curio trade (Brainard et al., 2011). Low population density and separation of male and female colonies are the principal threats to the species (Brainard et al., 2011; National Marine Fisheries Service, 2012).

NMFS evaluated the population's demographic, spatial structure, and vulnerability factors to determine whether the species was likely to have an "…extremely high risk of extinction with little chance for recovery…" by 2100 (Brainard et al., 2011). Elements that contribute to pillar coral's threatened status are susceptibility to ocean warming, disease, acidification, elevated nutrient levels, sedimentation, trophic effects of fishing, inadequate existing regulatory mechanisms to address global threats, rare general range-wide abundance, low relative recruitment rate, narrow overall distribution (based on narrow geographic distribution and moderate depth distribution), and restriction to the Caribbean (79 *Federal Register* 53852). The latest 5-year review continued to support these conclusions (National Marine Fisheries Service, 2022j).

F.3.2.7 Rough Cactus Coral (*Mycetophyllia ferox*)

F.3.2.7.1 Status and Management

Rough cactus coral is designated as a threatened species under the ESA.

There is now critical habitat designated for this species (88 Federal Register 54026, August 9, 2023) that expands on the critical habitat depicted for elkhorn coral as shown in Figure 3.5-1 (Critical Habitat for Elkhorn and Staghorn Coral and Five ESA-Listed Coral Species in the Study Area).

A 5-year review was completed by NMFS for this species in January 2022 (National Marine Fisheries Service, 2022j).

F.3.2.7.2 Habitat and Geographic Range

Rough cactus coral is known to occur as deep as 80 to 90 m (Brainard et al., 2011; National Marine Fisheries Service, 2012). Though reported to commonly occur at depths of 5 to 30 m (Aronson et al., 2008d), this could be an artifact of scuba diver-based survey intensity, which decreases dramatically below 30 m. Rough cactus coral occurs in patch and fore reef (the part of the reef exposed to the open ocean) habitat types, generally in lower energy parts of the reef (Brainard et al., 2011; National Marine Fisheries Service, 2012). It is known to occur throughout the Caribbean and southern Gulf of Mexico, but is absent from the Flower Garden Banks, Bermuda, and the southeast United States north of south Florida (79 *Federal Register* 53852).

In the Study Area, rough cactus coral is typically found in the southern and southeastern parts of the Gulf of Mexico Large Marine Ecosystem, the northern part of the Caribbean Sea Large Marine Ecosystem, and the southern part of the Southeast U.S. Continental Shelf Large Marine Ecosystem. Rough cactus coral range includes most portions of the Study Area where shallow-water coral reefs occur (e.g., Pulley Ridge). However, the available surveys on Pulley Ridge (minimum depth of 60 m) have not documented any ESA-listed coral species (Halley et al., 2005; Reed et al., 2019). The principal areas of coincidence between rough cactus coral habitat and the Study Area are near Puerto Rico and south Florida. Rough cactus coral is known to occur in the South Florida Ocean Measurement Facility Testing Range, Key West Range Complex, and adjacent to the Naval Air Station Key West. The principal areas of coincidence between rough cactus coral habitat and the Study Area are near Puerto Rico and south Florida. Rough cactus coral is known to occur in the South Florida Ocean Measurement Facility Testing Range, adjacent to the Naval Air Station Key West, and the Key West Range Complex. Potential habitat for the species is depicted in Figure F.3-1.

F.3.2.7.3 Population Trends

Though never abundant, rough cactus coral in the Florida Keys has declined by at least 80 percent since 1996 and perhaps by much more since the 1970s (Brainard et al., 2011). The abundance of rough cactus coral has been estimated to be at least hundreds of thousands of colonies in the Florida Keys and Dry Tortugas (50 *Federal Register* 53852). The latest 5-year review reported further declines in the Florida Keys population based on recent survey results (National Marine Fisheries Service, 2022j).

Rough cactus coral is a hermaphroditic brooder, releasing fully developed larvae in the late winter (February to March) (Aronson et al., 2008d). Recruitment rates are extremely low or absent, as evidenced by observation of an anchor-damaged site in the U.S. Virgin Islands over a 10-year period (Brainard et al., 2011). No colonies of rough cactus coral were observed to recruit to the site despite the presence of adults on an adjacent reef (50 *Federal Register* 53852).

F.3.2.7.4 Predator and Prey Interactions

Rough cactus coral is not known to be particularly susceptible to predators (Brainard et al., 2011), and feeding strategies and symbioses are not materially different than those described for elkhorn coral (Section F.3.2.1.4, Predator and Prey Interactions).

F.3.2.7.5 Species-Specific Threats

Though not especially susceptible to mortality from warm-water bleaching (Brainard et al., 2011; Lough & van Oppen, 2009), 15 percent of *Mycetophyllia* species were killed after a cold-water bleaching event in Florida (Lirman et al., 2011). Some coral diseases are characterized by the white-colored bands or pox they cause but are otherwise difficult to discriminate (Porter et al., 2001). While diseases such as “white plague” do not seem to be species specific (Porter et al., 2001), rough cactus coral in the Florida Keys has been particularly susceptible to this type of disease (Brainard et al., 2011).

NMFS evaluated the population’s demographic, spatial structure, and vulnerability factors to determine whether the species was likely to have an “…extremely high risk of extinction with little chance for recovery…” by 2100 (Brainard et al., 2011). Elements that contribute to rough cactus coral’s (*Mycetophyllia ferox*) threatened status are: high susceptibility to disease; susceptibility to ocean warming, acidification, trophic effects of fishing, elevated nutrient levels, and sedimentation; inadequate existing regulatory mechanisms to address global threats; threats by human impacts; rare general range-wide abundance; decreasing trend in abundance; low relative recruitment rate; moderate overall distribution (based on narrow geographic distribution and wide depth distribution); and restriction to the Caribbean (50 *Federal Register* 53852). The latest 5-year review continued to support these conclusions (National Marine Fisheries Service, 2022j).

F.3.2.8 Queen Conch (*Alger gigas*)

F.3.2.8.1 Status and Management

On September 7, 2022, NMFS announced a proposed rule to list the queen conch as a threatened species under the ESA throughout its range (Horn et al., 2022). The decision was made after a status review and after considering the efforts being made to protect the species. The proposal followed a lawsuit filed by WildEarth Guardians and Friends of Animals in 2016. The species was subsequently listed as threatened on March 15, 2024 (89 *Federal Register* 11208).

The queen conch may not be commercially or recreationally harvested in Florida waters per state law. In the Caribbean, NMFS and the Caribbean Fishery Management Council manage queen conch in federal waters, while the governments of Puerto Rico and the U.S. Virgin Islands manage queen conch in their territorial waters.

There is no critical habitat designated or currently proposed for this species.

F.3.2.8.2 Habitat and Geographic Range

The distribution of the queen conch in the Study Area includes South Florida, Puerto Rico, Bermuda, and the Flower Garden Banks in the Gulf of Mexico (Theile, 2001). Within this area, the species inhabits a wide range of habitats including seagrass beds, sand flats, algal beds, and coral reef areas, from a few inches deep to approximately 30 m (Brownell & Stevely, 1981). Juveniles are associated specifically with native seagrass beds (Boman et al., 2019), and adults spawn habitually in sandy areas less than 20 m deep, with rubble or otherwise hard substrate nearby (Glazer & Kidney, 2004; Stoner & Sandt, 1992). Of particular interest in the Study Area are the relatively high densities of adult queen conch reported on mesophotic reefs (38 to 44 m in depth) around Puerto Rico (García-Sais et al., 2012); mesophotic reefs in this depth range are also found along the margins of the Florida Reef Tract that intersects the Key West Range complex (Figure F.3-1).

F.3.2.8.3 Population Trends

The majority of available density estimates from various jurisdictions suggest the queen conch population is below the minimum threshold necessary to maintain their viability as a species. Of the estimated 743 million individuals remaining throughout the species range, 95 percent reside in jurisdictions outside of the Study Area (Horn et al., 2022). The current densities and population trends for queen conch in Study Area locations are relatively low (15.07 to 17.29 adults/acre) but increasing. Unfished populations in South Florida were increasing (1990–2020), and fished populations around Puerto Rico (1985–2015) and the U.S. Virgin Islands (1975–2010) were increasing slightly. As an associated measure, the area of seagrass habitat in the Study Area has declined (or increased, depending on location) only slightly compared to other areas in the queen conch's distribution (Waycott et al., 2009).

F.3.2.8.4 Predator and Prey Interactions

The early life stages (e.g., juveniles) of queen conch are the most vulnerable to predators, including stingrays, spiny lobster, octopus, nurse sharks, hermit crabs, predaceous snails, and other predators (Horn et al., 2022). Mortality due to predation decreases as a conch grows larger and more resilient (Iversen et al., 1986). Juvenile conch bury themselves in dense aggregations that predators find difficult to dislodge. The hard shell and nocturnal activity of the adult conch reduces the risk of predation from tulip snails, apple murex, common octopus, spiny lobsters, queen triggerfish, spotted eagle rays, a variety of hermit crabs, sea turtles, and sharks (Iversen et al., 1986; Jory & Iversen, 1983; Stoner & Ray, 1993). Only the larger predators can penetrate the thick shell of an older conch. Predation is not believed to currently be a factor influencing the status of the queen conch.

The species forages primarily on seagrass detritus and red/green macroalgae on shallow-water coral reefs and other hard bottom areas (Randall, 1964; Serviere-Zaragoza et al., 2009; Stoner & Sandt, 1992; Stoner & Waite, 1991). The location of seagrass beds and shallow-water coral reefs is depicted in Figure F.3-1, and overlaps only the Key West Range Complex, among the named training and testing areas.

F.3.2.8.5 Species-Specific Threats

The most significant threat to queen conch is a combination of overuse by commercial fisheries and insufficient regulatory protections (Horn et al., 2022). Many jurisdictions have regulations that are inadequately enforced or implement regulations that use inappropriate metrics for managing the queen conch fisheries (e.g., allowing for the legal harvest of juvenile conch). The species populations continue to decline and the few jurisdictions with adequate regulations and enforcement have yet to report any substantial recovery.

Other threats include decline in native seagrass coverage, environmental contaminants, oil spills, and microplastic pollution (Horn et al., 2022). However, the best available information indicates that these factors are not threatening the species persistence. The Caribbean region is also being impacted by climate change, where direct and indirect effects (e.g., ocean warming/acidification on shell formation and shifting habitat locations, respectively) are only beginning to be realized.

F.3.3 SPECIES NOT LISTED UNDER THE ENDANGERED SPECIES ACT

F.3.3.1 Corals, Hydroids, Jellyfish (Phylum Cnidaria)

F.3.3.1.1 Shallow-Water Hard Corals

Researchers compiled historical and recent information on the amount of hard reef structure covered by living corals at 90 reef locations in the wider Caribbean Sea (primarily shallow reefs in water depths of 1 to 20 m) (Jackson et al., 2014). Average coral coverage on the hard reef structure is estimated to be approximately 14 to 17 percent, down from approximately 35 percent during the period of 1970 to 1983. Shallow-water coral cover on the Flower Garden Banks was reportedly much higher than any area in southern Florida. The coverage estimates for living coral on high and low-relief areas (mostly hard substrate) of the Flower Garden Banks was 54 to 59 and 19 to 32 percent, respectively (Clark et al., 2014). The coverage of live-hard corals was also lower in the upper mesophotic zone than the shallow-water zone. Shallow-water coral reefs have also been documented south of the Tortugas at 30 m to greater than 50 m depths on a feature called “Riley’s Hump” (Weaver et al., 2006). The coverage of living hard corals on Pulley Ridge features averaged 1.5 percent, but there were no ESA-listed coral species observed (Reed et al., 2019). Pulley Ridge is considered the deepest shallow-water/mesophotic coral reef in the Study Area (Reed et al., 2019). Shallow-water coral reefs may contain ESA-listed coral species, and changes in overall coral coverage provides context for subsequent discussion of these species in Section F.3.2 (Endangered Species Act-Listed Species).

Hard, reef-forming coral species make up a small portion of the total coverage by living organisms on shallow-water coral reefs. For example, the coverage of macroalgae and sessile animals on Pulley Ridge features was 47 and 60 percent, respectively (Reed et al., 2019). Remote operated vehicle surveys of the Tortugas South Ecological Reserve in similar depths of water (Miller’s Ledge) suggested a much lower coverage of biota (5 percent), and very low (0.01 percent) coverage of hard corals (Reed et al., 2017; Weaver et al., 2006). Miller’s Ledge comes very close to overlapping the Key West operating area (OPAREA), although similar habitats at depth intersect the northern edge of the OPAREA. However, the shallow-water coral reef areas in the Florida Reef Tract (with the exception of Pulley Ridge) and the northwest Gulf of Mexico is encompassed by either the Florida Keys or Flower Garden Banks National

Marine Sanctuaries, respectively, where management rules prohibit impacting sensitive bottom habitats (refer to Section F.3.2, Endangered Species Act-Listed Species, for mapping).

The following information was updated from the 2018 Final EIS/OEIS:

- In 2016, a mass die-off of corals and other invertebrates (e.g., sponges, urchins, brittle stars, and clams) was documented in the Flower Garden Banks National Marine Sanctuary in the Gulf of Mexico (National Oceanic and Atmospheric Administration, 2016a, 2016b). The die-off was most likely caused by localized hypoxia produced by unusually high freshwater discharge from the Mississippi/Atchafalaya rivers (Kealoha et al., 2020). Coral disease has become another pressing issue affecting coral reef survival over the last 50 years. A large disease outbreak was documented in numerous coral species off southeastern Florida in 2014 (Precht et al., 2016). Reefs throughout the Caribbean have also been severely impacted by increased frequency and intensity of disease outbreaks leading to coral death. A recent example of this is stony coral tissue loss disease, which has quickly spread throughout the Caribbean, devastating coral reef ecosystems (Traylor-Knowles et al., 2022).
- In general, the less-studied mesophotic coral ecosystems of the South Florida area (Figure F.3-1) appear to be in better condition than shallow-water reefs in the region (Reed et al., 2021). Importantly, no stony coral tissue loss disease was observed in any mesophotic reef surveyed. Overall, the benthic diversity of the South Florida mesophotic reefs was high compared to shallow reefs, and the mesophotic ecosystems presented very different signature species (including only one of the ESA-listed coral species). However, the occurrence of ESA-listed coral species documented in Reed et al. (2021) did not coincide with the survey locations intersecting the Key West Range Complex.

F.3.3.1.2 Deep-Water Hard Corals

The following information was updated from the 2018 Final EIS/OEIS:

- Perhaps the largest continuous area of deep-water coral on Earth occurs on the Stetson-Miami Terrace area of Blake Plateau off the U.S. Southeast Atlantic coast at 400 m to greater than 900 m depths (Sowers, 2020); an area that also coincides with the Jacksonville Range Complex (refer to Section 3.3, Habitats, Figure 3.3-3, Artificial Reef Areas and Bottom Habitats in the Southeast Region of the Study Area, for mapping). Sowers (2020) used a mosaic of multibeam sonar mapping, topographic modeling, and validation work (using remote operated vehicles) to map coral habitats on this expansive feature. The resulting analysis suggests a very low percentage of living hard coral (1 to 7 percent) on topographic features that are likely coral habitats (e.g., peaks, ridges, slopes). Based on the validation work, those topographic features were mostly coral rubble and stands of dead coral. Communities of *L. pertusa* have also been found to inhabit substrate at relatively shallow depths of 180 to 250 m off the coast of northeastern Florida in the Jacksonville Range Complex (Ross et al., 2015a; U.S. Department of the Navy, 2010).

- Submersible and remote operated vehicle surveys on the west Florida slope in the eastern Gulf of Mexico estimated 37 percent of intermediate-hard substrate featured living *L. pertusa* or some other hard coral species (Ross et al., 2017). The Ross et al. (2017) survey covered a total of 56.76 km of bottom between the Key West Range Complex and the Naval Surface Warfare Center Panama City Testing Range in depths ranging from about 400 to 600 m (refer to Section 3.3, Habitats, Figure 3.3-4, Artificial Reef Areas and Bottom Habitats in the South Florida

Region of the Study Area, for mapping). They noted that mounds less than 525 m deep were usually caped with greater than 50 percent living coral colonies.

- Deep-water corals and sponges in other large marine ecosystems in the Study Area do not coincide with any named training or testing areas, though they could be impacted by a minority of activities that may occur anywhere in the Study Area. In the Newfoundland/Labrador Shelf Large Marine Ecosystem, for example, deep-sea corals were mapped using incidental by-catch samples from stock assessment surveys and fisheries observations (Wareham & Edinger, 2007). Thirteen alcyonaceans, 2 antipatharians, 4 solitary scleractinians, and 11 pennatulaceans were recorded. Corals were broadly distributed along the continental shelf edge and slope, with most species found deeper than 200 m; only soft corals were found on the continental shelf. Deep-water coral and sponge habitats in other large marine ecosystems of the Study Area are poorly documented, based on the available spatial data for global deep-water coral collections (Freiwald et al., 2021).

The results of habitat suitability modeling of seamounts located in less than 2,500 m water depth and rising at least 1,000 m off the bottom suggest the potential for deep-water corals to occur at seamounts located off the northeast U.S. continental shelf (Tittensor et al., 2009), which is consistent with the observation of corals on Mytilus Seamount and other seamounts in the Northeast U.S. Continental Shelf Large Marine Ecosystem (Freiwald et al., 2021; Quattrini et al., 2015).

F.4 FISHES

The following topics on fishes in general were updated from the 2018 Final EIS/OEIS Section 3.6.2.1 (General Background):

- Estimated number of fish species, biomass of fishes in particular ocean zones, number of species in different zones and regions of the ocean, and schooling behavior. This information more accurately characterizes the affected environment.
- Distribution and density of fishes at the surface and on the bottom at various depths. This information strengthens the reasoning presented in the 2018 Final EIS/OEIS for the low potential for vessels, in-water devices, and military expended materials to strike fish near the surface and in deep ocean areas.
- Information on the structural composition and typical fish assemblage of hard and soft bottom habitats. This information strengthens the reasoning presented in the 2018 Final EIS/OEIS for (1) the comparatively low potential for military expended materials and seafloor devices to strike fish in soft bottom areas, and (2) the greater potential for these items to strike fish in hard bottom areas, although standard operating procedures and mitigation measures limit activities over hard bottom.
- Information on vertical migration. This information generally strengthens the reasoning presented in the 2018 Final EIS/OEIS for the low potential for vessels, in-water devices, and military expended materials to strike fish near the surface during the day, when most activities occur.
- Visual and chemosensory feeding cues used by fishes. This information confirms the conclusion presented in the 2018 Final EIS/OEIS that the potential for fish to ingest an expended item would be influenced partly by feeding method.
- Updated information on various threats to fishes. This information provides context for analysis of cumulative impacts.

The following topics on ESA-listed fish species were updated from the 2018 Final EIS/OEIS Section 3.6.2.2 (Endangered Species Act-Listed Species):

- Population status of ESA-listed fish species based on latest 5-year reviews by NMFS and, where applicable, the USFWS. This information provides updated context for impacts analysis.
- Information on ESA-listed fish species, from sources other than 5-year reviews, regarding distribution, population trends, spawning activity, predation, and threats. This information more accurately characterizes the status of ESA-listed fish species in the Study Area.
- Designation of critical habitat for one ESA-listed fish species (Nassau grouper [*Epinephelus striatus*]).
- Scientific name change for one ESA-listed fish species (giant manta ray [*Mobula birostris*]).

F.4.1 GENERAL BACKGROUND

The following information was updated from the 2018 Final EIS/OEIS Section 3.6.2.1 (General Background):

- Fishes are the most numerous and diverse vertebrate group in the Study Area. Approximately 18,000 marine and estuarine fish species occur worldwide (Fricke et al., 2023).
- Although about 78 percent of marine fish species occur in waters less than 200 m deep and in close association with land, the results of a circumglobal survey cruise indicate that the biomass of mesopelagic fish (occurring at depths of 200 to 1,000 m) is likely much greater than the biomass of fishes found elsewhere in the ocean (Irigoien et al., 2014).
- For demersal fishes, the results of a modeling effort suggest a general pattern of decreasing biomass, abundance, and size with increasing depth in the world's oceans (Wei et al., 2011).
- In surface waters of the Study Area from the Virginia Capes Range Complex to the Jacksonville Range Complex, Willmott et al. (2021) reported an average daytime density of rays, sharks, and large bony fishes (e.g., tunas, mahi-mahi, billfish, and sunfish) of 1.66 per km^2.
- A review of information available on species richness found that the number of marine fish species changes with latitude in the Study Area (Fautin et al., 2010). The greatest number is estimated for the Gulf of Mexico, followed in decreasing order by the Southeast U.S. Continental Shelf and Northeast U.S. Continental Shelf Large Marine Ecosystems.

F.4.1.1 Habitat Use

The background information for general habitat use in the Study Area as described in the 2018 Final EIS/OEIS Section 3.6.2.1.1 (Habitat Use) has not appreciably changed. As such, the information presented in the 2018 Final EIS/OEIS remains valid.

F.4.1.1.1 Shoreline Habitats

The background information for shoreline habitats in the Study Area as described in the 2018 Final EIS/OEIS Section 3.6.2.1.1 (Habitat Use) has not appreciably changed. As such, the information presented in the 2018 Final EIS/OEIS remains valid.

F.4.1.1.2 Bottom Habitats

The following information in this section was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.1 (Habitat Use).

Hard bottom habitats include rocky and live hard bottom, which support biotic communities that vary according to depth. Hard bottom habitats in relatively shallow water are generally dominated by benthic macroalgae, whereas invertebrates are typically associated with deep-water hard bottom (down to about 2,500 m in the North Atlantic). Hard substrate at depths greater than about 2,500 m may support relatively sparse populations of sessile invertebrates, but generally does not support invertebrates that form habitats potentially used by fish (e.g., hard corals and sponges). Hard bottom habitats usually support higher fish densities than the surrounding habitats (Flávio et al., 2023), although the degree of association may vary considerably (Ross et al., 2015b). A study of the Middle Atlantic Bight found that, of the 38 total fish taxa observed at all habitat types, 33 were associated with natural hard bottom and shipwreck habitat (Ross et al., 2016).

Deep-water invertebrate beds may also support greater fish occurrence than surrounding soft bottom habitat. Deep-water beds include concentrations of hard corals, soft corals, sponges, or chemosynthetic communities. Deep-water hard corals typically form mounds of substrate called bio- or litho-herms. The number of fish species observed at deep-water coral mounds off the U.S. Atlantic coast varies by location, with over 50 species documented at the Cape Lookout *Lophelia* banks (South Atlantic Fishery Management Council, 2023). A comparatively lower number of fish species have been documented in association with chemosynthetic habitats. Surveys reported 11 fish species at various hydrothermal vent fields in the Atlantic Ocean, and one species at a cold seep on the Florida Escarpment (Biscoito et al., 2002). Demersal fishes observed at cold seeps along the continental margin off the northeastern U.S. consisted of blue antimora (*Antimora rostrata*) and eel species of the genus *Synaphobranchus* (Quattrini et al., 2015). The most abundant species observed at seeps associated with Baltimore and Norfolk Canyons included the eel *Dysommina rugosa*, shortbeard codling (*Laemonema barbatulum*), longfin hake (*Phycis chesteri*), freckled tonguefish (*Symphurus nebulosus*), and lanternfish (unidentified species) (Ross et al., 2015b). Fishes documented during a study of cold seep communities in the Gulf of Mexico (offshore of Louisiana) also included eel species of the genus *Synaphobranchus*, as well as hagfish species of the genus *Eptatretus* (Cordes et al., 2009).

Soft bottom habitats are composed of various mixtures of sand, silt, and clay, as well as small particles of biological origin (e.g., the shells and skeletal remains of small animals). Most substrate in the Study Area consists of this habitat type. Soft bottom habitat usually supports a low number of fish (individuals and species) compared with structured habitats, although there is often some overlap in species occurrence. For example, of 38 total fish species documented in a study of the Middle Atlantic Bight, 33 species were exclusively associated with hard habitat but only 6 were exclusively associated with soft substrate (Ross et al., 2016). The species richness of benthic fishes generally declines with increasing depth, likely because of reduced food availability (Quattrini et al., 2015). Additionally, benthic fishes in deep ocean areas are generally considered to be widely dispersed. Surveys of the abyssal plain in the northeast Atlantic Ocean (depth of about 4,800 m) resulted in a density estimate of 723 fish per km^2 located on and near the bottom (Milligan et al., 2016). Fish distribution appeared to be random, with no aggregations noted in the survey area. Trawl surveys conducted in the Gulf of Mexico from depths of about 200 to 3,000 m collected a total of 119 demersal species (Powell et al., 2003). Abundance and species richness were highest on the upper continental slope and decreased with depth.

The following specific information was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.1 (Habitat Use):

- Mesophotic reefs, which occur at depths where there is reduced light penetration and reduced photosynthesis in algae, are known in the Study Area to depths of 50 to 90 m and support numerous fish species. For example, over 60 species have been documented at Pulley Ridge off the southwest coast of Florida (National Oceanic and Atmospheric Administration, 2023).

- Research conducted in the mid-Atlantic region of the Study Area indicates that benthic fishes tend to ignore differences in bottom type at depths greater than about 1,400 m (Ross et al., 2015b).

F.4.1.1.3 Water Column Habitats

The background information for water column habitats in the Study Area as described in the 2018 Final EIS/OEIS Section 3.6.2.1.1 (Habitat Use) has not appreciably changed. As such, the information presented in the 2018 Final EIS/OEIS remains valid.

F.4.1.2 Movement and Behavior

The following information in this section was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.2 (Movement and Behavior).

Vertical migration may be associated with foraging, predator avoidance, thermoregulation, and navigation. Vertical movement over a 24-hour period is common and is known as diel vertical migration. However, vertical movements may also occur over other time periods. The typical pattern for large epipelagic and predatory fishes is descent below the euphotic zone during the day and a rise to the surface at night, with episodic variations attributed to thermoregulation and/or vertically migrating prey (Andrzejaczek et al., 2019). The typical pattern for small pelagic fishes is similar, with episodic descents or movement to the surface at night related to echolocating predators (Urmy & Benoit-Bird, 2021). An exception to this pattern has been noted for basking sharks (*Cetorhinus maximus*) and other planktivorous sharks and rays that, in some circumstances (e.g., in nearshore environments), ascend during the day and descend at night (Andrzejaczek et al., 2021; Sims et al., 2005). This behavior is attributed to tracking zooplankton prey that undergoes reverse diel migration, thermal fronts, or thermoregulation. Some large epipelagic fishes such as sharks also undertake short-period (minutes to hours) vertical movements, which occur while the animals are also moving horizontally and may be described as oscillatory movements (Andrzejaczek et al., 2019). The purpose of this behavior is uncertain, but it may increase the probability of detecting food cues throughout the water column and decrease the energy used for swimming. Giant manta rays tagged off the coast of Peru underwent vertical oscillations during the night, presumably for thermal recovery between foraging dives to cool water (Andrzejaczek et al., 2021). This behavior has been documented in bigeye tuna (*Thunnus obesus*) and some shark species as well (Andrzejaczek et al., 2019). Conversely, some species may undertake short dives during the day to reduce body temperature. Vertical movements of some epipelagic species may coincide with the lunar cycle (Andrzejaczek et al., 2019).

The following specific information was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.2 (Movement and Behavior):

- Schooling reduces individual predation risk by creating a collective structure in which to hide (Pavlov & Kasumyan, 2000). Fish schools generally dissociate at night, likely because of the role of vision in schooling behavior.
- Some fish species rely on visual cues for feeding while others, particularly benthic species, also rely on chemosensory cues (taste) (Roch et al., 2020). Fishes that rely on visual cues are more likely to ingest non-food items (e.g., plastic particles) that visually resemble natural food than those that primarily rely on chemosensory detection.

F.4.1.3 Hearing and Vocalization

Information on hearing and vocalization in fishes is provided in Appendix D (Acoustic and Explosive Impacts Supporting Information), Section D.5.1 (Fishes, Hearing and Vocalizations).

F.4.1.4 General Threats

The following information was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.4 (General Threats):

- An assessment of cumulative human impacts on the world's ocean found a slightly increasing trend off the U.S. Atlantic and Gulf of Mexico coasts, which was substantially lower than in many other areas of the world (Halpern et al., 2019).
- There are some emerging issues that could cause both negative and positive effects on marine fishes (e.g., increasing wildfire impacts, coastal darkening, colocation of marine activities) that have yet to be studied thoroughly (Herbert-Read et al., 2022).
- Human activities may produce physical and chemical stressors on marine habitats that grow in intensity as economies grow (Danish et al., 2019). Impacts associated with these activities are not randomly distributed across the patchwork of habitat types and ecosystems; most stressors are more prevalent closer to highly developed landscapes (Halpern et al., 2008).
- Threats are often correlated to some degree. For example, a fish that is stressed by exposure to a pollutant could potentially be more susceptible to disease.
- The results of a census survey on global biodiversity indicate that about 25 percent of marine fish have become threatened or extinct since 1500 (Isbell et al., 2023). The main direct drivers of marine fish biodiversity loss were identified as overexploitation, changes in land and sea use, and climate change. The main indirect drivers were governance and human population.
- Fishing is the most common exploitative activity (World Register of Introduced Marine Species, 2023).

F.4.1.4.1 Water Quality

The following information was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.4.1 (Water Quality):

- Pollution initially impacts fishes that occur near the sources of pollution but may also affect future generations from effects to reproduction and increased mortality across life stages.
- Estimated organic and nutrient pollution trends are declining in waters of the United States (Halpern et al., 2019).
- A study of commercial coastal fishery species before and after the Deepwater Horizon oil spill, which resulted in 134 million gallons of oil being released into the northern Gulf of Mexico in 2010, found that fish abundance and diversity were unchanged, except that abundance increased the year following the spill (Schaefer et al., 2016). The increase was attributed to a temporary fishery closure.

F.4.1.4.2 Development and Human Activities

The following information was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.4.2 (Commercial and Recreational Activities), in which the corresponding subsection is titled Commercial and Recreational Activities:

- The increase in abundance of commercial coastal species in the northern Gulf of Mexico following a fishery closure associated with the Deepwater Horizon oil spill demonstrates the overwhelming influence of commercial fishing on marine fish populations (Schaefer et al., 2016).
- Barrett et al. (2019) found a substantial increase in density and diversity in wild fish near aquaculture sites. However, the long-term survival and reproduction of these fishes was not determined. Fish collected near aquaculture sites were larger and heavier with no change in body condition but had a higher risk of infection and parasitism.

Anthropogenic Noise

Information on threats to fishes from anthropogenic noise is provided in Appendix D (Acoustic and Explosive Impacts Supporting Information), Section D.5 (Fishes).

F.4.1.4.3 Disease and Parasites

The following statement was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.4.3 (Disease and Parasites):

- While the incidence of disease in some types of marine animals has increased in recent years, there is an apparently decreasing trend for marine fishes. For example, Tracy et al. (2019a) noted that global incidence of reported disease in marine fishes declined from 1970 to 2013. The reason could be declines in abundance and population density of many species because of overexploitation or bycatch.

F.4.1.4.4 Invasive Species

The following statement was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.4.4 (Invasive Species):

- Although introduction of the Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) to the Atlantic, Gulf of Mexico, and Caribbean Sea remains a potential threat to native fishes, recent studies indicate expansion in the Study Area appears to be declining (Benkwitt et al., 2017; Campbell et al., 2021), and competition with native reef species is less acute than initially thought (Smith & Côté, 2021).

F.4.1.4.5 Climate Change

The following information was updated from the 2018 Final EIS/OEIS Section 3.6.2.1.4.5 (Climate Change):

- Regarding changes in fish distribution, a recent study found that the potential for a distribution shift is substantially influenced by a species' primary habitat constituents (Roberts et al., 2020). While some pelagic species, which are strongly influenced by water temperature and salinity, have shifted distribution over the past two decades, benthic species have generally retained their historical distribution.
- The results of a modeling effort suggest that warming water temperature could increase the rate at which sinking organic material is broken down by bacteria in shallow water, decreasing the amount of food available to fishes (and other marine organisms) in depths of 200 to 1,000 m, where the majority of fish biomass occurs (Crichton et al., 2023).

F.4.1.4.6 Marine Debris

Updated information on the threats to estuarine and ocean habitats from marine debris that are applicable to fishes is provided in Section F.4.1.4 (General Threats).

F.4.2 ENDANGERED SPECIES ACT-LISTED SPECIES

In the Study Area, nine fish species are listed as endangered or threatened under the ESA. The species are described in Section F.4.2.1 (Atlantic Salmon [*Salmo salar*]) through Section F.4.2.9 (Scalloped Hammerhead Shark [*Sphyrna lewini*]). Atlantic salmon, Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*), and Gulf sturgeon (*A. oxyrinchus desotoi*) are anadromous species that are primarily found in coastal waters, but spend substantial portions of their life cycle in estuarine and riverine waters. The shortnose sturgeon (*A. brevirostrum*) inhabits its natal river and estuary, and very rarely has been observed in coastal waters. Smalltooth sawfish (*Pristis pectinata*) are predominately found in estuarine and coastal waters but can also occur in freshwater and deeper ocean waters. The scalloped hammerhead is generally considered a marine fish but has early life stages that are estuarine. Nassau groupers (*Epinephelus striatus*) are marine fishes that inhabit deep coral reefs or rocky substrate in Florida and the Caribbean. Giant manta rays (*Mobula birostris*) and oceanic whitetip sharks (*Carcharhinus longimanus*) are primarily pelagic and oceanic in distribution and can occur throughout the Study Area.

F.4.2.1 Atlantic Salmon (*Salmo salar*)

F.4.2.1.1 Status and Management

The Gulf of Maine Distinct Population Segment of Atlantic salmon was listed as federally endangered in 2000 (65 *Federal Register* 69459). During 2009, the Gulf of Maine Distinct Population Segment was expanded to include Maine's Penobscot, Kennebec, and Androscoggin Rivers, which support remnant wild populations (74 *Federal Register* 29344). The Atlantic salmon is co-managed by NMFS and the USFWS because its life cycle spans marine, estuarine, and freshwater habitats. Although Atlantic salmon may occur elsewhere (e.g., hatchery programs and aquaculture), only the Gulf of Maine Distinct Population Segment is protected under the ESA.

In June 2009, critical habitat was designated in 45 areas within Maine for the Gulf of Maine Distinct Population Segment of Atlantic salmon (74 *Federal Register* 29300). NMFS revised the critical habitat designation slightly in August 2009 to exclude certain areas (74 *Federal Register* 39903). Critical habitat was designated to include all perennial rivers, streams, and estuaries and lakes connected to the marine environment within the range of the Gulf of Maine Distinct Population Segment of Atlantic salmon, except for those particular areas within the range that are specifically excluded. Within the distinct population segment, the physical and biological features for Atlantic salmon include sites for spawning and incubation, sites for juvenile rearing, and sites for migration. The physical and biological features of habitat are those features that allow Atlantic salmon to successfully use sites for spawning and rearing and sites for migration. These features include the following:

- substrate of suitable size and quality
- rivers and streams of adequate flow, depth, water temperature and water quality
- rivers, streams, lakes and ponds with sufficient space and diverse, abundant food resources to support growth and survival
- waterways that allow for free migration of both adult and juvenile Atlantic salmon
- diverse habitat and native fish communities in which salmon interact with while feeding, migrating, spawning, and resting

In 2015, NMFS focused efforts to protect species that are most at risk of extinction in the near future. The Atlantic salmon was selected as one of the eight species because of their critically low abundance and declining population trends. Key actions include reconnecting the Gulf of Maine with headwater streams, increasing the number of juveniles successfully emigrating into the marine environment, reducing mortality in international fishery in West Greenland waters, and increasing the understanding and ability to improve survival in the marine environment (National Marine Fisheries Service, 2016b).

The latest 5-year review for Atlantic salmon (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a) documents some success in reconnecting the Gulf of Maine with headwater streams. The minimum reclassification requirements for habitat have been met, and even exceeded, as dam removals and improvements to fish passage have increased the quantity of habitat that is both suitable and accessible for spawning and juvenile rearing. However, the review concluded that no change to the status of the species is warranted at this time.

F.4.2.1.2 Habitat and Geographic Range

Atlantic salmon is an anadromous and iteroparous (does not die after spawning like other salmon) species. After hatching, juveniles rear in their natal rivers and estuaries. After juveniles complete the smolting process (e.g., physiologically transforming into marine form called a smolt), they enter the estuarine portion of the Study Area in the Gulf of Maine, primarily at night, during the late spring when water temperatures reach 10° C (50° F) (Sheehan et al., 2012) and school in coastal waters primarily in the upper 3 m (10 ft.), although they may occur in deeper waters (Hedger et al., 2009). Adults migrate back to their natal river to spawn.

The historic range of Atlantic salmon in the northwestern Atlantic Ocean includes coastal drainages from northern Quebec, Canada, to Connecticut. Smolts migrate into marine habitats during approximately two weeks each spring, usually during May (McCormick et al., 1998). Spawning adults migrate into estuaries and natal rivers throughout the spring and summer with the peak occurring in June (Fay et al., 2006). By mid-summer, smolts migrate to the Gulf of Maine along the Scotian Shelf Large Marine Ecosystem (Fay et al., 2006). During their first summer, sub-adults inhabit the coastal waters off Canada, the Southern Grand Banks (Newfoundland-Labrador Shelf Large Marine Ecosystem), the Labrador Sea, and the northern Gulf of St. Lawrence (Reddin & Short, 1991). Decreasing nearshore water temperatures in autumn trigger offshore (greater than 3 nautical miles [NM] from shoreline) movements (Dutil & Coutu, 1988). Sub-adults overwinter in the Labrador Sea south of Greenland. Small percentages return to Gulf of Maine coastal rivers after their first winter at sea (Fay et al., 2006). Atlantic salmon also migrate great distances in the open ocean to reach feeding areas in the West Greenland Shelf Large Marine Ecosystem and in the Davis Strait between Labrador and Greenland (Fay et al., 2006; Reddin & Short, 1991). They spend up to two years feeding before returning to Gulf of Maine coastal rivers to spawn (Reddin & Short, 1991).

F.4.2.1.3 Population Trends

By the end of the 19th century, Atlantic salmon had been extirpated from the Androscoggin, Merrimack, and Connecticut Rivers. The Gulf of Maine Distinct Population Segment represents the last wild population. Populations have been extirpated or decreased from overfishing, land use practices, and development that eliminated spawning and rearing habitat and reduced water quality. The population remains in decline. With added conservation efforts, adult returns remain extremely low.

Adult return rates have continued to decline since the 1980s, which indicates low marine survival (Chaput, 2012). Population estimates have rarely exceeded 5,000 in any given year since 1967, whereas historical abundances (excluding the Penobscot River) likely exceeded 100,000 (Fay et al., 2006). Numerous conservation and restoration practices have slowed the population decline but have not increased recovery. The current average number of Atlantic salmon returning to Gulf of Maine rivers annually is estimated at 1,200 individuals (National Marine Fisheries Service, 2023a). Similar to salmon populations on the West Coast of the United States, changes in ocean conditions affect recovery rates.

The latest 5-year review of Atlantic Salmon (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020a) concluded that the Gulf of Maine distinct population segment remains at critically low abundance. The very low population sizes constitute a significant risk to the resiliency of the species through increasing losses in genetic fitness, loss of adaptive traits, and reduced ability to withstand catastrophic events. Whereas the population growth rate of naturally reared fish has improved in recent years, they are overshadowed by the small population sizes.

F.4.2.1.4 Predator and Prey Interactions

Upon ocean entry, smolts feed on fish larvae (Haugland et al., 2006), amphipods, euphausiids, and small fish (Fraser, 1987; Hislop & Shelton, 1993; Hislop & Youngson, 1984; Jutila & Toivonen, 1985). As they grow, small fishes become an increasingly dominant component of their diet. Striped bass, cod, haddock, fish-eating birds, and marine mammals feed on smolts and sub-adults in the marine environment. Adults prey on capelin, Atlantic herring, and sand lance (Hansen & Windsor, 2006). Adults are vulnerable to predation by seals and cormorants (Fay et al., 2006).

F.4.2.1.5 Species-Specific Threats

Incremental increases in marine survival (survival from emigrating smolts to adult returns) have a much greater impact on the population than comparable increases in freshwater survival (Legault, 2005), however, the factors contributing to low marine survival are not well understood. A review of existing studies indicates that mortality during the early marine migration varies between 8 and 71 percent, with predation being the most common cause of low survival in rivers and estuaries (Thorstad et al., 2015). In recent decades, individuals have migrated to sea at a younger age; these smaller smolts are subject to increased mortality (Russell et al., 2012). Parasitic crustaceans have also been noted to cause mortality and are common in areas with large aquaculture populations (Gargan et al., 2012; Krkosek et al., 2013).

The primary threats impacting the juvenile life stages include restricted fish passage (Baum, 1997), degraded water quality and aluminum toxicity (Kroglund et al., 2007), commercial aquaculture (Hansen & Windsor, 2006), and lack of spawning habitat (Fay et al., 2006). Increases in juvenile survival could enhance the probability of recovery, but only if marine survival is also increased. Current research shows that the catch and release recreational fishery does not negatively impacted the adult population during the spawning migration (Lennox et al., 2016).

F.4.2.2 Atlantic Sturgeon (*Acipenser oxyrinchus oxyrinchus*)

F.4.2.2.1 Status and Management

Atlantic sturgeon is co-managed by Atlantic States Marine Fisheries Commission and NMFS. Sharp declines in the abundance of Atlantic sturgeon resulting from historic overfishing, pollution, habitat loss, and habitat degradation led the Atlantic States Marine Fisheries Commission to issue a coast-wide moratorium on the commercial harvest in state waters in 1998 (63 *Federal Register* 9967). This was followed closely by a similar moratorium in federal waters issued by NMFS in early 1999 (64 *Federal Register* 9449). When the population continued to decline, National Oceanic and Atmospheric Administration listed the species as endangered or threatened throughout its range in 2012 (77 *Federal Register* 5880; 77 *Federal Register* 5914). The Chesapeake Bay, New York Bight, Carolina, and South Atlantic Distinct Population Segments are listed as endangered, and the Gulf of Maine Distinct Population Segment as threatened. The most recent 5-year reviews for the Atlantic sturgeon in the Gulf of Maine, New York Bight, and Chesapeake Bay concluded that no change to the status of the respective distinct population segments is warranted at this time (National Marine Fisheries Service, 2022c, 2022e, 2022h).

In August 2017, NMFS designated critical habitat for each of the five Atlantic sturgeon distinct population segments: Gulf of Maine, New York Bight, Chesapeake Bay, Carolina, and South Atlantic

(82 *Federal Register* 39160). All critical habitat designations are riverine waters between Maine and Georgia related to spawning or potential spawning habitat.

Critical habitat for the Gulf of Maine Distinct Population Segments of Atlantic sturgeon has been designated in the Penobscot, Kennebec, Androscoggin, and Piscataqua Rivers in Maine, Piscataqua River in New Hampshire, and Merrimack River in Massachusetts (82 *Federal Register* 39160).

Critical habitat for the New York Bight Distinct Population Segments of Atlantic sturgeon has been designated in the Connecticut River in Massachusetts, Connecticut and Housatonic Rivers in Connecticut, the Hudson River in New York, the Hudson and Delaware Rivers in New Jersey, and the Delaware River in Pennsylvania and Delaware (82 *Federal Register* 39160).

Critical habitat for the Chesapeake Bay Distinct Population Segments of Atlantic sturgeon has been designated in the Nanticoke and Potomac Rivers, as well as the Marshyhope Creek in Maryland, and the Rappahannock, York, Mattaponi, Pamunkey, and James Rivers in Virginia (82 *Federal Register* 39160).

Critical habitat for the Carolina Distinct Population Segment of Atlantic sturgeon has been designated in the Roanoke, Tar-Pamlico, Neuse, Northeast Cape Fear, Cape Fear, and Pee Dee Rivers in North Carolina; and Pee Dee, Black, Santee, and Cooper Rivers in South Carolina (82 *Federal Register* 39160).

Critical habitat for the South Atlantic Distinct Population Segment has been designated in the Edisto, Combahee, and Savannah rivers in South Carolina, the Ogeechee, Altamaha, Satilla, and St. Marys Rivers in Georgia, and the St. Marys River in Florida (82 *Federal Register* 39160).

The physical features essential for the conservation of Atlantic sturgeon belonging to the Gulf of Maine, New York Bight, and Chesapeake Bay Distinct Population Segments are those habitat components that support successful reproduction and recruitment. These include the following:

- hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0 to 0.5 ppt range) for settlement of fertilized eggs, refuge, growth, and development of early life stages
- aquatic habitat with a gradual downstream salinity gradient of 0.5 up to as high as 30 ppt and soft substrate (e.g., sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development
- water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support the following:
    - unimpeded movement of adults to and from spawning sites
    - seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary
    - staging, resting, or holding of sub-adults or spawning condition adults (Water depths in main river channels must also be deep enough [e.g., at least 1.2 m] to ensure continuous flow in the main channel at all times when any sturgeon life stage would be in the river.)
    - water, between the river mouth and spawning sites, especially in the bottom meter of the water column, with the temperature, salinity, and oxygen values that, combined, support the following: spawning, annual and interannual adult, sub-adult, larval, and juvenile survival, and larval, juvenile, and sub-adult growth, development, and recruitment (e.g., 13 to 26°C for spawning habitat and no more than 30°C for juvenile-

rearing habitat, and 6 milligrams per liter [mg/L] or greater dissolved oxygen for juvenile-rearing habitat)

The physical features essential for the conservation of Atlantic sturgeon belonging to the Carolina and South Atlantic Distinct Population Segments are those habitat components that support successful reproduction and recruitment. These include the following:

- hard bottom substrate (e.g., rock, cobble, gravel, limestone, boulder, etc.) in low salinity waters (i.e., 0.0 to 0.5 ppt range) for settlement of fertilized eggs and refuge, growth, and development of early life stages
- aquatic habitat inclusive of waters with a gradual downstream gradient of 0.5 up to as high as 30 ppt and soft substrate (e.g., sand, mud) between the river mouth and spawning sites for juvenile foraging and physiological development
- water of appropriate depth and absent physical barriers to passage (e.g., locks, dams, thermal plumes, turbidity, sound, reservoirs, gear, etc.) between the river mouth and spawning sites necessary to support the following:
  - unimpeded movement of adults to and from spawning sites
  - seasonal and physiologically dependent movement of juvenile Atlantic sturgeon to appropriate salinity zones within the river estuary
  - staging, resting, or holding of sub-adults or spawning condition adults (Water depths in main river channels must also be deep enough (at least 1.2 m) to always ensure continuous flow in the main channel when any sturgeon life stage would be in the river.)
- water quality conditions, especially in the bottom meter of the water column, with temperature and oxygen values that support the following:
  - spawning
  - annual and inter-annual adult, sub-adult, larval, and juvenile survival
  - larval, juvenile, and sub-adult growth, development, and recruitment (Appropriate temperature and oxygen values will vary interdependently and depending on salinity in a particular habitat. For example, 6.0 mg/L dissolved oxygen or greater likely supports juvenile-rearing habitat, whereas dissolved oxygen less than 5.0 mg/L for longer than 30 days is less likely to support rearing when water temperature is greater than 25° C. In temperatures greater than 26° C, dissolved oxygen greater than 4.3 mg/L is needed to protect survival and growth. Temperatures of 13 to 26° C likely support spawning habitat.)

F.4.2.2.2 Habitat and Geographic Range

Sub-adult and adult Atlantic sturgeon inhabit the Newfoundland-Labrador Shelf, Scotian Shelf, Northeast U.S. Continental Shelf, and Southeast U.S. Continental Shelf Large Marine Ecosystems, likely year-round. Juveniles, sub-adults, and adults also inhabit many of the estuarine and riverine systems that are included in the Study Area (e.g., Kennebec River in Maine, Chesapeake Bay, James River and York River in Virginia, Cooper River in South Carolina, St. Johns River in Florida, and St. Marys River and Kings Bay in Georgia). Larvae are not known to inhabit the Study Area.

Atlantic sturgeon are fairly well studied during their juvenile and spawning life phases in riverine environments, but their sub-adult and adult estuarine and marine phases are less understood. Females spawn highly adhesive eggs on cobble substrate located on river bottoms, which are fertilized by males. Breece et al. (2013) found that spawning habitat in the Delaware River was influenced by salinity and substrate composition. Larvae hatch out in four to seven days, and newly hatched young are active swimmers, frequently leaving the bottom and swimming throughout the water column. After 9 to 10 days, the yolk sac is absorbed, and the larvae begin to show more strictly benthic behavior. Adults return to lower estuarine or marine environments shortly after spawning. Juveniles remain riverine and estuarine residents for 1 to 4 years before migrating to the Atlantic Ocean (Balazik et al., 2012a). After reaching 76 to 92 cm in length (30 to 36 in.), sub-adults move from natal estuaries into the marine environment, and may undertake long range migrations (National Marine Fisheries Service, 2007b).

Spawning was originally thought to occur only in the spring along the Atlantic coast; however, recent research indicates that spawning primarily occurs in the fall in the South Atlantic (from Georgia to Chesapeake Bay) rather than spring (Balazik, 2012; Balazik & Musick, 2015; Hager, 2015; Kahn et al., 2014; Smith et al., 2015). Fall spawning also occurs in Chesapeake Bay tributaries: Pamunkey River, a tributary of the York River, and Marshyhope Creek, a tributary of the Nanticoke River (Hager et al., 2014; Horne & Stence, 2016; Kahn et al., 2014; Secor et al., 2022). There is currently no evidence of fall spawning in the New York Bight or Gulf of Maine distinct population segments (National Marine Fisheries Service, 2022e, 2022h).

Sub-adults disperse widely both north and south along the Atlantic coast and beyond the continental shelf (Bain, 1997). Sub-adults and adults were found to be strongly associated within a narrow range of depths 10 to 50 m over gravel and sand and, to lesser extent, silt and clay (Stein et al., 2004) and in temperatures around 20° C (Breece et al., 2016). However, recent acoustic monitoring research suggests that, in at least some areas, Atlantic sturgeon may occupy deeper portions of the continental shelf more often than previously thought (Hager & Mathias, 2018). A study of a potential wind energy site offshore of New York found that Atlantic sturgeons were present over the continental shelf mostly during winter (Ingram et al., 2019). The number of detections was highest in shallow water and generally decreased with increasing depth and distance from shore. Rothermel et al. (2020) provided more detailed information for marine habitats used by Atlantic sturgeon off the coast of Maryland, and the migratory patterns of Atlantic sturgeon through the area. Their findings also provide further information indicating that Atlantic sturgeon occur farther offshore in the late fall and winter months than in the spring and summer, though consistent occurrence in relatively shallow, nearshore waters was also supported. Atlantic sturgeon in the marine environment are known to concentrate around the mouths of inlets and large estuaries in spring, summer, and fall (Dunton et al., 2010; Erickson et al., 2011). While inhabiting these areas, sturgeon have been found to associate with river plumes or sandy and muddy substrate that may offer increased foraging opportunities (Breece et al., 2016; Laney et al., 2007; Oliver et al., 2013; Savoy & Pacileo, 2003).

Age of sexual maturity varies from 5 to 34 years depending on latitude, averaging 10 and 15 (Breece et al., 2013) years for males and females, respectively (National Marine Fisheries Service, 2007b). Sturgeon in the southern parts of the range tend to mature faster but experience shorter lifespans than sturgeon in the northern portions of the range. Despite extensive mixing in coastal waters, adults return to their natal river to spawn as indicated from tagging records. During non-spawning years, adults remain in marine waters either year-round or seasonally venture into either natal or non-natal estuarine environments (Bain, 1997; Hager et al., 2016). For example, as part of a Navy-funded research effort, Hager et al. (2016) found that sturgeon implanted with acoustic transmitters in the York River system in Virginia spent the summer and fall seasons of non-spawning years in either the mainstem of the Chesapeake Bay, the Delaware Bay and the Delaware River, or along the coast of New York and in the Hudson River.

Although generally considered benthic species, sturgeons are occasionally found at the surface and some species are known to leap into the air, possibly to take air into the swim bladder to maintain neutral buoyancy (Dunbar, 2015; Thorn & Falgiani, 2013; Watanabe et al., 2008). Most of their time is likely spent near the bottom, based on eight years of tracking data from 69 Atlantic sturgeon and shortnose sturgeon adults with depth-sensing acoustic tags in the Penobscot River, Maine (Dunbar, 2015). An investigation of depth and vertical movements of Atlantic sturgeon and shortnose sturgeon in the Penobscot River found that all fish remained at the bottom for extended time periods, sometimes for days (Dunbar, 2015). Out of hundreds of thousands of recorded data points, only 33 occurred at depths less than 0.5 m; these were assumed to be associated with surfacing behavior. Although a different species on a different continent, a study of seven Chinese sturgeons (*Acipenser sinensis*) found two swim patterns and depth profiles, presumably related to buoyancy and swim bladder function (Watanabe et al., 2008). Four individuals actively swam in the water column at depths of 7 to 31 m, surfacing occasionally. Three fish spent nearly all their time (88 to 94 percent) on the bottom, with relatively rapid swims to the surface followed by a glide back the bottom. A subsequent study involving Chinese sturgeons found that all individuals swam in an up-and-down pattern, moving between the bottom and approximately 5 m depth, and individuals occasionally surfaced (Watanabe et al., 2012).

F.4.2.2.3 Population Trends

Atlantic sturgeon is a long-lived (average lifespan of 60 years), late-maturing, estuarine-dependent, iteroparous, and anadromous species. Abundance estimates are available for some spawning rivers (National Marine Fisheries Service, 2020a) and are summarized below.

- Between 18,000 and 21,000 individuals were estimated for the St. John River, Canada, between 2013 and 2015, making this the largest population on the Atlantic coast.
- In 1995, a total of 9,500 juveniles were estimated in the Hudson River, New York (4,600 fish of wild origin and 4,900 hatchery-raised fish). Commercial fishery data indicated an estimated 870 adults in 1998.
- In 2014, approximately 3,700 juveniles were estimated in the Delaware River.
- Approximately 1,800 to 1,900 adults and sub-adults have been estimated for the Pee Dee River, South Carolina.
- Between 1,000 and 2,000 juveniles are thought to occur in the Altamaha River, Georgia.
- Approximately 300 adults were estimated for the York River, Virginia. This number is consistent with the 325 adults estimated as a result of a Navy tracking study (Hager & Mathias, 2018).
- Sampling in the Pamunkey and Mattaponi Rivers, which are tributaries of the York River, resulted in spawning-run abundance estimates ranging from 31 to 576 fish (95 percent confidence interval) between 2013 and 2018 (Kahn et al., 2019).
- Between 3 and 36 individuals are estimated for the Roanoke River.
- Between 0 and 23 individuals are estimated for the Neuse River.
- Between 35 and 152 individuals are estimated for the Cape Fear River.
- Between 108 and 231 individuals are estimated for the Satilla River.
- Between 819 and 1,446 individuals are estimated for the Savannah River.
- A low but unspecified number of adults and subadults occur in the St. Johns River, Florida during winter and spring, but the spawning population of this river appears to be extinct (Fox et al., 2018; Fox et al., 2016; National Marine Fisheries Service, 2007b; Waldman & Wirgin, 1998).

The Atlantic sturgeon has been overfished throughout its range, with landings peaking around the turn of the 20th century followed by drastic declines thereafter (Smith & Clugston, 1997). Historically, the species was recorded in 38 rivers from St. Croix, Maine to the St. Johns River, Florida. As of 2007, they were only known to still occupy 35 rivers (National Marine Fisheries Service, 2007b). However, spawning populations have been discovered in at least five new rivers since this estimate and preliminary research indicates there are likely spawning populations in several more rivers that have yet to be fully investigated.

In the early 1600s, Atlantic sturgeon was considered an important fishery (Atlantic Sturgeon Status Review Team, 2007). In the mid-1800s, incidental catch of Atlantic sturgeon in the shad and river herring seine fisheries indicated that the species was very abundant (Armstrong & Hightower, 2002). By 1870, females were collected for their eggs, which were sold as caviar. By 1890, over 3,350 metric tons were landed from rivers along the Atlantic coast (Smith & Clugston, 1997). Despite a moratorium on commercial fishing for this species since 1998, there has been no indication of recovery. The latest 5-year reviews for Atlantic sturgeon (National Marine Fisheries Service, 2022c, 2022e, 2022h) state that the Gulf of Maine, New York Bight, and Chesapeake Bay distinct population segments have likely neither improved nor declined relative to their status when they were listed in 2012. The lack of recovery is attributed to channel dredging and ship strikes, coastal development, pollution, poor water quality, and habitat degradation and loss (National Marine Fisheries Service, 2022c, 2022e, 2022h). Current population estimates for the Gulf of Maine, New York Bight, and Chesapeake Bay distinct population segments are 7,455, 34,566, and 8,811, respectively. These estimates encompass many age classes because subadults can be as young as one year old when they first enter the marine environment, and adults can live at least as long as 60 years (Balazik et al., 2012a; Hilton et al., 2016).

F.4.2.2.4 Predator and Prey Interactions

Atlantic sturgeon prey upon benthic invertebrates such as isopods, crustaceans, worms, and molluscs (National Marine Fisheries Service, 2022b). It has also been documented to feed on bottom-dwelling fish (Bain, 1997). Evidence of predation on sturgeon is scant, but it is speculated that juveniles may be eaten by the American alligator (*Alligator mississippiensis*), alligator gar (*Atractosteus spatula*), striped bass (*Morone saxatilis*) (Dadswell et al., 1984), and sharks. Recent observations suggest occasional predation on juvenile sturgeon by other protected species (e.g., gray seals, birds of prey) (National Marine Fisheries Service, 2022c, 2022e, 2022h). Predation of early Atlantic sturgeon life stages by introduced, non-native, catfish species has also been suggested as contributing to the very low capture rates of juvenile Atlantic sturgeon in the James River and in the other Chesapeake Bay tributaries. Based on DNA analysis of potential predator stomachs in the Pamunkey River, the highest percentage of sturgeon DNA (likely eggs or larvae) was found in striped bass, non-native common carp (*Cyprinus carpio*), and blue catfish (*Ictalurus furcatus*) (Bunch et al., 2021).

F.4.2.2.5 Species-Specific Threats

Overfishing of females for caviar prior to the 1900s resulted in large population declines. Current threats include bycatch; habitat degradation from dredging, dams, and water withdrawals; passage impediments including locks and dams; degraded water quality; ship strikes; and bycatch in commercial fisheries (Atlantic Sturgeon Status Review Team, 2007; Balazik et al., 2012b; Brown & Murphy, 2010; Foderaro, 2015; National Marine Fisheries Service, 2018a).

The copepod *Dichelesthium oblongum* parasitizes 93 percent of the Atlantic sturgeon sampled in the New York Bight (Fast et al., 2009). High parasite load, stress, and reduced immune suppression has been associated with Atlantic sturgeon inhabiting areas of poor water quality (e.g., sewage contamination). Hilton et al. (2016) reviewed diseases and parasites known to affect Atlantic sturgeon, which include but are not limited to various parasitic worms, copepods, and isopods.

F.4.2.3 Shortnose Sturgeon (*Acipenser brevirostrum*)

F.4.2.3.1 Status and Management

In 1967, the U.S. Department of Interior listed the shortnose sturgeon as endangered throughout its range (32 *Federal Register* 4001). The species remained listed following enactment of the ESA in 1973 (Wippelhauser & Squiers, 2015). NMFS recognizes 19 distinct population segments that inhabit 25 river systems ranging from the Saint John River in New Brunswick, Canada, to the St. Johns River, Florida (National Marine Fisheries Service, 1998). NMFS also recognizes three metapopulations (genetically similar groups), based on the current distribution of spawning individuals, including the Carolinian Province (southern metapopulation), Virginian Province (mid-Atlantic metapopulation), and Acadian Province (northern metapopulation) (National Marine Fisheries Service, 2023c). Critical habitat for this species remains under development.

F.4.2.3.2 Habitat and Geographic Range

Like all sturgeon species, shortnose sturgeon spawn by releasing eggs in freshwater rivers (National Marine Fisheries Service, 2023c). After the fertilized eggs hatch, the larvae orient into the current and away from light, generally staying near the bottom and seeking cover. Within two weeks, the larvae emerge from cover and swim in the water column, moving downstream from the spawning site. Within two months, juvenile behavior mimics adults, with active swimming (Deslauriers & Kieffer, 2012) and foraging at night along the bottom (Richmond & Kynard, 1995). In estuaries, juveniles and adults occupy areas with little or no current over a bottom composed primarily of mud and sand (Secor et al., 2000). Adults are found in deep water (10 to 30 m) in winter and in shallower habitat (2 to 10 m) during summer (Welsh et al., 2002). Juveniles are known to occur in the Study Area, particularly in the St. Johns River in Florida. When adult shortnose sturgeon enter marine waters, they generally stay close to shore (National Marine Fisheries Service, 2023c).

Although sturgeon are considered to be a benthic species, they have also been observed leaping from the water in riverine habitats (Dunbar, 2015; Thorn & Falgiani, 2013). Most of their time is likely spent near the bottom, based on eight years of tracking data from 69 Atlantic sturgeon and shortnose sturgeon adults with depth-sensing acoustic tags in the Penobscot River, Maine (Dunbar, 2015). Of the hundreds of thousands of data points collected for the study, only 33 were from depths of less than 0.5 m; these were assumed to be associated with surfacing behavior. As described in Section F.4.2.2.2 (Atlantic Sturgeon, Habitat and Geographic Range), studies of Chinese sturgeons found that individuals surfaced only occasionally, presumably to take in air to help maintain neutral buoyancy.

The geographic range of shortnose sturgeon extends along the coastal rivers and estuaries of eastern North America from the Saint John River, New Brunswick, Canada, to the St. Johns River, Florida (Kynard, 1997; National Marine Fisheries Service, 2023c). The species primarily occurs in rivers and estuaries of the Northeast and Southeast U.S. Continental Shelf Large Marine Ecosystems, occasionally moving into the nearshore coastal waters (Dadswell, 2006; National Marine Fisheries Service, 1998; Richmond & Kynard, 1995). Among Study Area-Inshore Locations of the Proposed Action (e.g., training areas, ports), shortnose sturgeon populations occur in the Kennebec, Piscataqua, James, Cooper, Savannah, St. Marys, and St. Johns Rivers (National Marine Fisheries Service, 2023c).

F.4.2.3.3 Population Trends

The shortnose sturgeon is a long-lived (average lifespan 30 years), riverine- and estuarine-habitat-dependent, iteroparous, and anadromous species. Populations were stable or possibly increasing in the 1990s (Wippelhauser et al., 2015). The current population status of shortnose sturgeon varies among the occupied rivers (National Marine Fisheries Service, 2023c). Overall, populations in the southeast are relatively small compared to northeast populations. Of southeast U.S. rivers that coincide with Study Area-Inshore Locations of the Proposed Action, the Savannah River supports the largest known shortnose sturgeon population with estimates ranging from 1,390 (2009) to 2,432 (2013). The status of other riverine populations that coincide with other proposed Study Area-Inshore Locations is currently unknown, though spawning has been documented in the Cooper and Kennebec Rivers.

F.4.2.3.4 Predator and Prey Interactions

Shortnose sturgeon are benthic omnivores that typically feed by suctioning prey from the bottom (National Marine Fisheries Service, 1998; Stein et al., 2004), but individuals have also been observed feeding off plant surfaces (Dadswell et al., 1984). Prey items include crustaceans, insect larvae, worms, and molluscs. Young-of-the-year were found to consume amphipods and dipteran (fly) larvae (National Marine Fisheries Service, 1998). Juveniles similarly feed primarily on insect larvae and small crustaceans. Adults feed primarily on molluscs, with freshwater mussels being a primary prey item.

Prey varies with season between northern and southern river systems. In northern rivers, some sturgeon feed in freshwater during summer and over sand-mud bottoms in the lower estuary during fall, winter, and spring (National Marine Fisheries Service, 1998). In southern rivers, feeding has been observed during winter at or just downstream the saltwater and freshwater interface (Kynard, 1997). In the Southeast U.S. Continental Shelf Large Marine Ecosystem, shortnose sturgeon reduces feeding activity during summer months (Sulak & Randall, 2002).

Young-of-the-year have been found in the stomachs of yellow perch (*Perca flavescens*) (Dadswell et al., 1984). Predation on sub-adults and adults is not well documented; however, they may possibly be preyed upon by gars, alligators, sharks, and seals (Dadswell et al., 1984; National Marine Fisheries Service, 1998).

F.4.2.3.5 Species-Specific Threats

The population decline has been attributed to pollution, overharvest in commercial fisheries, bycatch in current fisheries, and its resemblance to the formerly commercially valuable Atlantic sturgeon (National Marine Fisheries Service, 2023c). Other risk factors include dams that block access to spawning areas, poaching, non-native species, poor water quality in spawning and nursery habitats, contaminants (e.g., heavy metals, pesticides, and organochlorine compounds), siltation from dredging, bridge construction and demolition, impingement on power plant cooling water intake screens, impoundment operations, hydraulic dredging operations, and bycatch in commercial fisheries (Collins et al., 2000; National Marine Fisheries Service, 1998, 2023c).

F.4.2.4 Gulf Sturgeon (*Acipenser oxyrinchus desotoi*)

F.4.2.4.1 Status and Management

The Gulf sturgeon and the Atlantic sturgeon are members of the same species but do not overlap geographically. The Gulf sturgeon was federally listed in 1991 as threatened in the Gulf of Mexico Large Marine Ecosystem (56 *Federal Register* 49653) and is co-managed by NMFS and the USFWS. The fishery for Gulf sturgeon has been closed since the species was listed. Bycatch along the gulf coast was a major source of mortality (U.S. Fish and Wildlife Service, 1995), and efforts to reduce bycatch include gear modifications for nearshore trawl fisheries (Smith & Clugston, 1997). In 2009, NMFS and the USFWS concluded that the Gulf sturgeon population was stable and had achieved recovery objectives (U.S. Fish and Wildlife Service & National Marine Fisheries Service, 2009). However, in the same document, NMFS and the USFWS also concluded that the best available information indicates the Gulf sturgeon continues to meet the definition of a threatened species.

In 2003, NMFS and the USFWS collectively designated critical habitat for Gulf sturgeon within and adjacent to the states of Louisiana, Mississippi, Alabama, and Florida (68 *Federal Register* 13369). The physical and biological features essential for the conservation of Gulf sturgeon were determined to be those habitat components that support feeding, resting, sheltering, reproduction, migration, and physical features necessary for maintaining the natural processes that support these habitat components.

The physical and biological features include the following:

- abundant prey items within riverine habitats for larval and juvenile life stages, and within estuarine and marine habitats and substrates for juvenile, sub-adult, and adult life stages
- riverine spawning sites with substrates suitable for egg deposition and development, such as limestone outcrops and cut limestone banks, bedrock, large gravel or cobble beds, marl, soapstone, or hard clay
- riverine aggregation areas, also referred to as resting, holding, and staging areas, used by adult, sub-adult, and/or juveniles, generally, but not always, located in holes below normal riverbed depths, believed necessary for minimizing energy expenditures during freshwater residency and possibly for osmoregulatory functions
- a flow regime (i.e., the magnitude, frequency, duration, seasonality, and rate-of-change of freshwater discharge over time) necessary for (1) normal behavior, growth, and survival of all life stages in the riverine environment, including migration, breeding site selection, courtship, egg fertilization, resting, and staging; and (2) maintaining spawning sites in suitable condition for egg attachment, eggs sheltering, resting, and larvae staging
- water quality, including temperature, salinity, pH, hardness, turbidity, oxygen content, and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages
- sediment quality, including texture and other chemical characteristics, necessary for normal behavior, growth, and viability of all life stages
- safe and unobstructed migratory pathways necessary for passage within and between riverine, estuarine, and marine habitats (e.g., a river unobstructed by any permanent structure, or a dammed river that still allows for passage)

Most features of the critical habitat are not applicable to the marine portions of the Study Area. The Panama City OPAREA and the Naval Surface Warfare Center Panama City Division Testing Range overlap with Gulf sturgeon critical habitat (Figure 3.6-4, Critical Habitat for ESA-Listed Gulf Sturgeon Designated within the Study Area). This critical habitat (Unit 11) encompasses nearshore Gulf of Mexico waters off Escambia, Santa Rosa, Okaloosa, Walton, Bay, and Gulf Counties in Florida. Unit 11 provides a migration corridor for Gulf sturgeon transiting from winter habitat and feeding grounds in the Gulf of Mexico to spring and summer (and possibly fall) spawning and hatching habitat in the Yellow, Choctawhatchee, and Apalachicola Rivers. Gulf sturgeon inhabit the nearshore coastline between Pensacola and Apalachicola bays, in depths of less than 6 m during winter.

F.4.2.4.2 Habitat and Geographic Range

The Gulf sturgeon is a long-lived, riverine and estuarine habitat-dependent, iteroparous, and anadromous species. This species occurs in the Gulf of Mexico Large Marine Ecosystem in bays, estuaries and rivers, and in the marine environment from Florida to Louisiana (National Marine Fisheries Service, 2022f), including the nearshore portion of the Panama City OPAREA. The spring spawning migration toward natal rivers begins as riverine water temperatures reach approximately 64 to 72°F

(Edwards et al., 2003; Heise et al., 2004; National Marine Fisheries Service, 2022f; Rogillio et al., 2007). Spawning areas include the Suwannee, Apalachicola, Escambia, Choctawhatchee, Yellow/Blackwater, and Pascagoula Rivers (Chapman & Carr, 1995; Craft et al., 2001; Fox et al., 2000; Wooley & Crateau, 1985). Spawning occurs during autumn in some watersheds (e.g., Suwannee River) (Randall & Sulak, 2012). Post-spawned adults often inhabit estuaries and nearshore bays in water less than 10 m deep (Ross et al., 2009). In the marine environment, sub-adults and adults are generally thought to remain near the shoreline. For example, an acoustic tagging study found that Gulf sturgeons located offshore of the Florida Panhandle remained within 1,000 m of the shoreline during two of the three years of study, although sturgeons were detected farther offshore (at least 1,250 m; maximum distance unknown) during one year (Robydek & Nunley, 2012). Some individuals, particularly females between spawning years (Fox et al., 2002; Ross et al., 2009) move into deeper offshore waters for short periods during cold weather (Sulak et al., 2009). Gulf sturgeons have been captured or detected from 11 to 16.7 km offshore of the Suwannee River mouth (68 *Federal Register* 13369).

Sub-adult and adult foraging grounds include barrier island inlets with strong tidal currents and estuaries less than 2 m deep with clean sand substrate (Fox et al., 2002; Harris et al., 2005; Ross et al., 2009). Gulf sturgeon winter near beaches of northwestern Florida and southeast of the mouth of St. Andrew Bay (U.S. Fish and Wildlife Service & National Marine Fisheries Service, 2009), while others moved northeast of St. Andrew Bay at depths ranging from 4 to 12 m (12 to 40 ft.) at 0.5 to 2 miles (mi.) offshore, and likely feeding on prey associated with fine sand and shell hash substrates (U.S. Fish and Wildlife Service & National Marine Fisheries Service, 2009).

By December, only young-of-the-year and juveniles remain in the rivers (Carr et al., 1996; Foster & Clugston, 1997). Young-of-the-year nursery habitat includes riverine sandbars and shoals (Carr et al., 1996). Juveniles show high site fidelity rates for riverine habitats used during spring and summer (Rudd et al., 2014), prefer sand or vegetated habitats (Wakeford, 2001), tolerate high salinity levels for extended durations, and appear to use estuaries infrequently (Sulak et al., 2009). Adults inhabit nearshore waters from October to February (Robydek & Nunley, 2012) with distribution influenced by prey availability (Ross et al., 2009), particularly within the Suwannee River estuary (Harris et al., 2005).

Whereas sturgeons are a benthic species, they have also been observed leaping from the water in riverine habitats (Dunbar, 2015; Thorn & Falgiani, 2013). Most of their time is likely spent near the bottom, based on eight years of tracking data from 69 Atlantic sturgeon and shortnose sturgeon adults with depth-sensing acoustic tags in the Penobscot River, Maine Dunbar (2015). Of the hundreds of thousands of data points collected for the study, only 33 were from depths of less than 0.5 m; these were assumed to be associated with surfacing behavior. As described in Section F.4.2.2.2 (Atlantic Sturgeon, Habitat and Geographic Range), studies of Chinese sturgeons found that individuals surfaced only occasionally, presumably to take in air to help maintain neutral buoyancy.

F.4.2.4.3 Population Trends

Gulf sturgeon populations are stable or slowly increasing in eastern river systems, particularly the Suwannee River (U.S. Fish and Wildlife Service & National Marine Fisheries Service, 2009, 2022). The current status of populations in the western portion of the range (Louisiana and Mississippi) is uncertain but abundance is thought to be lower than that of eastern populations, possibly due in part to the effects of hurricanes. The results of a modeling effort suggest that current population levels in four of the seven evaluated river systems are likely at or exceeding the mean carrying capacity, given the current levels of available habitat (Ahrens & Pine, 2014). The implication is that, if the results are accurate, further population increases are unlikely unless habitat improvement or restoration occurs. In the remaining three rivers, extant Gulf Sturgeon populations are likely below their estimated carrying capacity levels. Population estimates in the Pearl and Pascagoula Rivers are lacking because research is limited (Rogillio et al., 2007).

F.4.2.4.4 Predator and Prey Interactions

Prey varies by life stage, but Gulf sturgeon is considered an opportunistic feeder. Adults typically do not feed while in freshwater and may lose from 12 to 30 percent of their body weight while inhabiting rivers. In estuarine and marine habitats, they prey upon a wide range of benthic invertebrates (Florida Museum of Natural History, 2017). Sharks are likely predators while sturgeon inhabit the marine environment (Florida Museum of Natural History, 2017).

F.4.2.4.5 Species-Specific Threats

Factors contributing to the decline include overfishing and habitat loss. Threats include dams (e.g., Pearl, Alabama, and Apalachicola Rivers), dredged material disposal, channel maintenance, oil and gas exploration, shrimp trawling, and poor water quality (U.S. Fish and Wildlife Service & National Marine Fisheries Service, 2009). Other threats include potential hybridization with non-native sturgeon from aquaculture farms and diseases.

F.4.2.5 Smalltooth Sawfish (*Pristis pectinata*)

F.4.2.5.1 Status and Management

The distinct population segment of smalltooth sawfish in the United States, which occurs between Florida and Cape Hatteras, North Carolina, was listed as endangered under the ESA by NMFS in 2003 and by the USFWS in 2005 (70 *Federal Register* 69464), and it is co-managed by both agencies (National Marine Fisheries Service, 2010d). Other distinct population segments of smalltooth sawfish (e.g., Bahamas and Cuba) are not expected in the Study Area. The most recent 5-year review of smalltooth sawfish concluded that no change to the status of the species is warranted at this time (National Marine Fisheries Service, 2018b).

In September 2009, NMFS designated approximately 840,472 acres in two units of critical habitat occupied by the U.S. distinct population segment of smalltooth sawfish (74 *Federal Register* 45353). The two units determined for critical habitat designations are the Charlotte Harbor Estuary Unit, which comprises approximately 221,459 acres of habitat, and the Ten Thousand Islands/Everglades Unit, which comprises approximately 619,013 acres of habitat. The two units are located along the southwestern coast of Florida between Charlotte Harbor and Florida Bay.

These areas contain the following physical and biological features that are essential to the conservation of smalltooth sawfish and that may require special management considerations or protection: red mangroves and shallow euryhaline habitats characterized by water depths between the mean high water line and 3 ft. (0.9 m) measured at mean lower low water. The Key West Range Complex does not overlap these areas; the northeastern boundary (Warning Area-174) of the Key West Range Complex is within approximately 9 NM of critical habitat at its closest point.

F.4.2.5.2 Habitat and Geographic Range

The smalltooth sawfish typically inhabit shallow tropical or subtropical estuarine and marine waters associated with sandy and muddy deep holes, limestone hard bottom, coral reefs, sea fans, artificial reefs, and offshore drilling platforms (Poulakis & Seitz, 2004). Mating likely occurs during spring (March to April) in Florida Bay (National Marine Fisheries Service, 2018b; Papastamatiou et al., 2015). Nursery areas of the smalltooth sawfish include estuaries and mangroves with the roots providing refuge from predators (National Marine Fisheries Service, 2009a, 2010d; Seitz & Poulakis, 2006; Simpfendorfer &

Wiley, 2005). Juveniles exhibit a high site fidelity to nearshore areas and residence up to 55 days, and upstream movement toward preferred lower salinity conditions (Poulakis et al., 2012; Simpfendorfer et al., 2011). Larger individuals may occur to a depth of about 120 m (Poulakis & Seitz, 2004), although adults are known to spend more time in shallower habitat (e.g., river mouths) than in deeper waters (Simpfendorfer & Wiley, 2005). One study found that larger juveniles and adults primarily occupy waters of less than 20-m depth near coastal mangroves (Carlson et al., 2014). Observations in offshore waters are rare.

The range of the species has been contracting based on a time series of observations (National Marine Fisheries Service, 2018b). The species has been recorded in the Southeast U.S. Continental Shelf and Gulf of Mexico Large Marine Ecosystem areas of the Study Area, but its range is primarily southern Florida. The most recent 5-year review indicated the species has been observed in nearshore bays and estuaries from eastern Louisiana to Kings Bay, Georgia, from 1990 to 2015 (National Marine Fisheries Service, 2018b). In more recent years, acoustic tag monitoring off the coast of Cape Canaveral has documented the occurrence of several smalltooth sawfish north of southern Florida (Iafrate et al., 2022).

F.4.2.5.3 Population Trends

No population estimates exist for the smalltooth sawfish. The best available data suggest that the current population is a small fraction of its historical size and restricted mostly to southern Florida (National Marine Fisheries Service, 2018b; Poulakis & Seitz, 2004; Simpfendorfer, 2006; Simpfendorfer et al., 2011). Data collected in the Everglades National Park since 1972 suggest that the population has stabilized and may be increasing. Completing a similar analysis with updated data, Carlson and Osborne (2012) found the abundance trend was stable to slightly increasing (approximately 5 percent per year), although variation was high. Evidence from other data sources also indicate the current population of smalltooth sawfish is at least stable throughout its core abundance area with the potential for expansion from Charlotte Harbor to Florida Bay (National Marine Fisheries Service, 2018b). Recent studies have also shown that the species' growth rate is faster (Scharer et al., 2012; Simpfendorfer et al., 2008), and that individuals mature earlier (Carlson & Simpfendorfer, 2015), than previously thought. In the context of population modeling, these characteristics suggest the population can recover faster than once thought, as long as current stressors do not increase substantially (Carlson & Simpfendorfer, 2015). Increased occurrence in Biscayne Bay and the adjacent reef tract was reported in a recent study (McDonnell et al., 2020). Overall, population data indicate the potential for eventual recovery of the species in the United States and possibly other regions (Brame et al., 2019).

F.4.2.5.4 Predator and Prey Interactions

Adult sawfishes have very few predators, although individuals may be preyed upon by various large shark species (Florida Museum of Natural History, 2018a, 2018b). Juveniles are preyed upon by crocodiles, large sharks, and dolphins (Florida Museum of Natural History, 2018a, 2018b; National Marine Fisheries Service, 2018b). Whereas the final listing rule for smalltooth sawfish did not identify predation as a causal listing factor, current data from acoustic monitoring, public encounter database data, satellite archival tagging data, and photographs suggests that small juveniles use red mangrove habitat to avoid predators (National Marine Fisheries Service, 2018b). Smalltooth sawfish are nocturnal feeders and use the saw-like rostrum to disrupt the substrate to expose crustaceans and to stun and slash schooling fish.

F.4.2.5.5 Species-Specific Threats

Factors contributing to the historic population decline included habitat degradation, commercial harvest, gear entanglements, bycatch in fisheries, poaching, and the illegal market for the saw-like rostrum (WildEarth Guardians, 2009). The most recent 5-year review concluded that mortality from fisheries bycatch, and habitat loss likely represent the largest threats continuing to impact smalltooth sawfish recovery (National Marine Fisheries Service, 2018b). Other stressors of unknown significance to smalltooth sawfish populations include marine pollution and debris, boating activity (causing degradation of habitat and physical disturbance or strike hazard), and extreme natural events (e.g., hurricanes, red tides, unusually cold water temperature). Recent data on fisheries bycatch from shrimp trawling (2009–2015) and the shark longline fishery (1994–2018) suggest interactions with smalltooth sawfish are rare and the survival rate of released individuals is probably high for at least some types of incidental capture (longlines and gillnets) (National Marine Fisheries Service, 2018b). For example, only nine smalltooth sawfish were caught in Florida shrimp trawls from 2009 to 2015, and there have been only 34 interactions in the shark longline fishery since 1994. Habitat loss is associated with permitted wetland conversions, erosion of coastal wetlands adjoining artificially hardened shorelines, growth of urban areas/impervious surfaces, water management practices in coastal watersheds, point- and non-point sources of pollution (e.g., stormwater discharge), channel dredging, and the exacerbating effects of climate change on habitat loss (National Marine Fisheries Service, 2018b).

F.4.2.6 Giant Manta Ray (*Mobula birostris*)

F.4.2.6.1 Status and Management

In January 2018, NMFS listed the giant manta ray as threatened throughout its range (83 *Federal Register* 2916). In December 2019, NMFS determined that designation of critical habitat is not prudent (84 *Federal Register* 66652). NMFS determined that there are currently no identifiable physical or biological features that are essential to conservation of the giant manta ray within areas under U.S. jurisdiction, and that therefore there are no areas that meet the definition of critical habitat for the species.

F.4.2.6.2 Habitat and Geographic Range

Giant manta rays are considered seasonal visitors to productive coastlines with regular upwelling, including oceanic island shores, and offshore pinnacles and seamounts. The species has also been observed in estuarine waters near oceanic inlets. They may occur in water depths from less than 10 m to over 1,000 m. They use sandy bottom habitat and seagrass beds, as well as shallow reefs, and the ocean surface both inshore and offshore. The species ranges globally and is distributed in tropical, subtropical, and temperate waters. They can migrate seasonally, usually more than approximately 621 mi. (1,000 km), however, they are not likely to cross ocean basins (National Marine Fisheries Service, 2022d). Data from a tagging study indicate at least some populations may not be as migratory as previously thought (Miller & Klimovich, 2016). The timing of migratory movements seems to vary between regions and may be influenced by zooplankton movements, water currents and tidal patterns, water temperature, and possibly mating behavior.

Farmer et al. (2022) integrated decades of sightings and survey effort data from multiple sources in a distribution model for giant manta ray off the eastern United States, including the Gulf of Mexico. The results indicate the species is most commonly detected at productive nearshore and shelf-edge upwelling zones at surface thermal frontal boundaries within a temperature range of approximately 15 to 30°C. Oceanic mantas in other parts of the world have exhibited reverse diel vertical migrations (i.e., ascend during the day, descent at night), which was attributed to such thermal fronts (Andrzejaczek et al., 2021).

Giant manta rays occur in the Northeast U.S. Continental Shelf Large Marine Ecosystem, where they feed on plankton in the upwelling region from Cape Hatteras to the Gulf of Maine. Farmer et al. (2022) predicted high nearshore concentrations of giant manta ray off northeast Florida during April, with the distribution extending northward along the shelf-edge as temperatures warm. Seasonal movements were predicted to result in higher numbers north of Cape Hatteras, North Carolina from June to October, and then south to Savannah, Georgia, from November to March as temperatures cool. The Loop Current, which is created by oceanic waters entering the Gulf of Mexico from the Yucatan channel and exiting through the Straits of Florida, has upwelling along its edges, in its rings, and in associated eddies (Heileman & Rabalais, 2008). These rings, eddies, and upwelling zones are areas where giant manta rays may feed. In the Gulf of Mexico, Farmer et al. (2022) predicted the highest nearshore concentrations of giant manta ray near the Mississippi River delta from April to June and again from October to November. In the Caribbean Sea Large Marine Ecosystem, areas of high productivity include localized regions of upwelling and nearshore habitats like coral reefs, mangroves, and seagrass beds (Heileman & Mahon, 2008). These areas attract giant manta rays for feeding. In addition, manta rays occur at cleaning stations on coral reefs where fishes remove parasites from them (Miller & Klimovich, 2016).

F.4.2.6.3 Population Trends

No stock assessments exist for the giant manta ray, and information on population sizes is lacking. Most estimates of subpopulations are based on anecdotal observations by divers and fishermen, with current populations estimated between 100 and 1,500 individuals (Miller & Klimovich, 2016). Regional populations are thought to be geographically fragmented and sparsely distributed. Giant manta rays are generally solitary but aggregate at cleaning sites and to feed and mate. They reach maturity at age 10 and have one pup every two to three years (National Marine Fisheries Service, 2022d). Although there is uncertainty regarding range-wide abundance, the best available information indicates that the species has experienced potentially significant population declines because of fisheries-related mortality (Miller & Klimovich, 2016).

F.4.2.6.4 Predator and Prey Interactions

Manta rays feed primarily on zooplankton and may also consume small- to moderate-sized fishes (Miller & Klimovich, 2016). The gill plates of the giant manta ray filter the water as they swim, straining out any plankton or other prey items (Defenders of Wildlife, 2015a). Because of their size, the only predators of giant manta rays are thought to be large sharks and toothed whales (e.g., bull sharks [*Carcharhinus leucas*] and killer whales [*Orcinus orca*]).

F.4.2.6.5 Species-Specific Threats

Threats to giant manta rays include fisheries and bycatch, destruction or modification of habitat, and disease and predation. The international market highly values the gill plates of the giant manta ray for use in traditional medicines. They also trade their cartilage and skins and consume the manta ray meat or use it for bait. Bycatch occurs in purse seine, gillnet, and trawl fisheries as well (National Marine Fisheries Service, 2022d). Fisheries exist outside the Study Area in Indonesia, Sri Lanka, India, Peru, Mexico, China, Mozambique, and Ghana (Food and Agriculture Organization of the United Nations, 2013). Other potential threats include degradation of coral reefs, interaction with marine debris, marine pollution, and boat strikes (Food and Agriculture Organization of the United Nations, 2013).

F.4.2.7 Nassau Grouper (*Epinephelus striatus*)

F.4.2.7.1 Status and Management

The Nassau grouper was previously one of the most common groupers in U.S. waters. The species is currently scarce due primarily to overfishing. NMFS listed the Nassau grouper as threatened under the ESA throughout its range in 2016.

In January 2024, NMFS designated critical habitat for the Nassau grouper (refer to Figure 3.6-6, Critical Habitat for ESA-Listed Nassau Grouper Designated in the Study Area). The designation consists of about 2,385 square km of aquatic habitat in waters off the coasts of southeastern Florida, Puerto Rico, Navassa, and the U.S. Virgin Islands. NMFS identified the following physical and biological features considered essential to the conservation of Nassau grouper:

- nearshore to offshore areas necessary for recruitment, development, and growth, containing a variety of benthic types that provide cover from predators and habitat for prey, consisting of:

    (1) nearshore shallow subtidal marine nursery areas with substrate that consists of unconsolidated calcareous medium to very coarse sediments and shell and coral fragments and may also include cobble, boulders, whole corals and shells, or rubble mounds, to support larval settlement and provide shelter from predators during growth and habitat for prey
    - (2) intermediate hard bottom and seagrass areas in close proximity to the nearshore shallow subtidal marine nursery areas that provide refuge and prey resources for juvenile fish (The areas include seagrass interspersed with areas of rubble, boulders, shell fragments, or other forms of cover; inshore patch and fore reefs that provide crevices and holes; or substrates interspersed with scattered sponges, octocorals, rock and macroalgal patches, or stony corals.)
    - (3) offshore linear and patch reefs in close proximity to intermediate hard bottom and seagrass areas that contain multiple benthic types, for example, coral reef, colonized hard bottom, sponge habitat, coral rubble, rocky outcrops, or ledges, to provide shelter from predation during maturation and habitat for prey
    - (4) structures between the subtidal nearshore area and the intermediate hard bottom and seagrass area and the offshore reef area including overhangs, crevices, depressions, blowout ledges, holes, and other types of formations of varying sizes and complexity to support juveniles and adults as movement corridors that include temporary refuge that reduces predation risk as Nassau grouper move from nearshore to offshore habitats
- marine sites used for spawning and adjacent waters that support movement and staging associated with spawning

Approximately 800 acres (3.2 km^2) of proposed critical habitat overlaps waters (generally 50 yards from shore) included in the Naval Air Station Key West Integrated Natural Resources Management Plan. Within this area, juvenile Nassau grouper and habitats identified as essential features (nearshore shallow subtidal marine nursery, intermediate hard bottom, and seagrass areas in close proximity to the nearshore nursery areas) may be present. NMFS determined that the Integrated Natural Resources Management Plan provides benefits to Nassau grouper and areas included in the proposed critical habitat. Therefore, pursuant to section 4(a)(3)(B)(i) of the ESA, NMFS determined that areas covered by the Integrated Natural Resources Management Plan are ineligible for designation as critical habitat for Nassau grouper.

F.4.2.7.2 Habitat and Geographic Range

Nassau grouper is a long-lived, late-maturing perch-like bony fish. This species is a solitary fish apart from spawning aggregations (Starr et al., 2007). These fish inhabit high-relief coral reefs and rocky bottoms from nearshore to a depth of 100 m and rest on or near the bottom, with juveniles inhabiting macroalgae and seagrass beds and patch reefs (Bester, 2012). Larvae are pelagic, floating with ocean currents for one to two months. This species also occupies caves and large overhangs (National Marine Fisheries Service, 2015a). Spawning aggregation sites are typically located near significant geomorphological features, such as projections of the reef as little as 50 m from the shore (81 *Federal Register* 42268).

Nassau grouper congregate in large numbers at specific areas to spawn after the appropriate water temperature and moon phase cues (usually within a period of 10 days overlapping the full moon) between November and February (Archer et al., 2012; National Marine Fisheries Service, 2015a, 2022g; Semmens et al., 2006). Spawning aggregations of several thousand individuals have been reported (Bester, 2012).

The geographic range within the Study Area is limited to around Bermuda, the southeast coast of Florida (Cape Canaveral to Florida Keys), Florida Straits (ocean corridor connecting the Gulf of Mexico with the U.S. Southeast Continental Shelf), Dry Tortugas National Park, Pulley Ridge, Florida Bay, Flower Garden Banks, and areas around Puerto Rico (Bester, 2012; National Marine Fisheries Service, 2013c). Within the Study Area, spawning aggregations have only been documented around Bermuda, Puerto Rico and the U.S. Virgin Islands (National Marine Fisheries Service, 2013c).

F.4.2.7.3 Population Trends

Based on the size and number of spawning aggregations, the total Nassau grouper population appears to be a small fraction of its historical size (National Marine Fisheries Service, 2022g). The worldwide population is thought to have declined at one time to as few as approximately 10,000 individuals (Cornish & Eklund, 2003). More recent data indicate that abundance may be increasing in some areas. The number of individuals observed at a spawning aggregation site near St. Thomas (U.S. Virgin Islands) increased by about 87 percent between 2004 and 2009, although the total number was relatively low (118 fish in 2009) (Kadison et al., 2010). The number of individuals at a spawning site off Little Cayman Island increased from about 2,000 fish to 5,223 fish between 2003 and 2018 (Waterhouse et al., 2020). Research has shown strong genetic differentiation in subpopulations in the Caribbean that may correlate to larvae dispersal barriers (Jackson et al., 2014).

F.4.2.7.4 Predator and Prey Interactions

Nassau groupers are preyed upon by barracuda (*Sphyraena barracuda*), king mackerel (*Scomberomorus cavalla*), moray eels (*Gymnothorax* spp.), sandbar sharks (*Carcharhinus plumbeus*), great hammerhead sharks (*Sphyrna mokarran*), and other large groupers including Nassau groupers (Bester, 2012).

Adult Nassau groupers are opportunistic ambush predators, feeding on a variety of fishes, shrimps, crabs, lobsters, and octopuses (Sadovy & Eklund, 1999). Adults have been observed feeding on the invasive lionfish in the Caribbean and are currently being studied as a potential biocontrol option (Mumby et al., 2011). Nassau grouper larvae are filter and particulate feeders that prey on dinoflagellates, fish larvae, and mysids (Sadovy & Eklund, 1999).

F.4.2.7.5 Species-Specific Threats

Nassau grouper is sensitive to overexploitation because of the slow growth rate, late reproduction age (five-plus years), large size, and long lifespan (Morris et al., 2000; Sadovy & Eklund, 1999). The decline in population is the result of overharvest and collapse of spawning aggregations (Aguilar-Perera, 2006; Ehrhardt & Deleveaux, 2007) and is exacerbated by coastal development (Stallings, 2009) and lack of effective regulations in some areas (National Marine Fisheries Service, 2022g). Commercial and recreational landings declined in both pounds landed and average fish size from 1986 and 1991. As a result, moratoriums on take and possession were established in 1990 (National Marine Fisheries Service, 2013c). By 2000, abundance had decreased approximately 60 percent over the last three generations (Cornish & Eklund, 2003).

Damage to spawning sites limits reproductive success of adults if alternative habitats are unavailable. Loss of macroalgae and seagrass beds is damaging to Nassau grouper populations, as it often results in low recruitment rates (Sadovy & Eklund, 1999).

Fishing moratoriums have been ineffective at preventing illegal harvest that occurs in Puerto Rico and other U.S. waters. Declines have also resulted from overfishing with spear guns and bycatch of juvenile in fine mesh nets (National Marine Fisheries Service, 2015a).

The marine isopod *Excorallana tricornis* is a known parasite of the Nassau grouper, sometimes resulting in infestations immediately following spawning (Semmens et al., 2006).

F.4.2.8 Oceanic Whitetip Shark (*Carcharhinus longimanus*)

F.4.2.8.1 Status and Management

NMFS completed a comprehensive status review of the oceanic whitetip shark and, based on the best scientific and commercial information available, including the status review report (Young et al., 2016), proposed that this species warrants listing as a threatened species under the ESA (81 *Federal Register* 96304). On January 30, 2018, NMFS published the Final Rule listing this species as threatened. On March 5, 2020, NMFS concluded that designation of critical habitat is not prudent because there are no identifiable physical or biological features that are essential to the conservation of the oceanic whitetip shark within areas under U.S. jurisdiction, and that there are no areas outside of the geographical area occupied by the species under U.S. jurisdiction that are essential to its conservation (85 *Federal Register* 12898).

F.4.2.8.2 Habitat and Geographic Range

Oceanic whitetip sharks are found worldwide in warm tropical and subtropical waters between the 20° North and 20° South latitude near the surface of the water column (Young et al., 2016). In the Western Atlantic, oceanic whitetips occur from Maine to Argentina, including the Caribbean and Gulf of Mexico. This species has a clear preference for open-ocean waters, with abundances decreasing nearer to continental shelves. However, individuals are occasionally found in nearshore waters (Rigby et al., 2019). Preferring warm waters near or over 20°C (68°F), and offshore areas, the oceanic whitetip shark is known to undertake seasonal movements to higher latitudes in the summer (National Marine Fisheries Service, 2022i) and may regularly survey extreme environments (deep depths, low temperatures) when foraging (Young et al., 2016). The presence of oceanic whitetip sharks increases further away from the continental shelf in deep-water areas, but the species prefers to inhabit the surface waters in deep-water areas at less than 328 ft. (Defenders of Wildlife, 2015b). Within this habitat, tagged oceanic whitetip sharks have exhibited average daily ascents to 10 to 35 m at dawn and dusk, followed by descents to 25 to 45 m during the day and 30 to 60 m at night (Tolotti et al., 2017). The variability is attributed to thermoregulation and depth of the mixing zone. The oceanic whitetip shark has declined in the northwest Atlantic, western central Atlantic, and Gulf of Mexico (Baum et al., 2015). The species also occurs in waters of the Southeast U.S. Continental Shelf, Caribbean Sea, and Newfoundland-Labrador Shelf Large Marine Ecosystems.

F.4.2.8.3 Population Trends

Globally, catches of oceanic whitetip sharks generally peaked between 1995 and 2000, followed by precipitous declines over next 10 years (Young & Carlson, 2020). In the Atlantic region from the 1990s to the early 2000s, the oceanic whitetip shark had declined by an estimated 9 to 70 percent, depending on the data source and area. The most significant decline reported was a 99.9 percent decrease in abundance in the Gulf of Mexico based on a comparison of longline research surveys from 1954 to 1957 and data from fisheries observers from 1995 to 1999. However, because of temporal changes in fishing gear and practices over the time period, the study may have exaggerated or underestimated the magnitude of population decline (Baum et al., 2015).

F.4.2.8.4 Predator and Prey Interactions

As one of the major apex predators in the tropical open ocean waters, the oceanic whitetip shark feeds primarily on fishes and cephalopods, although they also feed on birds, marine mammals, mollusks, and crustaceans.

F.4.2.8.5 Species-Specific Threats

Threats include pelagic longline and drift net fisheries bycatch, targeted fisheries (for the shark fin trade), and destruction or modification of its habitat and range (Baum et al., 2015; Defenders of Wildlife, 2015b). Legal and illegal fishing activities in the Atlantic have caused significant population declines for the oceanic whitetip shark. It is caught as bycatch in tuna and swordfish longlines in the northwest Atlantic and Gulf of Mexico. Habitat degradation has occurred because of pollutants that bioaccumulate and biomagnify to high levels in their bodies as a result of their high position in the food chain, long life, and large size (Defenders of Wildlife, 2015b).

F.4.2.9 Scalloped Hammerhead Shark (*Sphyrna lewini*)

F.4.2.9.1 Status and Management

The Central and Southwest Atlantic distinct population segment of the scalloped hammerhead shark is listed as threatened under the ESA (79 *Federal Register* 38213, July 3, 2014). The Study Area does not coincide with any other distinct population segments. There are no designated critical habitat marine areas within the jurisdiction of the United States. In 2020, NOAA Fisheries completed a 5-year review of the scalloped hammerhead shark and determined there was no reason to change the status of the species or any of the four distinct population segments (National Marine Fisheries Service, 2020c).

F.4.2.9.2 Habitat and Geographic Range

The scalloped hammerhead shark is a coastal and semi-oceanic species distributed in temperate to tropical waters (National Marine Fisheries Service, 2020c). In the western Atlantic, their range extends from New Jersey to points south of the Study Area, including the Gulf of Mexico and the Caribbean Sea (Bester, 1999) with seasonal migration along the eastern United States. Juveniles rear in coastal nursery areas (Duncan & Holland, 2006) with all ages occurring in the Gulf Stream, but rarely inhabits the open ocean (Kohler & Turner, 2001). Scalloped hammerhead sharks that are part of the Central and Southwest Atlantic distinct population segment are only found in the southernmost portion of the Study Area in the vicinity of Puerto Rico. Scalloped hammerhead sharks that occur in other portions of the Study Area are not protected under the ESA.

Scalloped hammerhead sharks inhabit the surface to depths of 275 m (Duncan & Holland, 2006) or more (Daly-Engel et al., 2012). Coastal waters with temperatures between 23 and 26°C are preferred habitats (Castro, 1983; Compagno, 1984), with animals generally remaining close to shore during the day and moving into deeper waters to feed at night (Bester, 1999). Ketchum et al. (2014b) found scalloped hammerheads formed daytime schools at specific locations in the Galapagos Islands, but dispersed at night, spending more time at the northern islands during part of the warm season (December to February) compared with the cool season. Ketchum et al. (2014a) used acoustic telemetry to show that scalloped hammerheads were highly selective of location (i.e., habitat on up-current side of island) and depth (i.e., top of the thermocline) while refuging, where they may carry out essential activities such as cleaning and thermoregulation, and also perform exploratory vertical movements by diving the width of the mixed layer and occasionally diving below the thermocline while moving offshore, most likely for foraging. More recent analysis of satellite-tracked sharks, including scalloped hammerheads, in the Atlantic, Indian, and Pacific Oceans indicate the species occupies predictable habitat hotspots (Queiroz et al., 2019; Queiroz et al., 2016); scalloped hammerheads preferred fronts characterized by strong sea surface temperature and high productivity gradients (e.g., Gulf Stream edge). Hoffmayer et al. (2013) also found that tagged sharks exhibited consistent and repeated diel vertical movement patterns, making more than 76 deep nighttime dives to a maximum depth of 964 m, possibly representing feeding behavior. A genetic marker study suggests that females remain close to coastal habitats, while males disperse across larger open-ocean areas (Daly-Engel et al., 2012). However, females in the northern Gulf of Mexico exhibited more shelf-edge (greater than 200-m depth) occurrence than males, which displayed more mid-shelf use (Wells et al., 2018).

In the southernmost portion of the Study Area, scalloped hammerhead sharks may exhibit habitat partitioning with other shark species. In tropical areas, the scalloped hammerhead was more associated with deep (70 to 80 m) seamounts, whereas other sharks occupied shallower (less than 40 m) reefs and lagoons (Tickler et al., 2017). The nursery habitat for this distinct population segment, and for the species in general, is currently unknown. However, based on surveys of Pacific distinct population segments, it likely includes river deltas and nearshore reefs (National Marine Fisheries Service, 2020c). The species likely spawns in the open water throughout its range (National Marine Fisheries Service, 2020c).

F.4.2.9.3 Population Trends

The scalloped hammerhead shark has undergone substantial declines throughout its range (Baum et al., 2003). There is some evidence of population increase in the Southeast U.S. Continental Shelf Large Marine Ecosystem (Ward-Paige et al., 2012). Globally, landings of scalloped hammerhead sharks peaked at 8,000 metric tons in 2002 and declined to 1,000 metric tons in 2009 (Food and Agriculture Organization of the United Nations, 2005, 2009). Modeling results estimate the overall population range from approximately 142,000 to 169,000 individuals in 1981 and between 24,000 and 28,000 individuals in 2005 (Miller et al., 2013).

There is some evidence of population increase in the Southeast U.S. Continental Shelf Large Marine Ecosystem (Ward-Paige et al., 2012). The current 5-year status review also noted that analysis of more recent data (1994 to 2017) from the North Atlantic and Gulf of Mexico showed a population increase after fisheries management measures were enacted (National Marine Fisheries Service, 2020c). For the Central and Southwest distinct population segment, there was an estimated 62.7 percent decline in catch per unit effort of hammerheads from longline vessels operating in the South Atlantic from 1998 to 2007 (National Marine Fisheries Service, 2020c). This also applies to the Central and Southwest distinct population segment because most of the South Atlantic fleet, including mostly South American vessels, overlapped with the Central and Southwest Atlantic distinct population segment.

F.4.2.9.4 Predator and Prey Interactions

Scalloped hammerhead sharks have few predators. The most recent 5-year review concluded that predation is not considered a stressor outside of human consumption and use (National Marine Fisheries Service, 2020c). Sharks locate potential prey by odor, particularly from injured prey, or low-frequency sounds, inner ear (vibrations), lateral line (turbulence) with vision coming into play at closer range (Moyle & Cech, 2004). They feed primarily at night (Compagno, 1984) on a wide variety of fishes such as sardines, herring, anchovies, and jacks, and also feed on invertebrates, including squid, octopus, shrimp, crabs, and lobsters (Bester, 1999).

F.4.2.9.5 Species-Specific Threats

The primary threat to scalloped hammerhead populations is mortality from commercial, recreational, and artisanal longline fisheries, including at-vessel mortality (i.e., finning and bycatch) (Miller et al., 2014). This species is highly susceptible to bycatch because of schooling habits (Food and Agriculture Organization of the United Nations, 2012). For the Central and Southwest Atlantic distinct population segment, longline fisheries off South America reported up to approximately four percent scalloped hammerhead of the total shark catch (many of which were juveniles) (Feitosa et al., 2018). However, none of the longline fishing trips observed in the U.S. shark fishery coincide with the area where the Central and Southwest Atlantic distinct population segment occurs (National Marine Fisheries Service, 2020c). Of the 12 shark species observed in shark bycatch, the scalloped hammerhead was considered the most vulnerable to bycatch (Gallagher et al., 2014).

F.4.3 SPECIES NOT LISTED UNDER THE ENDANGERED SPECIES ACT

The background information for fish species not listed under the ESA in the Study Area as described in the 2018 Final EIS/OEIS Section 3.6.2.3 (Species Not Listed Under the Endangered Species Act) has not appreciably changed. As such, the information presented in the 2018 Final EIS/OEIS remains valid.

F.5 MARINE MAMMALS

F.5.1 GENERAL BACKGROUND

The following topics were updated from the 2018 Final EIS/OEIS:

- Vessel strike and climate change information was updated in Section F.5.1.5.
- Unusual Mortality Event declared for North Atlantic right whales (*Eubalaena glacialis*) in Canada and the United States.
- NMFS revised the taxonomy and common name for the Gulf of Mexico subspecies of Bryde’s whale (*Balaenoptera edeni*) under the ESA. The revision classifies this new species as Rice’s whales (*Balaenoptera ricei*).
- A Federal Register Notice by the USFWS announced the initiation of a 5-year status review for polar bears under the ESA.
- NMFS published the proposed amendments to North Atlantic right whale vessel speed restrictions in the Federal Register as an effort to reduce vessel-whale collision potential.
- NMFS published the Proposed Rule for Critical Habitat in the Gulf of Mexico for Rice’s Whales in the Federal Register.
- USFWS published 90-day findings for two petitions to reclassify the West Indian manatee under the ESA, and announced the initiation of a status review.

- The National Oceanic and Atmospheric Administration, with support of the U.S. Navy, is in the process of updating and revising Biologically Important Areas using new methodology and scoring criteria. Final Biologically Important Area designations and manuscripts for the East Coast and Gulf of Mexico are currently pending.

Marine mammals are a diverse group of approximately 132 species (Committee on Taxonomy, 2023). Four main types of marine mammals are recognized: cetaceans (whales, dolphins, and porpoises), pinnipeds (seals, sea lions, and walruses), sirenians (manatees, dugongs, and sea cows), and other marine carnivores (sea otters and polar bears) (Jefferson et al., 2008; Rice, 1998). The order Cetacea is divided into two suborders – Odontoceti and Mysticeti. The different feeding strategies between mysticetes and odontocetes affect their distribution and occurrence patterns (Goldbogen et al., 2015). The toothed whales, dolphins, and porpoises (suborder Odontoceti) range in size from slightly longer than 3.3 ft. (1 m) to more than 60 ft. (18 m) and have teeth, which they use to capture and consume individual prey. The baleen whales (suborder Mysticeti) are universally large (more than 15 ft. [5 m] as adults). They are called baleen whales because, instead of teeth, they have a fibrous structure along the upper jaw made of keratin, a type of protein similar to that found in human fingernails, which enables them to filter or extract food from the water for feeding. They are batch feeders that use baleen instead of teeth to engulf, suck, or skim large numbers of prey, such as small schooling fish, shrimp, or microscopic sea animals (i.e., plankton) from the water or out of ocean floor sediments (Heithaus & Dill, 2009). Mysticetes are further divided into four families, two of which are found in the Study Area (right whales and rorquals) and two that are not found within the Study Area (gray whales and pygmy right whales). Rorquals have a series of longitudinal folds of skin, often referred to as throat grooves, running from below the mouth back toward the navel. Rorquals are slender and streamlined in shape, compared with their relatives the right whales, and most have narrow, elongated pectoral flippers. Detailed reviews of the different groups of cetaceans can be found in Perrin et al. (2009).

Pinnipeds are divided into three families: Phocidae (true seals), Otariidae (fur seals and sea lions), and Odobenidae (walrus). Most marine mammal species live in a marine habitat, though pinnipeds, sea otters, and polar bears spend time in terrestrial habitats, while manatees and certain dolphin species sometimes occupy freshwater habitats (Jefferson et al., 2015; Rice, 1998). The exact number of formally recognized marine mammal species changes periodically with new scientific understanding or findings (Rice, 1998). For a list of current species classifications, see the formal list *Marine Mammal Species and Subspecies* maintained online by the Society for Marine Mammalogy (Committee on Taxonomy, 2023).

All marine mammals in the United States are protected under the Marine Mammal Protection Act (MMPA), while select species also receive additional protection under the ESA. Within the framework of the MMPA, a marine mammal "stock" is defined as "a group of marine mammals of the same species or smaller taxon (subspecies) in a common spatial arrangement that interbreed when mature" (16 United States Code [U.S.C.] section 1362). Per NMFS guidance, "for purposes of management under the MMPA, a stock is recognized as being a management unit that identifies a demographically independent biological population" (National Marine Fisheries Service, 2016a). However, in practice, recognized management stocks may fall short of this ideal because of a lack of information or, in some cases, stocks may even include multiple species in a management unit.

The ESA provides for listing species, subspecies, or distinct population segments of species, all of which are referred to as "species" under the ESA. The Interagency Policy Regarding the Recognition of Distinct Vertebrate Population Segments under the ESA (61 *Federal Register* 4722, February 7, 1996) defines a distinct population segment as, "any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature." If a population meets the criteria to be identified as a distinct population segment, it is eligible for listing under the ESA as a separate species (National Marine Fisheries Service, 2016a). However, MMPA stocks do not necessarily coincide with distinct population segments under the ESA (81 *Federal Register* 62660, September 8, 2016).

For summaries of the general biology and ecology of marine mammals beyond the scope of this section, see Rice (1998), Twiss and Reeves (1999), Hoelzel (2002), Berta et al. (2006), (Jefferson et al., 2008), Jefferson et al. (2015), and Committee on Taxonomy (2008). Additional species profiles and information on the biology, life history, distribution, and conservation of marine mammals can also be found through the following organizations:

- NMFS Office of Protected Resources (includes species distribution maps)
- Ocean Biogeographic Information System-Spatial Ecological Analysis of Megavertebrate Populations (known as OBIS-SEAMAP) species profiles
- National Oceanic and Atmospheric Administration Cetacean Density and Distribution Mapping Working Group
- International Whaling Commission
- International Union for Conservation of Nature, Cetacean Specialist Group
- Marine Mammal Commission
- Society for Marine Mammalogy

Detailed reviews of the different groups of cetaceans can be found in Jefferson et al. (2015), Heithaus and Dill (2009), and Perrin and Geraci (2002).

Mammal species that are extralimital to the Study Area, including bowhead whale, narwhal, beluga whale, ringed seal, bearded seal, walrus, and polar bear, are unlikely to be exposed to stressors under the Proposed Action and are not included in the analysis for this Supplemental EIS/OEIS (See 2018 Final EIS/OEIS for additional information on these species).

F.5.1.1 Group Size

Many species of marine mammals, particularly odontocetes, are highly social animals that spend much of their lives in groups called “pods.” The sizes and structures of these pods are dynamic and, based on the species, can range from several individuals to several thousand individuals. Similarly, aggregations of mysticete whales may form during particular breeding or foraging seasons, although they do not persist through time as a social unit. Marine mammals that live or travel in groups are more likely to be detected by observers, and group size characteristics are incorporated into many of the density and abundance calculations. Group size characteristics are also incorporated into acoustic effects modeling to represent a more realistic patchy distribution for the given density estimates. The behavior of aggregating into groups is also important for the purposes of mitigation and monitoring, since animals that occur in larger groups have an increased probability of being detected. A comprehensive and systematic review of relevant literature and data was conducted using available published and unpublished literature, including journals, books, technical reports, survey cruise reports; and raw data from cruises, theses, and dissertations.

F.5.1.2 Habitat Use

Marine mammals occur in every marine environment in the Study Area, from coastal and inshore waters to the open ocean. Their distribution is influenced by multiple factors, primarily patterns of major ocean currents, bottom relief, water temperature, water depth, and salinity, which, in turn, affect prey distribution and productivity. The continuous movement of water from the ocean bottom to the surface creates a nutrient-rich, highly productive environment for marine mammal prey in upwelling zones (Jefferson et al., 2015). For most cetaceans, prey distribution, abundance, and quality largely determine where they occur at any specific time (Heithaus & Dill, 2009). Most of the baleen whales are migratory, but many of the toothed whales do not migrate in the strictest sense. Instead, they undergo seasonal dispersal or shifts in density distribution and occupy habitats preferable for feeding, breeding, and other important behaviors. Pinnipeds occur mostly in coastal habitats or over continental shelves, while manatees and polar bears are strongly associated with coastal waters as habitat for reproducing, resting, and, in some cases, feeding, though polar bears can also range far offshore.

In 2011, the National Oceanic and Atmospheric Administration convened a working group to map cetacean density and distribution within U.S. waters (Ferguson et al., 2015; National Oceanic and Atmospheric Administration, 2022). The specific objective of the Cetacean Density and Distribution Mapping Working Group was to create comprehensive and easily accessible regional cetacean density and distribution maps that are time and species specific. Separately, to augment this more quantitative density and distribution mapping and provide additional context for marine mammal impact analyses, the Cetacean Density and Distribution Mapping Working Group also identified (through literature search, current science compilation, and expert elicitation) areas of importance for cetaceans, such as reproductive areas, feeding areas, migratory corridors, and areas in which small or resident populations are concentrated. Areas identified through this process have been termed as biologically important areas (Ferguson et al., 2015; Van Parijs, 2015).

It is important to note that biologically important areas were not meant to define exclusionary zones or serve as sanctuaries or marine protected areas and have no direct or immediate regulatory consequences. Ferguson et al. (2015) outlines the envisioned purpose for the biologically important area designations. The identification of biologically important areas is intended to be a “living” reference based on the best available science at the time, which will be maintained and updated as new information becomes available. As new empirical data are gathered, these referenced areas can be calibrated to determine how closely they correspond to the reality of a species’ habitat uses and updated as necessary (see (National Oceanic and Atmospheric Administration, 2022). This evolution may include new information that indicates a biologically important area is no longer important to an essential life function, or may show that a species may migrate to different areas due to environmental changes. Products of the initial assessment process were compiled and published in March 2015 (Ferguson et al., 2015; LaBrecque et al., 2015a, 2015b).

In 2015, eighteen biologically important areas were identified for seven species within the Study Area (LaBrecque et al., 2015a, 2015b): minke whales, sei whales, fin whales, North Atlantic right whales, humpback whales, harbor porpoises, and bottlenose dolphins. The National Oceanic and Atmospheric Administration is currently in the process of updating biologically important areas for cetaceans based on a new application and scoring system by Harrison et al., (Harrison et al., 2023), and this document will be updated with those areas as soon as they are made available for the East Coast and Gulf of Mexico regions.

Biologically important areas are considered when developing mitigation measures because they are important areas for feeding, reproduction, and migration; or because they have a small, resident population within the Study Area.

F.5.1.3 Dive Behavior

Most marine mammals spend a considerable portion of their lives underwater while traveling or feeding. Some species of marine mammals have developed specialized adaptations to allow them to make deep dives lasting over an hour, primarily for foraging on deep-water prey such as squid. Other species spend the majority of their lives close to the surface and make relatively shallow dives. The diving behavior of a particular species or individual has implications for an observer's ability to detect them for purposes of mitigation and monitoring. In addition, their relative distribution through the water column is an important consideration when conducting acoustic exposure and direct strike analyses. Information and data on diving behavior for each marine mammal species were compiled and summarized in a technical report (U.S. Department of the Navy, 2017) that provides estimates of time at depth based on available research. The dive data and group size information compiled in this technical report was incorporated into the Navy acoustic effects modeling.

F.5.1.4 Hearing and Vocalization

Refer to Appendix D (Acoustic and Explosive Impacts Supporting Information) for summary and details regarding the hearing and vocalization of marine mammals.

F.5.1.5 General Threats

Marine mammal populations can be influenced by various natural factors and human activities. There can be direct effects, such as from disease or activities such as hunting and whale watching, or indirect effects, such as reduction in prey availability or lowered reproductive success of individuals. Twiss and Reeves (1999) and the National Marine Fisheries Service (2011a) provide a general discussion of marine mammal conservation and the threats they face. New research published on threats to marine mammals does not dispute information in the 2018 Final EIS/OEIS, but does provide verification of emerging threats, and updates on existing threats. The information in the 2018 Final EIS/OEIS remains valid for qualitative effects analysis of general threats to marine mammals.

F.5.1.5.1 Vessel Strike

Ship strikes are a growing issue for most large marine mammals and mortality is a major concern for endangered species, especially those that occupy areas with high levels of vessel traffic, because the likelihood of vessel strike is greater (Currie et al., 2017; Van der Hoop et al., 2013; Van der Hoop et al., 2015). Marine mammals that spend increased amounts of time at or near the surface face an increased risk of a collision. Vessel strike could result in debilitating or fatal impacts that include blunt force injuries (fractures, abrasions, and contusions) or sharp force injuries (cuts, gashes, and lacerations) (Campbell-Malone et al., 2008; Hill et al., 2017; Rommel et al., 2007; Schoeman et al., 2020). Collision risk is also increased for species inhabiting coastal waters of urbanized areas with increased traffic of small- and medium-sized vessels (Neilson et al., 2012).

The reasons that marine mammals are struck by vessels are likely several fold, including acoustic shadowing of an approaching vessel (i.e., the vessel body interferes with sound from the propulsion unit propagating forward), complex acoustic propagation conditions (Gerstein et al., 2005), and vessels approaching at speeds where avoidance is difficult (Martin et al., 2015; McKenna et al., 2015). Additionally, research has indicated the risk of an animal-vessel collision has a greater likelihood of occurrence with increases vessel traffic transiting through areas containing high densities of marine mammals (Cates et al., 2017; Redfern et al., 2019; Rockwood et al., 2017). However, no single factor exists to explain differences in vessel collision risk between species or individuals, and further species-specific research is necessary to better quantify this variability (Schoeman et al., 2020).

The severity of injuries is typically dependent on the size and speed of the vessel (Knowlton & Kraus, 2001; Laist et al., 2001; Vanderlaan & Taggart, 2007). An examination of all known ship strikes from all shipping sources (civilian and military) indicates vessel speed is a principal factor in whether a strike is fatal (Jensen & Silber, 2003; Knowlton & Kraus, 2001; Laist et al., 2001; Vanderlaan & Taggart, 2007). In assessing records in which vessel speed was known, Laist et al. (2001) found a direct relationship between the occurrence of a whale strike and the speed of the vessel involved in the collision. The authors concluded that most fatalities occurred when a vessel was traveling more than 13 knots. Jensen and Silber (2003) detailed 292 records of known or probable ship strikes of all large whale species from 1975 to 2002. Of these, vessel speed at the time of collision was reported for 58 cases. Of these cases, 39 (or 67 percent) resulted in serious injury or death. Operating speeds of vessels that struck various species of large whales ranged from 2 to 51 knots. The majority (79 percent) of these strikes occurred at speeds of 13 knots or greater. The average speed that resulted in serious injury or death was 18.6 knots (Jensen & Silber, 2003). Pace and Silber (2005) found that the probability of death or serious injury increased rapidly with increasing vessel speed. Specifically, the predicted probability of serious injury or death increased from 45 percent to 75 percent as vessel speed increased from 10 to 14 knots and exceeded 90 percent at 17 knots. Higher speeds during collisions result in greater force of impact, but higher speeds also appear to increase the chance of severe injuries or death by pulling whales toward the vessel. Computer simulation modeling showed that hydrodynamic forces pulling whales toward the vessel hull increase with increasing speed (Clyne et al., 1999; Knowlton et al., 1995; Silber et al., 2010).

In the Northwest Atlantic, vessel strikes are thought to be a leading cause of anthropogenic mortality in humpback whales, North Atlantic right whales, fin whales, and other large whale species (Hayes et al., 2022; Stepanuk et al., 2021). Since 2017, ongoing Unusual Mortality Events, also known as a mass die-off, have been declared for the North Atlantic right whales (National Marine Fisheries Service, 2024d) and humpback whales along the Atlantic coast in response to an increase in strandings (National Marine Fisheries Service, 2024b). While not every deceased individual was able to be examined post-mortem, a large proportion of those that were documented suggest pre-mortem entanglement or vessel strike to be the preliminary cause of mortality (Hayes et al., 2023; National Marine Fisheries Service, 2024d).

In August of 2022, NMFS proposed changes to the North Atlantic right whale vessel speed regulations (87 *Federal Register* 46921). Anthropogenic-induced trauma for both adults and juveniles was the leading cause of mortality from 2003 to 2018 (Sharp et al., 2019), and mortality rates continue to outnumber reproductive output, considering there are less than 100 reproductively active females in the population. Adjustments outlined in the Proposed Rule include: (1) spatial and temporal modifications of current speed restriction areas known as Seasonal Management Areas, (2) speed restrictions for vessels between 35 ft. (10.7 m) and 65 ft. (19.8 m), (3) the creation of a Dynamic Speed Zone Framework for the implementation of speed restrictions for the presence of whales outside of Seasonal Management Areas, and (4) updates to the speed rule's safety deviation process. These changes are considered essential to mitigate the ongoing species decline and further enhance conservation measures (87 *Federal Register* 46921).

F.5.1.5.2 Climate Change

As the state of the global climate warms, marine mammal populations are exposed to the ongoing effects of changing oceanic conditions, and is an emerging factor pertaining to marine mammal conservation and management (Baker et al., 2016; Fleming et al., 2016; Salvadeo et al., 2010; Shirasago-Germán et al., 2015; Silber et al., 2017; Simmonds & Eliott, 2009). The impacts of climate change have been documented to affect marine mammal species directly through habitat loss (especially for species that depend on ice or terrestrial areas) or gain, which may result in shifting distribution to match physiological tolerance under changing environmental conditions (Silber et al., 2017). Indirect effects to marine mammals include changes in prey quality and quantity, shifting prey distribution, as well as oceanographic variability (Gulland et al., 2022; van Weelden et al., 2021). Research by (Lettrich et al.,

2023) found that 72 percent of U.S. pinniped and cetacean populations in the western North Atlantic, Gulf of Mexico, and Caribbean Sea are considered highly susceptible to climate-induced ecological change. Specifically, exposure to changes in water temperature, pH, and dissolved oxygen are leading factors that indirectly increase marine mammal climate vulnerability in terms of prey distribution, availability, and nutritional quality, as well as acoustic propagation, and habitat preference (Lettrich et al., 2023). For example, North Atlantic right whales have exhibited a summertime divergence from traditional foraging habitats in the Gulf of Maine, Bay of Fundy, and Scotian Shelf. In response to climate-induced shifts in prey distribution, a sizeable portion of the population now feeds in the southern Gulf of St. Lawrence, which increases the whales' susceptibility to vessel strikes and gear entanglements (Meyer-Gutbrod et al., 2023; Meyer-Gutbrod et al., 2021; Record et al., 2019).

Species or populations with limited ranges, specialized diets, or similarly limiting ecological features may be particularly vulnerable to climate-induced ecosystem shifts (Baker et al., 2016), especially if species management relies on historical distributions (Record et al., 2019). In more northern latitudes, the loss of sea ice coverage and changing pack ice habitat are impacting marine mammals species that are dependent on ice for resting, foraging, and reproduction (Jay et al., 2012; Laidre et al., 2015; Rode et al., 2014). Changes in prey can influence marine mammal foraging success, which in turn affects reproductive success and survival. Warmer ocean temperatures may appear to benefit cold-sensitive marine species, such as the Florida manatee; however, findings suggest that major threats to manatee populations, including vessel strikes from increased vessel traffic and harmful algal blooms, would likely increase as a result of climate change (Edwards, 2013).

Harmful algal blooms may become more prevalent in warmer ocean temperatures with increased salinity levels, such that blooms will begin earlier, last longer, and cover a larger geographical range; with a projected potential to shift northward. (Edwards, 2013; Gobler, 2020; Moore, 2008; Townhill et al., 2018). Similarly, the neurotoxins produced from these blooms are known to have detrimental consequences on species across the marine food web including species of fish, marine mammals, invertebrates, and humans (Griffith & Gobler, 2020; Landsberg, 2002). For example, warming ocean waters have been linked to the spread of harmful algal blooms into the North Pacific where waters had previously been too cold for most of these algae to thrive. The spread of the algae and associated blooms has led to disease in marine mammals in locations where algae-caused diseases had not been previously known (Lefebvre et al., 2016).

Climate change may indirectly influence marine mammals through changes in human behavior, such as increased shipping and oil and gas extraction, which benefit from sea ice loss (Alter et al., 2010). Ultimately, impacts from global climate change may result in an intensification of current and ongoing threats to marine mammals (Edwards, 2013). In addition, the ability of marine mammals to alter behaviors may serve as a buffer against measurable climate change-induced impacts and could delay or mask any adverse effects until critical thresholds are reached (Baker et al., 2016).

Marine mammals are influenced by climate-related phenomena, such as storms and other extreme weather patterns such as the 2015 to 2016 El Niño in the ocean off the west coast of the United States. Generally, little is known about how large storms and other weather patterns affect marine mammals other than the fact that mass strandings (when two or more marine mammals become beached or stuck in shallow water) sometimes coincide with hurricanes, typhoons, and other tropical storms (Bradshaw et al., 2006; Marsh, 1989; Rosel & Watts, 2008) or other oceanographic conditions. There have also been correlations in time and space between strandings and the occurrence of earthquakes. However, there has been no scientific investigation demonstrating evidence for or against a relationship between earthquakes and the occurrence of marine mammal strandings. Indirect impacts may include altered water chemistry in estuaries (low dissolved oxygen or increased nutrient loading), causing quantifiable die-offs of fish species (Burkholder et al., 2004), as well as changing prey distribution and availability for cetaceans (Stevens et al., 2006). Human responses to extreme weather events may indirectly affect behavior and reproductive rates of marine mammals. For example, Miller et al. (2010) reported an increase in reproductive rates in bottlenose dolphins in the Mississippi Sound after Hurricane Katrina, presumably resulting from an increase in fish abundance due to a reduction in fisheries landings, a decrease in recreational and commercial boat activities (National Marine Fisheries Service, 2007a), and an increase in the number of reproductively active females available during the breeding seasons following the storm. Smith et al. (2013) supplemented the findings from this study and documented a marked increase in foraging activity in newly identified foraging areas that were observed during the two-year study period after the storm.

Habitat deterioration and loss is a major concern for almost all coastal and inshore species of marine mammals, with effects ranging from depleting a habitat’s prey base to the complete loss of habitat (Ayres et al., 2012; Kemp, 1996; Smith et al., 2009). Many researchers predict that if oceanic temperatures continue to rise with an associated effect on marine habitat and prey availability, then either changes in foraging or life history strategies, including poleward shifts in many marine mammal species distributions, should be anticipated (Alter et al., 2010; Fleming et al., 2016; Ramp et al., 2015; Salvadeo et al., 2010; Sydeman et al., 2015). Poloczanska et al. (2016) analyzed climate change impact data that integrate multiple climate-influenced changes in ocean conditions (i.e., temperature, acidification, dissolved oxygen, and rainfall) to assess anticipated changes to a number of key ocean fauna across representative areas.

In relation to the Study Area, the density of krill, an important prey item for marine mammals, has likely decreased in the southwest Atlantic because phytoplankton, a food source for krill, are also declining with warming temperatures and decreasing sea ice extent (Poloczanska et al., 2016). However, Poloczanska et al. (2016) also reports that zooplankton have displayed the highest rate of range expansion within the northeast Atlantic, supporting the general expectation that marine species will shift poleward within open oceans. On the other hand, for the northern Gulf of Mexico where coastlines prohibit poleward distributional shifts, marine species distributions, including fish and marine invertebrates, have displayed a depth shift toward cooler waters (Poloczanska et al., 2016). A similar marine mammal distributional response may occur based on observations made on select prey species, but marine mammal responses to climate change are currently unknown (Poloczanska et al., 2016).

Copepod distribution is also being affected by changing temperatures in the Northwest Atlantic. It is predicted that the North Atlantic ecosystem will become more favorable to carnivorous copepods in the next century while in contrast the prevalence of herbivorous copepods will diminish as waters become warmer (McGinty et al., 2021). Species turnover in copepod communities are predicted to range from 5 to 75 percent with the highest turnover rates concentrated in regions of pronounced temperature increases and decreases, such as the cooling, freshening area in the subpolar North Atlantic south of Greenland, and in an area of significant warming along the Scotian shelf (McGinty et al., 2021). Large copepods (greater than 2.5 mm), a crucial food source for whales, may have an advantage in the cooling waters due to their life-history strategy (McGinty et al., 2021). Deoxygenation, another consequence of climate change and warming seas, has also been shown to reduce copepod growth rates (Roman & Pierson, 2022).

F.5.2 ENDANGERED SPECIES ACT-LISTED SPECIES

As shown in Table 3.7-1 (Marine Mammal Occurrence within the Study Area), there are marine mammal species and applicable stocks listed under the ESA and occurring within in the Study Area.

F.5.2.1 North Atlantic Right Whale (*Eubalaena glacialis*)

F.5.2.1.1 Status and Management

The North Atlantic right whale is considered one of the most critically endangered populations of marine mammals in the world (Clapham et al., 1999; National Marine Fisheries Service, 2017). The size of this stock is considered extremely low relative to the Optimum Sustainable Population in the U.S. Atlantic Exclusive Economic Zone, and this species is listed as endangered under the ESA. A recovery plan for the North Atlantic right whale is in effect (National Marine Fisheries Service, 2005). The North Atlantic right whale has been protected from commercial whaling since 1949 by the International Convention for the Regulation of Whaling (62 Stat. 1716; 161 United Nations Treaty Series 72). Since the Convention's inception, the United States has remained a member nation under the Whaling Convention Act of 1949 (16 U.S.C. § 916 - 916l). An ESA status review by the National Marine Fisheries Service in 2017 concluded that the western North Atlantic stock remains endangered and has been declining since 2011 (Pace III et al., 2017). Relative to populations of southern right whales, there are also concerns about growth rate, percentage of reproductive females, and calving intervals in the North Atlantic right whale population. The total level of human-caused mortality and serious injury is unknown, but average annual detected (i.e., observed) human-caused mortality from 2014 through 2018 was 8.15 (Hayes et al., 2021; Henry et al., 2021). Any mortality or serious injury to individuals within this stock should be considered significant. This is a strategic stock because the average annual human-related mortality and serious injury rates exceed potential biological removal and because the North Atlantic right whale is an endangered species.

F.5.2.1.2 Habitat and Geographic Range

The western North Atlantic right whale population ranges primarily from calving grounds in coastal waters of the southeastern United States to summer feeding grounds in the Great South Channel, Jordan Basin, Georges Bank along its northeastern edge, Cape Cod and Massachusetts Bays, the Bay of Fundy, and the Roseway Basin on the Scotian Shelf. However, recent acoustic data suggests broad-scale use of the U.S. Eastern seaboard during much of the year (Davis et al., 2017). Movements within and between habitats are extensive. Telemetry data show lengthy and somewhat distant excursions, including into deep water off the continental shelf (Baumgartner & Mate, 2005; Mate et al., 1997).

LaBrecque et al. (2015a) identified three seasonal right whale feeding areas located in or near the Study Area (Figure F.5-1) based on vessel and aerial survey efforts: (1) February to April in Cape Cod Bay and Massachusetts Bay, (2) April to June in the Great South Channel and on the northern edge of Georges Bank, and (3) June to July and October to December on Jeffreys Ledge in the western Gulf of Maine. A potential mating area was identified in the central Gulf of Maine (from November through January) based on a demographic study of North Atlantic right whale habitats, and the migratory corridor area along the U.S. East Coast between the southern calving grounds and northern feeding areas. The migratory corridor was substantiated through vessel- and aerial-based survey data, photo-identification data, radio-tracking data, and expert judgment. Reproductive female North Atlantic right whales generally migrate south to calving grounds in November and December and migrate north to the feeding areas in March and April.

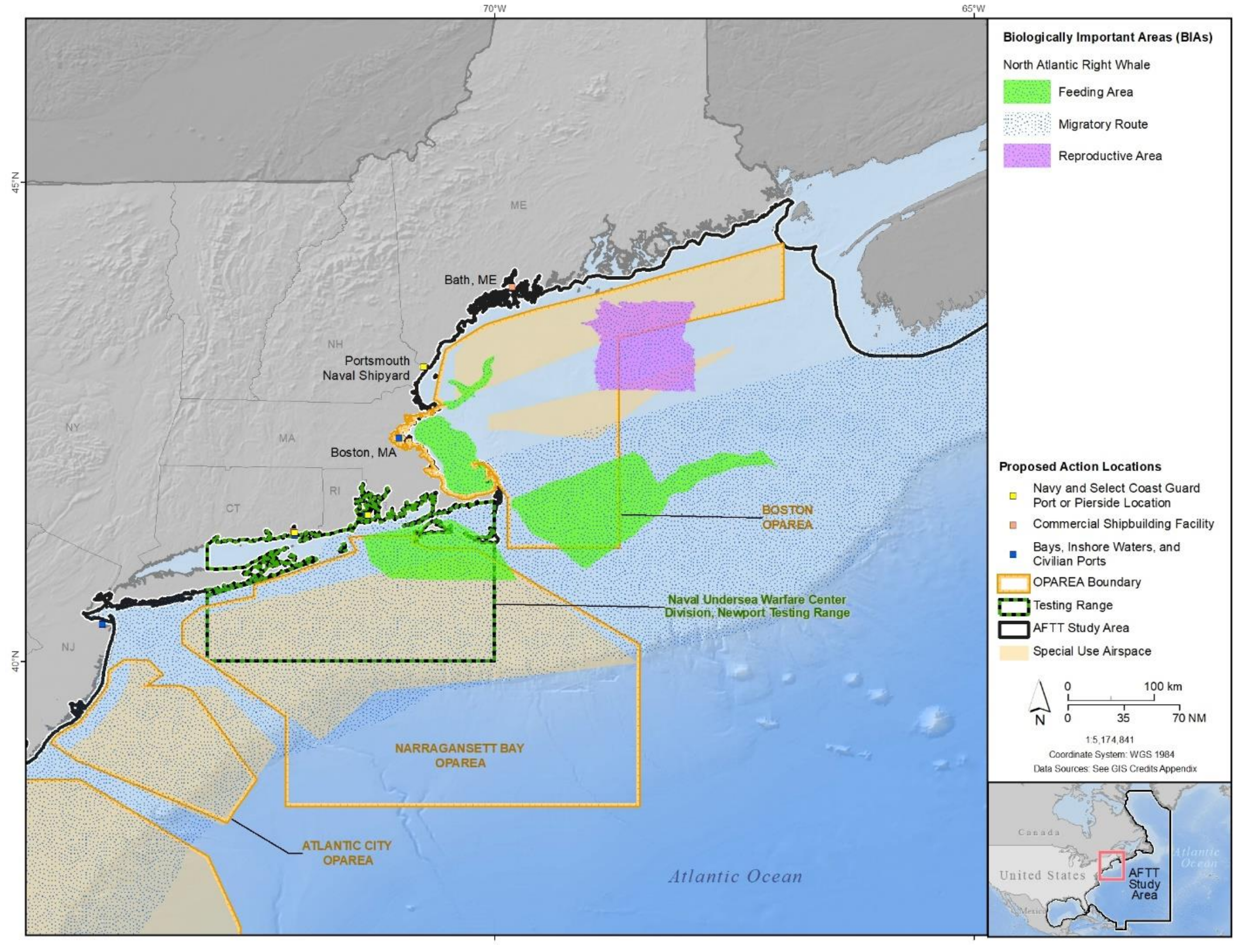


Figure F.5-1: Biologically Important Areas for North Atlantic Right Whales in the Study Area – Northeast

An important shift in habitat use patterns in 2010 was highlighted in an analysis of right whale acoustic presence along the U.S. Eastern seaboard from 2004 to 2014 (Davis et al., 2017). This shift was also reflected in visual survey data in the greater Gulf of Maine region. Between 2012 and 2016, visual surveys detected fewer individuals in the Great South Channel and the Bay of Fundy (Davies et al., 2019), while the number of individuals using Cape Cod Bay in spring increased (Mayo et al., 2018). In addition, right whales abandoned the Jordan Basin in the central Gulf of Maine in winter (Cole et al., 2013) but have since been seen in large numbers in a region south of Martha's Vineyard and Nantucket Islands (Leiter et al., 2017), an area outside of the 2016 Northeastern U.S. Foraging Area Critical Habitat. Since 2013, increased detections and survey effort in the Gulf of St. Lawrence indicate right whale presence in late spring through early fall (Cole et al., 2016; Khan et al., 2016; Khan et al., 2018; Simard et al., 2019). Aerial surveys of the Gulf of St. Lawrence between 2015 and 2019 showed that 40 percent of the population now utilizes this habitat area with potential residencies up to 5 months (Crowe et al., 2021). Updated density information quantifies and supports the distribution shifts in this species between 2003-2009 and 2010-2020, as well as correlates positively with acoustic detections for this species along the U.S. Eastern Seaboard (Roberts et al., 2024; Salisbury et al., 2015).

The winter range for North Atlantic right whales includes the Southeast U.S. Continental Shelf Large Marine Ecosystem. (LaBrecque et al., 2015a) used habitat analyses of sea surface temperature, water depth, and aerial sighting data to delineate a calving area in the southeast Atlantic, extending from Cape Lookout, North Carolina, to Cape Canaveral, Florida, that overlaps with the Study Area (Figure F.5-2). This area, identified as biologically important, encompasses waters from the shoreline to the 25-m isobath from mid-November through late April. Passive acoustic monitoring conducted offshore of Cape Hatteras and in Onslow Bay, North Carolina, in 2011 and 2007, respectively, confirmed winter occurrence of North Atlantic right whales in these areas (McLellan et al., 2014).

Four right whale sightings were documented during monthly aerial surveys approximately 50 mi. (80 km) offshore of Jacksonville, Florida, from 2009 to May 2016, including a female that was observed giving birth in 2010 (Foley et al., 2011). These sightings occurred well outside existing ESA-designated critical habitat (Foley et al., 2011; U.S. Department of the Navy, 2011). While sightings have generally occurred within nearshore continental shelf waters off northeastern Florida and southeastern Georgia, detections of North Atlantic right whales were recorded in deeper waters during these monitoring efforts (Davis et al., 2017; Kumar et al., 2013; Norris et al., 2012), suggesting that distribution of this species extends further offshore than sighting data previously indicated (Oswald et al., 2016). A noteworthy number of right whales (36 unique individuals, or approximately 10 percent of the estimated population at the time) were seen mid-shelf and offshore of Virginia during the 2022-2023 winter months and were sighted in the same areas on multiple days of survey effort (Aschettino et al., 2024), indicating that the whales are not simply migrating through the area. Recent studies have indicated that migration in North Atlantic right whales is “condition-dependent partial migration,” where full migration to the breeding grounds can be skipped if tradeoffs such as reproductive costs or foraging opportunities are present for an individual whale (Gowan et al., 2019). Habitat shifts such as ocean warming are influencing right whale movement overall (Meyer-Gutbrod et al., 2021).

Right whales have occasionally been recorded in the Gulf of Mexico Large Marine Ecosystem (Moore & Clark, 1963; Ward-Geiger et al., 2011), but their occurrence there is likely extralimital. The few published records from the Gulf of Mexico represent either distributional anomalies, normal wanderings of occasional animals, or a more extensive historical range beyond the currently known calving and wintering ground in the waters of the southeastern United States (Moore & Clark, 1963; Ward-Geiger et al., 2011).

Two critical habitats (Figure 3.7-1, Designated Critical Habitat for North Atlantic Right Whales in the Study Area) are designated by NMFS for North Atlantic right whales to encompass physical oceanographic and biological features essential to conservation of the species (81 *Federal Register* 4838). The northern foraging unit includes the Gulf of Maine and Georges Bank region where oceanographic and bathymetric conditions favor the distribution and aggregation of *Calanus finmarchicus*, a fundamental prey source. The southern calving unit includes the coast of North Carolina, South Carolina, Georgia, and Florida. The essential features for this unit include calm sea surface conditions, a sea surface temperature range between 7 to 17˚C, and depths of 6 to 28 m (81 *Federal Register* 4838). These two ESA-designated critical habitats were designated in January 2016 to replace three smaller previously designated critical habitats (Cape Cod Bay/Massachusetts Bay/Stellwagen Bank, Great South Channel, and the coastal waters of Georgia and Florida in the southeastern United States) that had been designated by NMFS in 1994 (81 *Federal Register* 4838). Two additional critical habitat areas in Canadian waters, Grand Manan Basin and Roseway Basin, were identified in Canada’s final recovery strategy for the North Atlantic right whale (Brown et al., 2009).

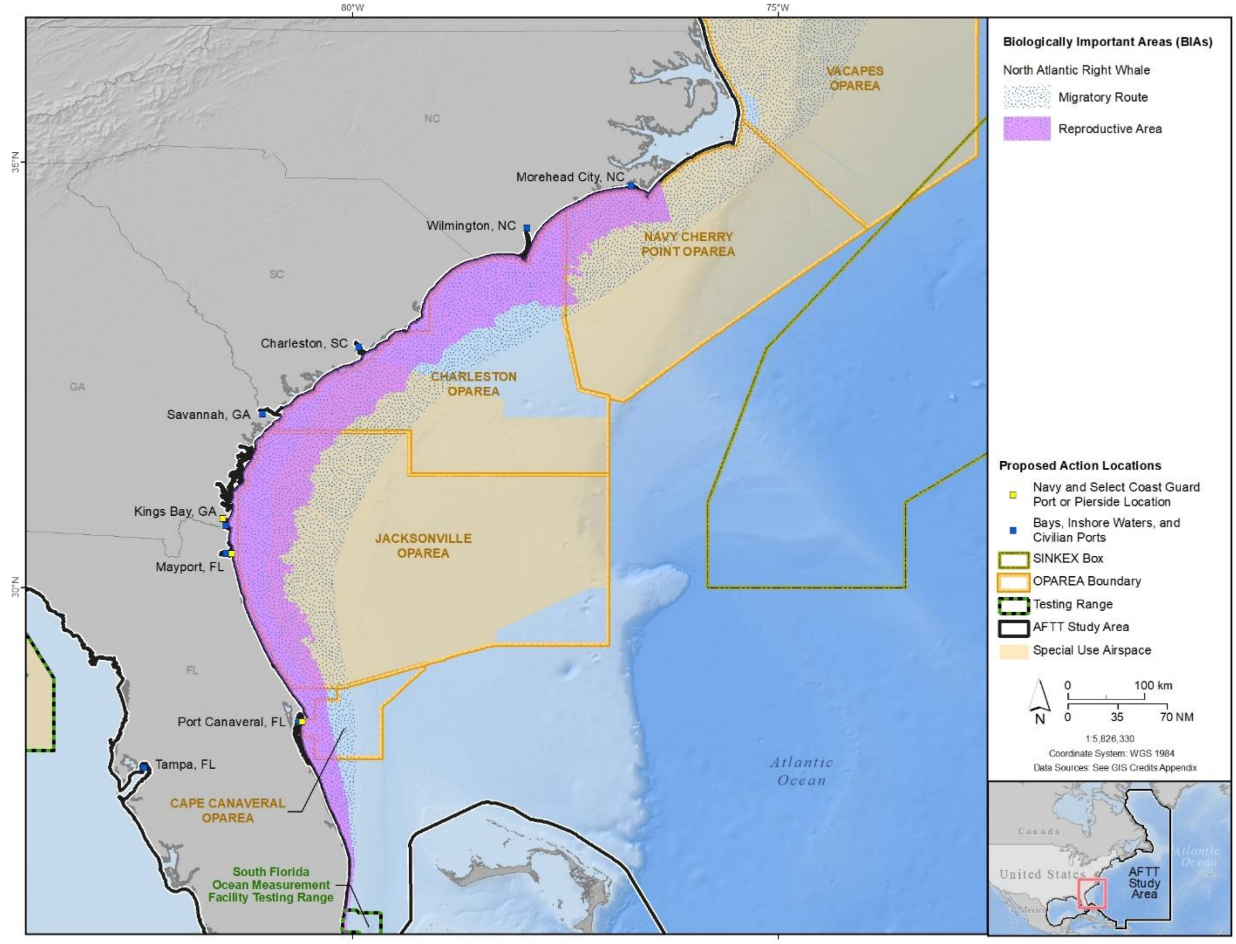


Figure F.5-2: Biologically Important Areas for North Atlantic Right Whales in the Study Area – Southeast

F.5.2.1.3 Population Trends

Examination of the abundance estimates for the years 1990–2011 (see Hayes et al., 2022, Figures 2a, 2b) suggests that abundance increased at about 2.8 percent per annum from posterior median point estimates of 270 individuals in 1990 to 481 in 2011, but that there was a 100 percent chance that abundance declined from 2011 to 2019 when the final estimate was 368 individuals (Hayes et al., 2022). The overall abundance decline between 2011 and 2019 was 23.5 percent (confidence interval = 21.4 percent to 26.0 percent) (Hayes et al., 2022). There has been a considerable change in right whale habitat use patterns in areas where most of the population had been observed in previous years (e.g., Davies et al., 2019), exposing the population to new anthropogenic threats (Hayes et al., 2018). Pace III et al. (2021) found a significant decrease in mean survival rates since 2010, correlating with the observed change in area-use patterns. There were 17 right whale mortalities reported in 2017 (Daoust et al., 2017). This number exceeds the largest estimated mortality rate during the past 25 years. Further, despite high survey effort, only five and zero calves were detected in 2017 and 2018, respectively. In 2019, seven calves were identified (Pettis et al., 2022). Calf numbers slightly improved in 2020, 2021, and 2022 with 10, 20, and 15 calves, respectively (National Marine Fisheries Service, 2024f).

F.5.2.1.4 Predator and Prey Interactions

Killer whales and large sharks are known predators of the North Atlantic Right Whale. Calves and juveniles are known to be the primary target of killer whales, and analysis of scars on some individuals suggests that they are also attacked by false killer whales (Jefferson et al., 2015; Kenney, 2009).

The North Atlantic right whale feeds primarily on the copepod *Calanus finmarchicus* (a type of zooplankton) and on other copepods and small invertebrates, such as krill and larval barnacles (Jefferson et al., 2015). Right whales are skim feeders and are known to feed below or at the surface (Kenney et al., 2001) or within a few meters of the seafloor on near-bottom aggregations of copepods (Baumgartner, 2009; Baumgartner et al., 2009; Warren, 2009). The copepod *C. finmarchicus* is one of the most common species of prey found throughout the North Atlantic right whale's range (Baumgartner & Mate, 2003; Jefferson et al., 2015).

F.5.2.1.5 Species-Specific Threats

Primary sources of human-caused serious injury and mortality include entanglement in fishing gear and ship strikes. The estimated annual rate of total mortality is 18.6 animals for the period 2013 through 2017 (Pace, 2021). This estimate accounts for detected mortality and serious injury, as well as undetected (cryptic) mortality within the population. The detection rate of mortality and serious injury for the 5-year period 2013 through 2017 was 51 percent of the model's annual mortality estimates (Pace, 2021).

Entanglement records from 1990 through 2007 maintained by NMFS Northeast Regional Office included 46 confirmed right whale entanglements, including right whales in weirs (stationary nets fixed in place), gillnets, and trailing line and buoys. From 2017 to 2023, 9 mortalities, 22 serious injuries, and 31 sub-lethal injuries have been attributed to entanglement (National Marine Fisheries Service, 2024d).

Because whales often free themselves of gear following an entanglement event, scarring may be a better indicator of fisheries interaction than entanglement records. A review of scars detected on identified individual right whales over a period of 30 years (1980 to 2009) documented 1,032 definite, unique entanglement events on the 626 individual whales identified (Knowlton et al., 2012). Most individual whales (83 percent) were entangled at least once, and almost half of them (306 of 626) were definitely entangled more than once. About a quarter of the individuals identified in each year (26 percent) were entangled during that year. Juveniles and calves were entangled at higher rates than were adults. Scarring rates suggest that entanglements are occurring at about an order of magnitude greater than that detected from observations of whales with gear on them.

Ship strikes pose a particularly serious threat to the North Atlantic right whale. Vessel speed as well as angle of approach can influence the severity of ship strikes (Silber et al., 2010). Research shows that the probability of right whales dying after being struck by a ship is more than 80 percent when a vessel is traveling at 15 knots or more; when speeds are reduced to 10 knots or less, the chance of mortality drops to just above 20 percent. To reduce the number of ship strikes, NMFS has established regulations (73 *Federal Register* 60173) imposing speed restrictions in seasonal management areas for commercial ships 65 ft. or longer. For additional detail on ship strikes and right whales, refer to Section 3.7.3.4 (Physical Disturbance and Strike Stressors).

F.5.2.2 Blue Whale (Balaenoptera musculus)

F.5.2.2.1 Status and Management

Western North Atlantic blue whales are listed as endangered under the ESA and designated as a depleted and strategic stock under the MMPA. A final recovery plan was published for the blue whale in U.S. waters in November 2020 (National Marine Fisheries Service, 2020b). Blue whales in the western North Atlantic are classified as a single stock (Hayes et al., 2021).

Widespread whaling over the last century is believed to have decreased the worldwide population to approximately 1 percent of its pre-whaling population size, although some authors have concluded that their pre-whaling population size was about 200,000 animals (Branch, 2007; Širović et al., 2004). There was a documented increase in the blue whale population size in some areas between 1979 and 1994, but there is no evidence to suggest an increase in the population since then (Barlow, 1994; Barlow & Taylor, 2001; Carretta et al., 2010).

F.5.2.2.2 Habitat and Geographic Range

The distribution of the blue whale in the western North Atlantic generally extends from the Arctic to at least mid-latitude waters. Blue whales are most frequently sighted in the waters off eastern Canada, with the majority of recent records from the Gulf of St. Lawrence. Members of the North Atlantic population spend much of their time in continental shelf waters from eastern Canada (near the Quebec north shore) to the St. Lawrence Estuary and Strait of Belle Isle. Sightings were reported along the southern coast of Newfoundland during late winter and early spring (Reeves et al., 2004). Blue whales may be found in Labrador Current, North Atlantic Gyre, and Gulf Stream open-ocean areas. Migratory movements in the western North Atlantic Ocean are largely unknown. Acoustic data indicate that blue whales winter as far north as Newfoundland and as far south as Bermuda and Florida, and they have been sighted along the mid-Atlantic ridge (Ryan et al., 2013).

The blue whale is best considered as an occasional visitor in U.S. Atlantic Exclusive Economic Zone waters, which may represent the current southern limit of its feeding range (Cetacean and Turtle Assessment Program, 1982). Historical blue whale observations collected by Reeves et al. (2004) show a broad longitudinal distribution in tropical and warm temperate latitudes during the winter months, with a narrower, more northerly distribution in summer. Blue whales tagged in the Gulf of St. Lawrence in late fall left the St. Lawrence Estuary and used habitat more than 1,000 km offshore, as well as shelf and coastal waters of the eastern United States and Canada (Lesage et al., 2016).

Although the exact extent of their southern boundary and wintering grounds are not well understood, blue whales are occasionally found in waters off the U.S. Atlantic coast (Waring et al., 2013). Yochem and Leatherwood (1985) summarized records that suggested an occurrence of this species south to Florida and the Gulf of Mexico. Blue whale strandings have been recorded as far south as the Caribbean and the Gulf of Mexico (Waring et al., 2010). Monthly aerial surveys were conducted offshore of Cape Hatteras (2011 to 2017) and Onslow Bay (2007 to 2011), North Carolina, with no documented visual sightings of blue whales (McLellan, 2017). Engelhaupt et al. (2020) reported two sightings of blue whales off the coast of Virginia in April 2018 and February 2019, between 100 and 135 km offshore; the whale sighted in February was seen feeding with a congregation of fin whales. Acoustic monitoring has also been conducted in the same region since 2011 and resulted in the detections of blue whales on bottom-mounted high-frequency acoustic recording packages (McLellan et al., 2014; Read et al., 2014). Davis et al. (2020) documented acoustic detections of blue whales from North Carolina north to the Davis Strait region year-round between 2004 and 2014, with a shift northward in years after 2010.

Critical habitat has not been designated for this species at this time.

F.5.2.2.3 Population Trends

There are insufficient data to determine population trends for this species (Hayes et al., 2020).

F.5.2.2.4 Predator and Prey Interactions

This species preys almost exclusively on various types of zooplankton, especially krill. They lunge feed and consume approximately 6 tons (5,500 kg) of krill per day (Jefferson et al., 2015; Pitman et al., 2007). They often feed at depths greater than 100 m, where their prey maintains dense groupings (Acevedo-Gutiérrez et al., 2002; Calambokidis et al., 2009; Croll et al., 2001). Blue whales are documented as being preyed upon by killer whales (Jefferson et al., 2015; Pitman et al., 2007). There is little evidence that killer whales attack this species in the North Atlantic or southern hemisphere, but 25 percent of photo-identified whales in the Gulf of California carry rake scars from killer whale attacks (Sears & Perrin, 2009).

F.5.2.2.5 Species-Specific Threats

Threats to the North Atlantic blue whale are poorly known but may include ship strikes, pollution, entanglement in fishing gear, and long-term changes in climate that may affect their prey distribution.

F.5.2.3 Fin Whale (*Balaenoptera physalus*)

F.5.2.3.1 Status and Management

The fin whale is found in all of the world's oceans and is the second-largest species of whale (Jefferson et al., 2015). Fin whales have three recognized subspecies: the North Atlantic fin whale (*Balaenoptera physalus physalus*), the North Pacific fin whale (*B. p. velifera*), and the southern fin whale (*B. p. quoyi*), which occurs only in the southern hemisphere (Committee on Taxonomy, 2023). Only the North Atlantic subspecies is expected to occur within the Study Area.

Fin whales in the Northwest Atlantic are listed as endangered under the ESA and the species is considered a depleted and strategic stock under the MMPA. A final recovery plan was published in August 2010 for fin whales in U.S. waters (National Marine Fisheries Service, 2010b). The International Whaling Commission recognizes seven management stocks of fin whales in the North Atlantic Ocean: (1) Nova Scotia, (2) Newfoundland-Labrador, (3) West Greenland, (4) East Greenland-Iceland, (5) North Norway, (6) West Norway-Faroe Islands, and (7) British Isles-Spain-Portugal. NMFS assumes management of the western North Atlantic stock, which is likely equivalent to the Nova Scotia management stock. The stock identity of North Atlantic fin whales has received relatively little attention, and whether the current stock boundaries define biologically isolated units has long been uncertain (Waring et al., 2016). Fin whales in the Gulf of St. Lawrence may be a separate stock (Ramp et al., 2014).

F.5.2.3.2 Habitat and Geographic Range

Fin whales prefer temperate and polar waters and are rarely seen in warm tropical waters (Reeves et al., 2002a). They typically congregate in areas of high productivity and spend most of their time in coastal and shelf waters but can also be found in waters to approximately 2,000 m deep (Aissi et al., 2008; Reeves et al., 2002a). Fin whales are often seen closer to shore after periodic patterns of upwelling (underwater motion) and the resultant increased krill density (Azzellino et al., 2008). This species is highly adaptable, following prey, typically off the continental shelf (Azzellino et al., 2008; Panigada et al., 2008). Fin whales are likely common in Labrador Current, North Atlantic Gyre, and Gulf Stream open-ocean areas while undergoing seasonal migrations. However, some fin whales remain in higher latitudes during colder months and in lower latitudes during warmer months, indicating that seasonal fin whale movements differ from the seasonal migrations of other mysticetes, such as blue whales and humpback whales (Edwards et al., 2015). Fin whales are also common off the Atlantic coast of the United States seaward to the continental shelf edge (at about the 1,000-fathom contour). In the mid-Atlantic region, they tend to occur north of Cape Hatteras where they accounted for about 46 percent of the large whales observed in surveys conducted between 1978 and 1982 (National Marine Fisheries Service, 2010b). Recent surveys under the Atlantic Marine Assessment Program for Protected Species have shown that fin whale density increases along the shelf break from Cape Hatteras northward during summer months and decreases during winter and spring (Bureau of Ocean Energy Management, 2021). During the summer, fin whales in this region tend to congregate in feeding areas between 41°20’ north and 51°00’ north, from the shore seaward to the 1,000-fathom contour. In the western Atlantic, they winter from the edge of sea ice (near the Gulf of St. Lawrence) south to the Gulf of Mexico and the West Indies (Ramp et al., 2024).

Fin whale sightings and acoustic detections are greatest in New England waters during spring and summer, with scattered sightings over the northeast shelf in winter, indicating that some fin whales are present during the non-feeding season (Hain et al., 1992; Morano et al., 2012; Waring et al., 2014). Fin whales are also observed in the Gulf of Maine, the Bay of Fundy, the Gulf of St. Lawrence, and in offshore areas of Nova Scotia (Coakes et al., 2005; Johnston et al., 2005). Near the Bay of Fundy, fin whales are known to congregate close to the tip of Campobello Island, where they feed within localized upwelling areas and fronts in the Northeast United States Continental Shelf Large Marine Ecosystem (Johnston et al., 2005).

New England waters are considered a major feeding ground for fin whales, and there is evidence that females continually return to this area (Waring et al., 2010). Forty-nine percent of fin whales sighted on the feeding grounds of Massachusetts Bay were sighted again within the same year, and 45 percent were sighted again in multiple years (Waring et al., 2010). LaBrecque et al. (2015a) identified three feeding areas for fin whales in the North Atlantic within the Study Area: (1) June to October in the northern Gulf of Maine, (2) year-round in the southern Gulf of Maine, and (3) March to October east of Montauk Point, as substantiated through vessel-based survey data, photo-identification data, and expert judgment (Figure F.5-3).

Visual and acoustic surveys between 2014 and 2023 have documented fin whale presence in the mid-Atlantic region. Biopsy samples and satellite tagging data have also been collected, including re-sights of several individuals over the continental shelf. Vessel based surveys and satellite tagging efforts in recent years have also shown fin whales frequently occur off the coast of Virginia during winter months; observations included foraging behavior as well as adult and juvenile pairs (Aschettino et al., 2024). Fin whales have been detected frequently throughout the winter months during passive acoustic monitoring efforts conducted from 2007 through 2015 within the continental shelf break and slope waters off Onslow Bay, North Carolina (Hodge et al., 2014, 2015, 2016; U.S. Department of the Navy, 2013b). Visual surveys and passive acoustic monitoring conducted from 2007 to 2011 in Onslow Bay, North Carolina, indicate fin whale occurrence in this area between late fall and early spring (Hodge, 2011). High-frequency recording packages deployed between November 2007 and April 2010 in Onslow Bay detected 20-Hz pulses from fin whales primarily in the winter months, starting in November and continuing through mid-April, suggesting that fin whales are migrating past Onslow Bay during this time (Hodge, 2011). In the western Atlantic, limited data indicate that some fin whales winter from the edge of sea ice (near the Gulf of St. Lawrence) south to the Gulf of Mexico and the West Indies (Clark, 1995). Critical habitat has not been designated for this species at this time.

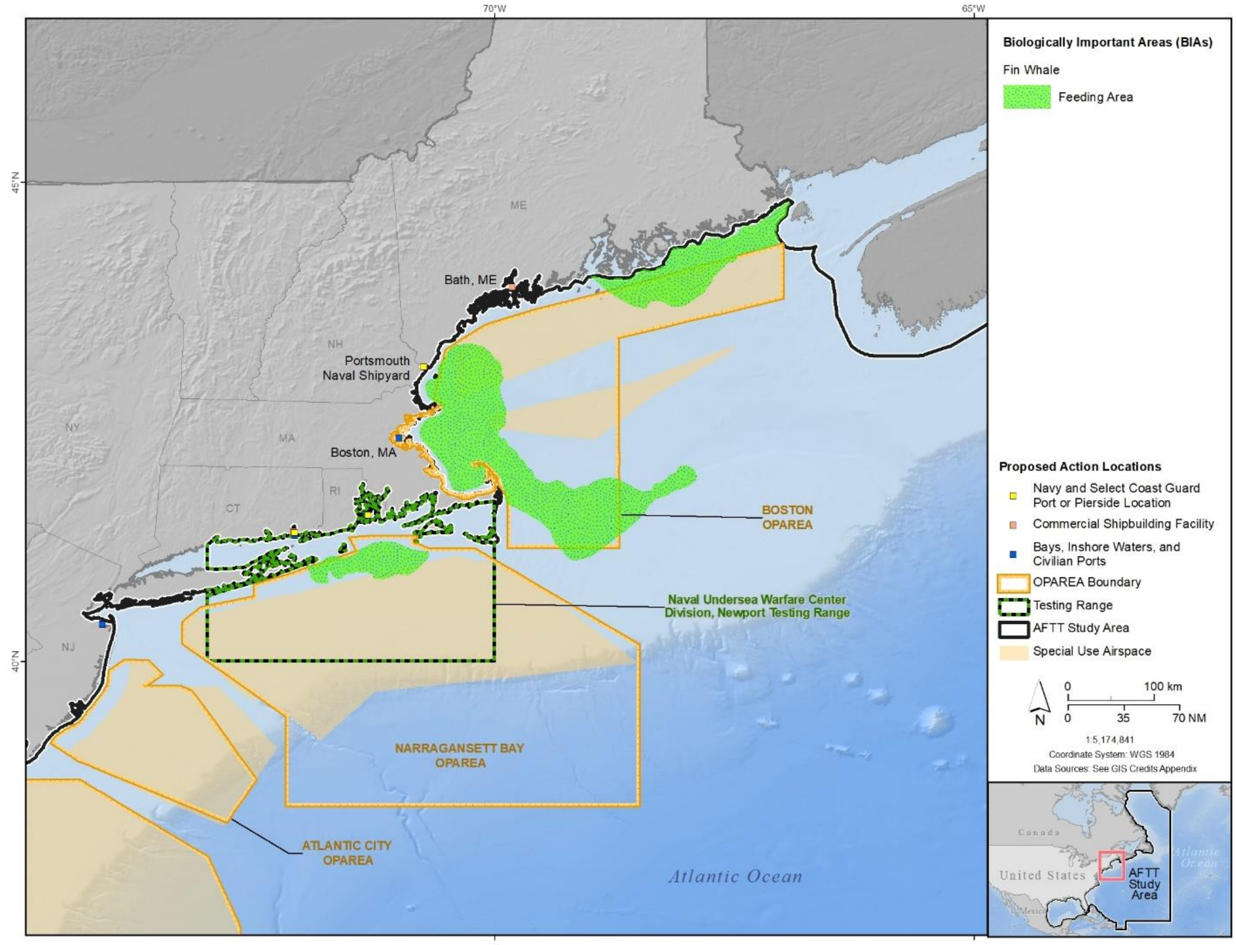


Figure F.5-3: Biologically Important Areas for Fin Whales in the Study Area

F.5.2.3.3 Population Trends

A population trend analysis has not been conducted for this stock (Hayes et al., 2021).

F.5.2.3.4 Predator and Prey Interactions

This species preys on small invertebrates such as copepods, as well as squid and schooling fishes such as capelin, herring, and mackerel (Goldbogen et al., 2006; Jefferson et al., 2015). The fin whale is not known to have a significant number of predators. However, in regions where killer whales are abundant, some fin whales exhibit attack scars on their flippers, flukes, and flanks, suggesting possible predation by killer whales (Aguilar, 2009).

F.5.2.3.5 Species-Specific Threats

Fin whales are susceptible to both ship strikes and entanglement in fishing gear.

F.5.2.4 Rice's Whale (*Balaenoptera ricei*)

F.5.2.4.1 Status and Management

Rice's whale was formerly known as the Northern Gulf of Mexico stock of Bryde's whale. It was designated a separate species in 2021 based on genetic and morphometric data distinguishing it from other subspecies of Bryde's whale (Rosel et al., 2021).

Rice's whale is listed as endangered under the ESA and considered depleted under the MMPA. The population is very small (fewer than 100 animals), exhibits very low genetic diversity, and has a restricted range, which places the stock at great risk of demographic and environmental stochasticity. There was no statistically significant trend in population size for this species.

F.5.2.4.2 Habitat and Geographic Range

Referred to as the core distribution area by NMFS, Rice's whales occur almost exclusively in the northeastern Gulf of Mexico in the De Soto Canyon area, along the continental shelf break between 100 m and 400 m depth, with a single sighting at 408 m (Hansen et al., 1996; Maze-Foley & Mullin, 2006; Mullin & Fulling, 2004; Mullin & Hoggard, 2000; Rice et al., 2014; Rosel et al., 2016; Rosel et al., 2021; Širović et al., 2014; Soldevilla et al., 2017). Rice's whales have been sighted in all seasons within the De Soto Canyon area (Deepwater Horizon Marine Mammal Injury Quantification Team, 2015; Maze-Foley & Mullin, 2006; Mullin, 2007; Mullin & Hoggard, 2000). Two strandings from the southeastern U.S. Atlantic coast share the same genetic characteristics with those from the northern Gulf of Mexico (Rosel & Wilcox, 2014; Rosel et al., 2021), but it is unclear whether these are extralimital strays (Mead, 1977) or whether they indicate the population extends from the northeastern Gulf of Mexico to the Atlantic coast of the southern United States (Rosel & Wilcox, 2014; Rosel et al., 2021). There have been no confirmed sightings of Rice's whales along the U.S. East Coast during NMFS cetacean surveys (Rosel et al., 2016). Between 2000 and 2021, data in the OBIS-SEAMAP indicated there were eight sightings of Rice's whales in the Gulf of Mexico portion of the Study Area, totaling 21 individuals (Halpin et al., 2009).

While their core distribution primarily lies within continental U.S. waters, research by Soldevilla et al. (2024) provides the first evidence of Rice's whale presence in Mexican waters using autonomous passive acoustic recording devices in the Mexican continental slope from 2020 to 2022. Rice's whales were detected 14.9 percent of days across a period of 680 days throughout the year, with a total of 579 western long-moan calls detected. These new findings suggest Rice's whales have a broader distribution than previously understood, and have a transboundary range throughout the Gulf of Mexico beyond U.S. waters (Soldevilla et al., 2024).

On July 24, 2023, NMFS released the Proposed Rule for the designation of critical habitat for the Rice's whale in the Gulf of Mexico (Figure 3.7-3, Proposed Critical Habitat for the Rice's Whales in the Study Area) in accordance with section 4(b)(2) of the ESA (88 *Federal Register* 47453). The proposed area covers 28,270.65 square miles along continental shelf and slope waters between 100 m and 400 m isobaths; spanning from the U.S. Exclusive Economic Zone boundary off the southwestern coast of Texas, to the boundary between the South Atlantic Fishery Management Council and the Gulf of Mexico Fishery Management Council off the southeastern coast of Florida (88 *Federal Register* 47453). This continental shelf and slope region is the critical habitat feature deemed biologically important (Figure F.5-4 and Figure F.5-5) and essential for Rice's whale conservation due to prey density, favorable oceanographic conditions, and productivity, as well as noise conditions sufficient for communication, navigation, foraging, and threat detection (88 *Federal Register* 47453). The area proposed for Rice's whale critical habitat overlaps with the Study Area in the Gulf of Mexico. During the evaluation process, interference with mission-essential Department of Defense operations for training and testing was one of the many factors included when determining the critical habitat area. As such, NMFS anticipates a reinitiation of existing ESA consultations within the next decade to address effects to Rice's whale critical habitat, including a programmatic consultation with U.S. Navy AFTT operations (88 *Federal Register* 47453). A final critical habitat designation has not been assigned for this species at this time.

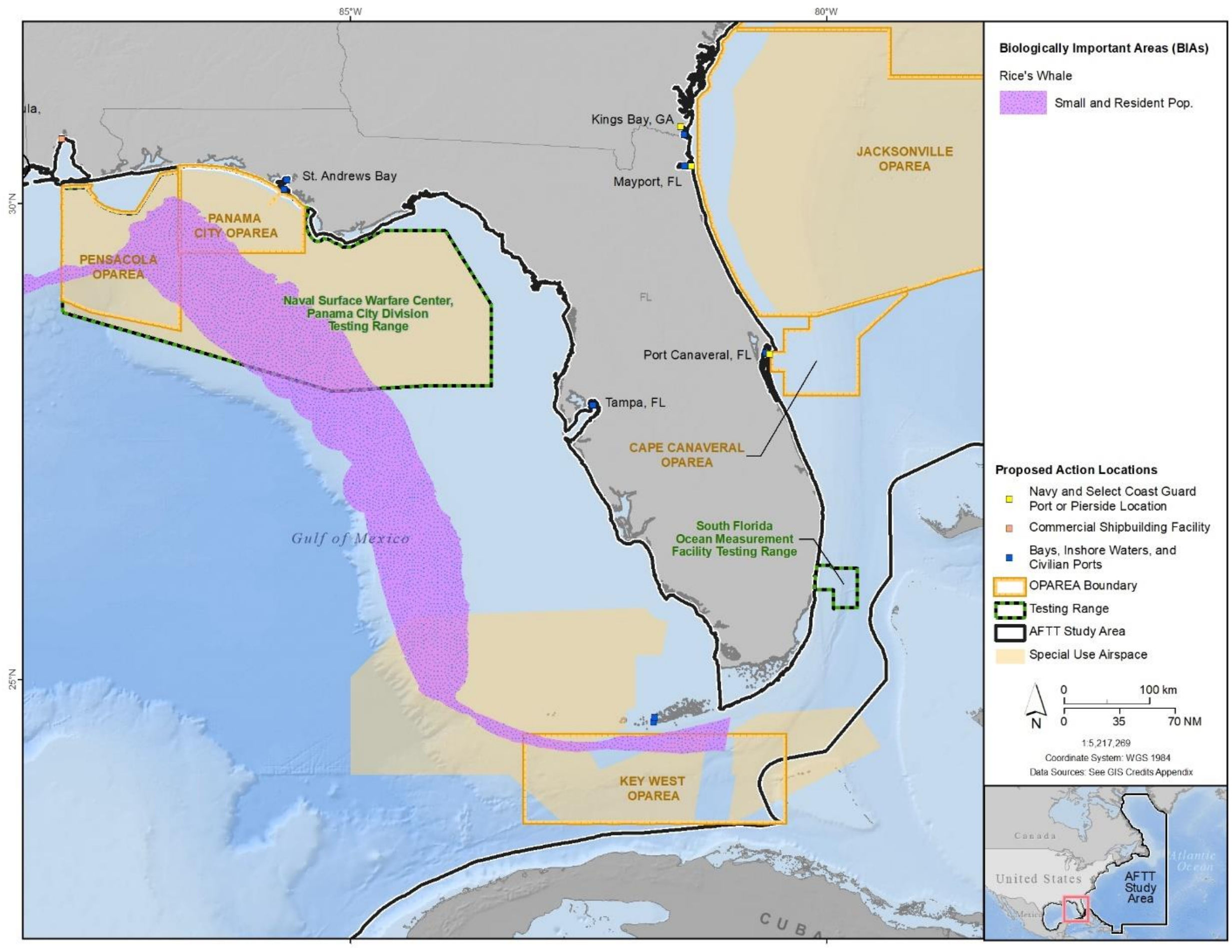


Figure F.5-4: Biologically Important Areas for Rice's Whales in the Study Area – Eastern Gulf of Mexico

F.5.2.4.3 Population Trends

The best abundance estimate available for Rice's whale is 51 (coefficient of variation = 0.50). This estimate is from summer 2017 and summer/fall 2018 oceanic surveys covering waters from the 200-m isobath to the seaward extent of the U.S. Exclusive Economic Zone (Garrison et al., 2020). The statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys. In addition, because these surveys are restricted to U.S. waters, it is not possible to distinguish between changes in population size and Gulf-wide shifts in spatial distribution. However, the potential for biological removal for the Rice's whale is 0.1. The mean modeled annual human-caused mortality and serious injury due to the *Deepwater Horizon* oil spill exceeds potential biological removal for this species (Hayes et al., 2023).

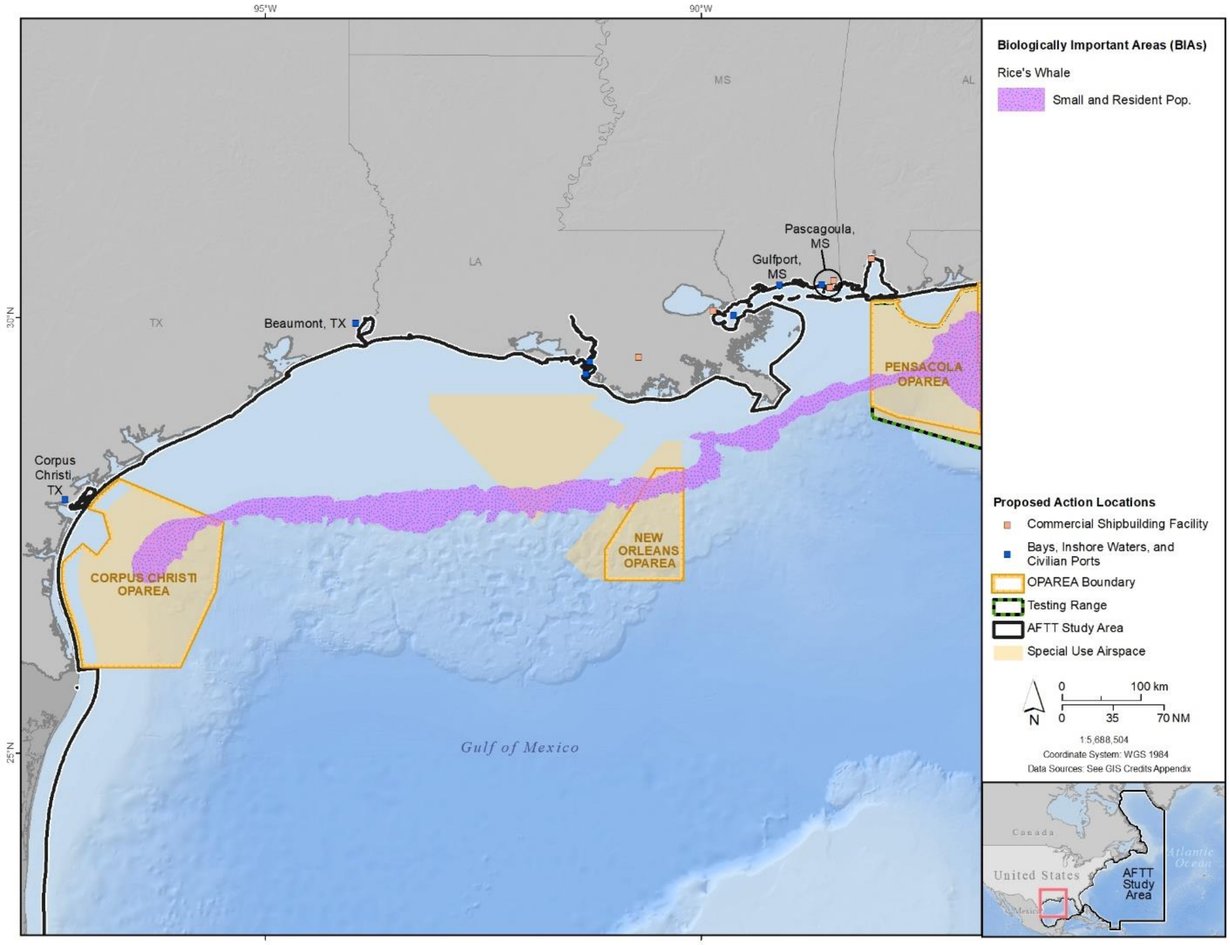


Figure F.5-5: Biologically Important Areas for Rice's Whales in the Study Area – Western Gulf of Mexico

F.5.2.4.4 Predator and Prey Interactions

Similar species (Bryde's whale) worldwide prey on pelagic schooling fishes such as order Clupeiformes (sardines, herring, menhaden, anchovies) or family Carangidae (Rosel et al., 2021) However, diet is poorly characterized for Rice's whale because surface feeding has never been observed (Rosel et al., 2021). Acousonde tag data from a Rice's whale in the northeastern Gulf of Mexico revealed a diel diving pattern with the whale staying near the surface at night and making repeated deep dives to depths greater than 200 m during the day, likely at or near the seafloor (Soldevilla et al., 2017). Some of these deep dives included lunges near the seafloor associated with foraging, which is unusual for members of the species complex. It is not known what they may have been feeding on at those depths, however it may have been lanternfish (Myctophidae) and hatchetfish (Sternoptychidae) (Rosel et al., 2021). In an attempt to understand the predator-prey dynamics of Rice's whales, Kiszka et al. (2023) conducted the first study to examine the feeding ecology and drivers of prey selection in their core habitat of the northeastern Gulf of Mexico. Trawl surveys conducted to establish nektonic community composition, abundance, and availability, paired with analyses of biopsied skin samples using Bayesian stable isotope mixing models, Kiszka et al. (2023) identified and ranked four potential prey preferences: silver-rag driftfish (*Ariomma bondi*), Dumeril's lanternfish (*Diaphus dumerilii*), longfin inshore squid (*Doryteuthis pealeii*), and Atlantic pearlside (*Maurolicus weitzmani*). Further analysis into the dietary composition of potential prey indicated that silver-rag driftfish was significantly ($p<0.05$) more nutrient rich in terms of wet and dry energy, lipid, and protein content compared to the other three species. When evaluating potential prey quality versus quantity, the results of this study suggest Rice's whales selectively forage on prey with the highest energy content rather than species with the highest abundance (Kiszka et al., 2023). Similar to other baleen whales, Rice's whales are likely subject to occasional attacks by killer whales.

F.5.2.4.5 Species-Specific Threats

Rice's whales are under particular threat from habitat destruction, modification or curtailment of habitat range during energy exploration and development, oil spills and oil spill response, as well as marine debris (Rosel et al., 2016; Rosel et al., 2021). It is hypothesized their population was reduced by 22 percent as a result of the *Deepwater Horizon* oil spill (Hayes et al., 2023).

F.5.2.5 Sei Whale (*Balaenoptera borealis*)

F.5.2.5.1 Status and Management

The sei whale is listed as endangered under the ESA and is considered a depleted and strategic stock under the MMPA. A recovery plan for the sei whale was finalized in 2011 (National Marine Fisheries Service, 2011b). There are two stocks of sei whale in the Northwest Atlantic: a Nova Scotia stock and a Labrador Sea stock (Waring et al., 2013; Waring et al., 2016). The Nova Scotia stock is considered in the management unit under NMFS jurisdiction; it includes the continental shelf waters of the northeastern United States and extends northeastward to south of Newfoundland. The Labrador Sea stock is outside of NMFS jurisdiction but occurs within the Study Area.

F.5.2.5.2 Habitat and Geographic Range

Sei whales have a worldwide distribution and are found primarily in cold temperate to subpolar latitudes. Sei whales are typically found in the open ocean and are rarely observed near the coast (Horwood, 2009; Jefferson et al., 2015). During the winter, sei whales are found from 20° N to 23° N and during the summer from 35° North to 50° North (Horwood, 2009; Masaki, 1976, 1977; Smultea et al., 2010). They are considered absent or at very low densities in most equatorial areas and in the Arctic Ocean. Satellite tagging data indicate sei whales feed and migrate east to west across large sections of the North Atlantic (Olsen et al., 2009); however, they are not often seen within the equatorial Atlantic. In the Study Area, the open-ocean range includes the Labrador Current, North Atlantic Gyre, and Gulf Stream open-ocean areas. Habitat suitability analyses suggest that the recent distribution patterns of sei whales in U.S. waters appear to be related to waters that are cool (less than 10° C), with high levels of chlorophyll and inorganic carbon, and where the mixed layer depth is relatively shallow (less than 50 m) (Chavez-Rosales et al., 2019; Palka et al., 2017).

Sei whales spend the summer feeding in subpolar high latitudes and return to lower latitudes to calve in winter. However, no migratory corridor for sei whales has been identified in U.S. Atlantic waters (LaBrecque et al., 2015a). There are no known sei whale mating or calving grounds in U.S. Atlantic waters (LaBrecque et al., 2015a).

(LaBrecque et al., 2015a) delineated a feeding area for sei whales in the northeast Atlantic between the 25-m contour off coastal Maine and Massachusetts to the 200-m contour in the central Gulf of Maine, including the northern shelf break area of Georges Bank (Figure F.5-6). The feeding area also includes the southern shelf break area of Georges Bank from 100 to 2,000 m and the Great South Channel. Feeding activity in the U.S. Atlantic waters is concentrated from May through November with a peak in July and August. Spring is the period of greatest abundance in Georges Bank and into the Northeast

Channel area, along the Hydrographer Canyon (Cetacean and Turtle Assessment Program, 1982; Waring et al., 2010). Although uncommon near the coastline, two strandings of sei whales have been reported on the Virginia coast in 2003 and 2011 (King, 2011; Swingle et al., 2014).

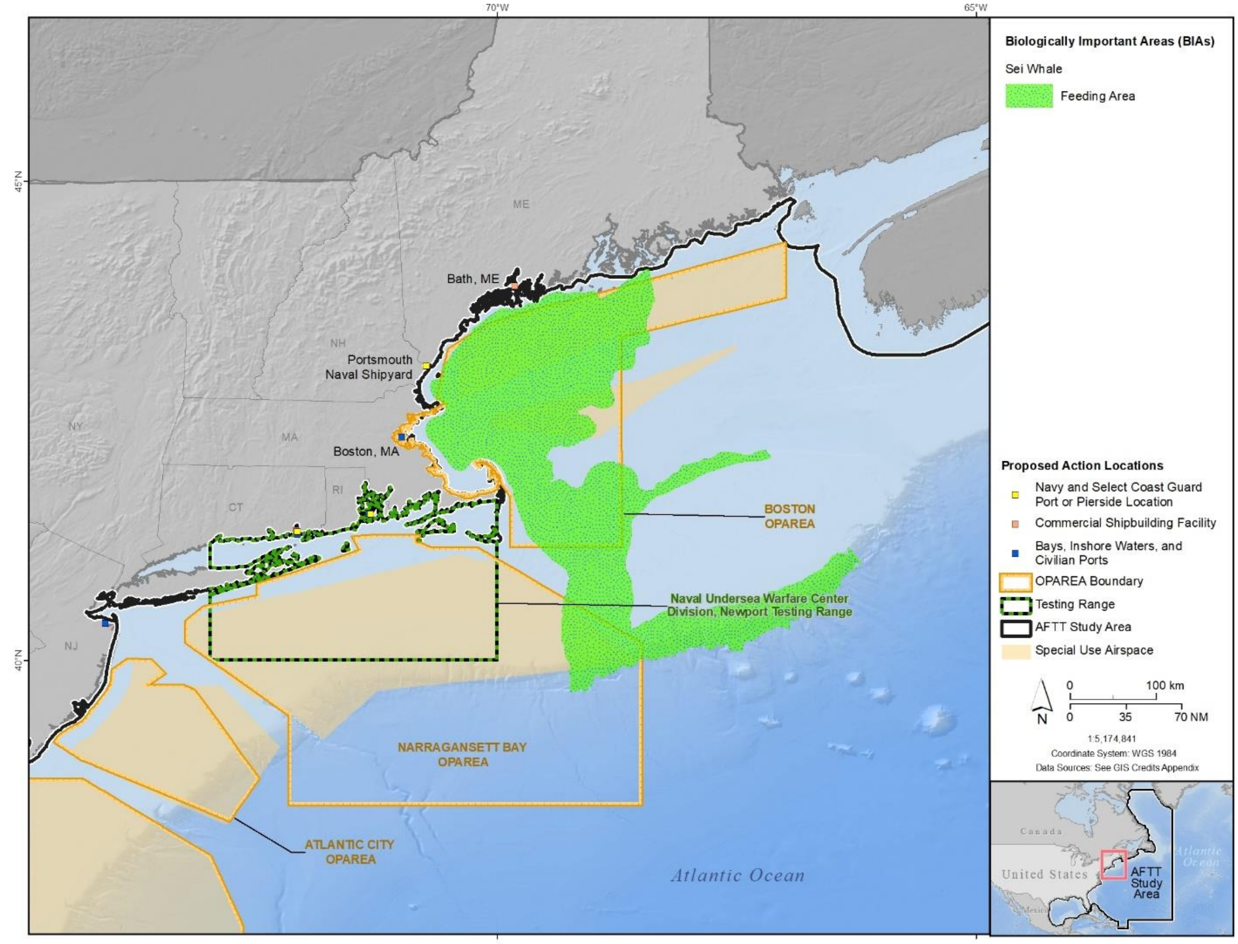


Figure F.5-6: Biologically Important Areas for Sei Whales in the Study Area

Passive acoustic monitoring conducted offshore of Cape Hatteras, North Carolina, since 2011 resulted in the detections of sei whales on bottom-mounted high-frequency acoustic recording packages that were not observed during visual surveys (McLellan et al., 2014). Passive acoustic monitoring conducted offshore of Jacksonville, Florida, from 2009 through 2020 also included detections of sei whales on marine acoustic recording units (Oswald et al., 2016) and detections on high-frequency acoustic recording packages (Hodge & Read, 2013).

Critical habitat has not been designated for this species at this time.

F.5.2.5.3 Population Trends

Commercial whaling in the 19th and 20th centuries depleted populations in all areas throughout the species' range. While they appear to be recovering in the northern hemisphere as a result of protective legislation, a trend analysis has not been conducted for this stock (Hayes et al., 2020).

F.5.2.5.4 Predator and Prey Interactions

Sei whales feed on copepods, amphipods, euphausiids, shoaling fish, and squid (Horwood, 2009); (Nemoto & Kawamura, 1977). Feeding occurs primarily around dawn, which appears to be correlated with vertical migrations of prey species (Horwood, 2009). Unlike other rorquals, the sei whale skims to obtain its food; like other rorqual species, it does exhibit other feeding behaviors such as lunging and gulping (Horwood, 2009). Similar to other baleen whales, sei whales are likely subject to occasional attacks by killer whales.

F.5.2.5.5 Species-Specific Threats

There are no significant species-specific threats to sei whales in the northwest Atlantic.

F.5.2.6 Sperm Whale (*Physeter macrocephalus*)

F.5.2.6.1 Status and Management

The sperm whale has been listed as an endangered species since 1970 under the precursor to the ESA (National Marine Fisheries Service, 2009b) and is listed as depleted and strategic under the MMPA. Whether the northwestern Atlantic population is discrete from northeastern Atlantic is currently unresolved. The International Whaling Commission recognizes one stock for the North Atlantic, based on reviews of many types of studies (e.g., tagging, genetics, catch data, mark and recapture, biochemical markers). A recovery plan is in place for the sperm whale in U.S. waters (National Marine Fisheries Service, 2010c). There are currently two stocks of sperm whales recognized within the Study Area managed under NMFS jurisdiction: the western North Atlantic and the Gulf of Mexico stocks. In 2013, NMFS determined that a petition to list the Gulf of Mexico stock as a distinct population segment was not warranted based on a review of best available information on physical, physiological, ecological, and behavioral factors (78 *Federal Register*, 68032–68037, November 13, 2013). A 5-year review for sperm whales was finalized in 2015 (National Marine Fisheries Service, 2015c).

F.5.2.6.2 Habitat and Geographic Range

Sperm whales are found throughout the world's oceans in deep waters to the edge of the ice at both poles (Leatherwood & Reeves, 1983; Rice, 1989; Whitehead, 2002). Sperm whales show a strong preference for deep waters (Rice, 1989; Whitehead, 2003). Their distribution is typically associated with waters over the continental shelf break, continental slope, and into deeper mid-ocean regions. However, in some areas, adult males are reported to consistently frequent waters with depths less than 100 m and as shallow as 40 m (Jefferson et al., 2008; Jefferson et al., 2015; Romero et al., 2001). Typically, sperm whale concentrations correlate with areas of high productivity. These areas are generally near drop-offs and areas with strong currents and steep topography (Gannier & Praca, 2007; Jefferson et al., 2015). Sperm whale migration is not well understood and is not as seasonally based as that observed in mysticete whales. Sperm whales may be found in Labrador Current, North Atlantic Gyre, and Gulf Stream open-ocean areas. Recent surveys under the Atlantic Marine Assessment Program for Protected Species between 2010 and 2020 have shown that the density of sperm whales is higher north of Cape Hatteras, with infrequent sightings south of Cape Hatteras, and had peak average abundance estimates during summer (Bureau of Ocean Energy Management, 2021).

The nature of linkages of the U.S. habitat with those to the south, north, and offshore is unknown, but sperm whales that occur in the eastern U.S. Exclusive Economic Zone in the Atlantic Ocean likely represent only a fraction of the total stock. Historical whaling records compiled by Schmidly (1981) suggested an offshore distribution off the southeast United States, over the Blake Plateau, and into deep ocean waters. Distribution along the East Coast of the United States is centered along the shelf break and over the slope. In winter, sperm whales are concentrated east and northeast of Cape

Hatteras, North Carolina. In spring, the center of distribution shifts northward to east of Delaware and Virginia and is widespread throughout the central portion of the Mid-Atlantic Bight and the southern portion of Georges Bank. In summer, the distribution is similar but now also includes the area east and north of Georges Bank and into the Northeast Channel region, as well as the continental shelf (inshore of the 100-m isobath) south of New England. In fall, sperm whale occurrence south of New England on the continental shelf is at its highest level, and there remains a continental shelf edge occurrence in the mid-Atlantic Bight. Similar inshore (less than 200 m) observations were made on the southwestern and eastern Scotian Shelf, particularly in the region of "the Gully" (Whitehead & Weilgart, 1991).

Aerial surveys conducted offshore of Cape Hatteras, North Carolina, from 2011 through 2017 have resulted in a common occurrence of sperm whales, primarily in the spring and summer months (McLellan et al., 2014).

Passive acoustic monitoring conducted in Onslow Bay, North Carolina, between 2007 and 2013 confirmed year-round occurrence of sperm whales, along with a nocturnal increase in occurrence of clicks and greater vocal activity on recorders located in deeper waters of the monitoring area (Hodge, 2011; Read et al., 2014; U.S. Department of the Navy, 2013a). Researchers confirmed occurrence of sperm whale vocalizations in Onslow Bay on a recorder deployed at water depths of 230 m and 366 m, along with regular nocturnal occurrence of sperm whale clicks near the shelf break, suggesting that foraging activities were occurring at that time (Hodge et al., 2013). This diel pattern contrasts with what was recorded offshore of Cape Hatteras (Stanistreet et al., 2013). Habitat models also support findings of sperm whale occurrence in the U.S. Economic Exclusion Zone waters offshore of Onslow Bay (Best et al., 2012). Visual surveys in Onslow Bay and analysis of remotely sensed oceanographic data were used to determine the effects of dynamic oceanography. The findings from this study indicate that the presence of Gulf Stream frontal eddies and the location of the Gulf Stream Front influenced sperm whale vocalization rates, among other species (Thorne et al., 2012).

The sperm whale is the most common large cetacean in the northern Gulf of Mexico (Palka & Johnson, 2007). The distribution of sperm whales in the Gulf of Mexico is strongly linked to surface oceanography, such as Loop Current eddies that locally increase production and availability of prey (O'Hern & Biggs, 2009). Most sperm whale groups were found within regions of enhanced sea surface chlorophyll abundance (O'Hern & Biggs, 2009). Ship-based and aerial-based surveys indicate that sperm whales are widely distributed only in waters deeper than 200 m in the northern Gulf of Mexico (Waring et al., 2014), specifically inhabiting the continental slope and oceanic waters (Fulling et al., 2003; Maze-Foley & Mullin, 2006; Mullin & Fulling, 2004; Mullin & Hoggard, 2000; Mullin et al., 2004). Seasonal aerial surveys confirm that sperm whales are present in the northern Gulf of Mexico in all seasons (Hansen et al., 1996; Mullin & Hoggard, 2000; Mullin et al., 1994b). Sperm whales aggregate at the mouth of the Mississippi River and along the continental slope in or near cyclonic cold-core eddies (counterclockwise water movements in the northern hemisphere with a cold center) or anticyclone eddies (clockwise water movements in the northern hemisphere) (Davis et al., 2007) Habitat models for sperm whale occurrence indicate a high probability of suitable habitat along the shelf break off the Mississippi delta, Desoto Canyon, and western Florida (Best et al., 2012).

NMFS winter ship surveys of waters surrounding Puerto Rico and the U.S. Virgin Islands indicate that sperm whales inhabit continental slope and oceanic waters (Roden & Mullin, 2000; Swartz & Burks, 2000; Swartz et al., 2002). Earlier sightings from the northeastern Caribbean were reported by Erdman (1970), Erdman et al. (1973), and Taruski and Winn (1976), where these and additional sightings from Puerto Rican waters are summarized by Mignucci-Giannoni (Mignucci-Giannoni, 1988). For years up to 1989, Mignucci-Giannoni found 43 records for sperm whales in waters of Puerto Rico, U.S. Virgin Islands, and British Virgin Islands and suggested these whales occur from late fall through winter and early spring but are rare from April to September. In addition, sperm whales are one of the most common species to strand in Puerto Rico and the Virgin Islands (Mignucci-Giannoni et al., 1999). In the southeast Caribbean, both large and small adults, as well as calves and juveniles of different sizes, are reported (Watkins et al., 1985).

Critical habitat has not been designated for this species at this time.

F.5.2.6.3 Population Trends

There is no reliable estimate of total sperm whale abundance, and no trend analysis has been conducted for the North Atlantic stock of sperm whales (Hayes et al., 2020).

There has been considerable variation in point estimates of northern Gulf of Mexico sperm whale abundance based on data collected from 1991 to 2018. Differences in temporal abundance will be difficult to interpret without a Gulf of Mexico-wide (including waters belonging to Mexico and Cuba) understanding of sperm whale abundance, and the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys.

F.5.2.6.4 Predator and Prey Interactions

Sperm whales socialize for predator defense as well as foraging. Sperm whales feed on squid, other cephalopods (a type of mollusk), bottom-dwelling fish, and invertebrates (Davis et al., 2007; Marcoux et al., 2007; Rice, 1989). The precise method regarding how sperm whales search for, detect, and capture their prey remains uncertain. Site-specific ecological factors, such as predation pressure and food availability, likely influence fundamental aspects of sperm whale social organization (Fais et al., 2015; Jaquet & Gendron, 2009). False killer whales, pilot whales, and killer whales have been documented harassing and, on occasion, attacking sperm whales (Baird, 2009b).

F.5.2.6.5 Species-Specific Threats

There are no significant species-specific threats to sperm whales in the northwest Atlantic or Gulf of Mexico.

F.5.2.7 West Indian Manatee (*Trichechus manatus*)

F.5.2.7.1 Status and Management

West Indian manatees are divided into two subspecies: the Florida (*Trichechus manatus latirostris*) and Antillean (*Trichechus manatus manatus*) (Lefebvre et al., 2001). Both are currently listed as threatened under the ESA (82 *Federal Register* 16668), and are also considered a strategic stock, and depleted, under the MMPA (U.S. Fish and Wildlife Service, 2014a, 2014b, 2023f, 2023g). The USFWS reclassified the West Indian manatee from endangered to threatened in 2017 due to substantial improvements in the species' overall status since the original listing in 1967 (81 *Federal Register* 1000). A new 5-year status review was initiated in July 2021 (86 *Federal Register* 37178), and final revised Stock Assessment Reports were made available for both the Florida and Puerto Rico stocks in March of 2023 (88 Federal Register 18572; U.S. Fish and Wildlife Service, 2023f; U.S. Fish and Wildlife Service, 2023g). In October of 2023, a Proposed Rule by the USFWS announced two 90-day findings on petitions to reclassify the West Indian manatee, including subspecies and associated populations, under the ESA. The first petition requests the Puerto Rico stock of the Antillean subspecies to be classified as a distinct population segment with the designation of critical habitat. The second petition requests to reclassify both subspecies of West Indian Manatee as endangered. The information provided in both petitions warranted plans to initiate a new status review as of the date the petition findings were published (88 *Federal Register* 70634).

Both subspecies of West Indian manatees may be found in the Study Area, although the Antillean manatee (i.e., Puerto Rico stock) only occurs in the Caribbean Sea Large Marine Ecosystem, extending eastward to Puerto Rico (Lefebvre et al., 2001). The Florida stock is closely monitored and managed by the USFWS and the Florida Fish and Wildlife Conservation Commission (U.S. Fish and Wildlife Service, 2014b). The Florida manatee population is divided into four management units: the Upper St. Johns River, Atlantic Coast, Southwest Florida, and Northwest Florida. The Puerto Rico stock is managed by the USFWS Caribbean Ecological Services Field Office in Boquerón, Puerto Rico with jurisdiction only in Puerto Rico and the U.S. Virgin Islands (U.S. Fish and Wildlife Service, 2014a). This population is currently considered as a single population with minimal, if any subdivisions within Puerto Rico.

F.5.2.7.2 Habitat and Geographic Range

Manatees are found in coastal marine, brackish, and freshwater habitats. They are typically found in seagrass beds, canals, creeks, embayments, and lagoons near the mouths of rivers, sloughs, and salt marshes (Gaskins et al., 2020; Lefebvre et al., 2000). Habitat selection is influenced by food, water temperature, and the availability of freshwater sources. Females with calves are influenced by additional factors when selecting habitats, including ambient noise, currents, and increased amounts of forage (Gannon et al., 2007). Groups of manatees, sometimes in the hundreds, often congregate near sources of warm water during the colder winter months (Deutsch et al., 2003; Jefferson et al., 2008) (Jefferson et al., 2015).

As part of the 12-month finding to revise critical habitat, the USFWS recognized the significance of warm water to the survival of the Florida manatee and the importance of availability and adequacy of warm-water refuges. Additional features essential to conservation that were considered in the analysis for revising critical habitat included adequate forage within dispersal distance of a warm-water refuge, areas needed for calving and nursing, and important travel corridors for movements throughout Florida and beyond (75 *Federal Register* 1574).

Florida manatees are found throughout the southeastern United States and their distribution shifts seasonally. Since manatees are a subtropical species with little tolerance for cold, they are generally restricted to the inland and coastal waters of peninsular Florida during the winter, when they shelter in or near warm-water springs, industrial effluents, and other warm-water sites (Hartman, 1979; Lefebvre et al., 2001). The Florida manatee, in particular, is highly susceptible to cold stress and mortality when water temperatures drop below 20° C (68° F) (Irvine, 1983).

In the Study Area, the West Indian manatee (Florida subspecies/stock) has the potential to occur from the mid-Atlantic United States to the Caribbean. The primary range extends along both the Atlantic and Gulf coasts of Florida, while the secondary range extends as far north as the Chesapeake and Delaware Bays on the east side and into the northern Gulf of Mexico as far west as Texas. Occasional animals have been sighted as far north as Massachusetts and as far south as the Bahamas and Cuba (U.S. Fish and Wildlife Service, 2023f). During the winter, Florida manatees shelter in or near warm-water springs and spring complexes, heated industrial effluents, and other warm-water sites such as passive thermal basins (U.S. Fish and Wildlife Service, 2023f). During the warmer months, manatees are reported regularly in Florida, coastal Georgia, South Carolina, Alabama, Mississippi, and eastern Louisiana (Cloyed et al., 2019; Hieb et al., 2017), but can leave these areas to disperse great distances (Lefebvre et al., 2001; U.S. Fish and Wildlife Service, 2023f); and increased sightings have been reported in the mid-Atlantic states such as North Carolina and Virginia between June and October (Cummings et al., 2014).

The Antillean subspecies of the West Indian manatee is only found in eastern Mexico and Central America, northern and eastern South America, and in the Greater Antilles (Lefebvre et al., 1989) within the Caribbean Large Marine Ecosystem. All studies suggest that manatees in Puerto Rico are most often detected in protected areas around cays, in secluded bays, and in shallow seagrass beds east of San Juan; the east, south, and southwest coasts; and not far from freshwater sources (U.S. Fish and Wildlife Service Caribbean Field Office, 2009). Lefebvre et al. (2000) debates that manatees in Puerto Rico most likely do not exhibit seasonal long-distance travel patterns because, unlike the manatees in Florida, they do not need to seek thermal refuges during the winter months. Telemetry studies confirm that individual manatees in Puerto Rico have fidelity to relatively discrete areas but may move longer distances through nearshore and coastal waters to other use areas (Slone et al., 2006).

The distribution of the Antillean manatee extends eastward only to Puerto Rico, except for a few extralimital sightings: one sighting reported in 1988 in St. Thomas, U.S. Virgin Islands, and a more recent confirmed sighting of two manatees in St. Croix (May to June 2018), and transient animals are known to occur in the Lesser Antilles (Lefebvre et al., 2001). Manatees in Puerto Rico are thought to be more widespread than previously understood (Collazo et al., 2019). Manatee distribution in Puerto Rico during surveys from June 2010 to March 2014 followed the general historical distribution for the island (Collazo et al., 2019). The distribution and area-specific concentrations observed (e.g., Ceiba, Patillas, Guayama, Salinas, Guayanilla, Guánica, and Cabo Rojo–Mayagüez), validates the selection of hotspots as the most important areas for manatees, with the exception of Rio Grande's Ensenada Comezón and Punta Petrona in Santa Isabel, which should be considered hotspots in future studies (Collazo et al., 2019). The offshore islands of Puerto Rico including Caja de Muertos, Culebra, and Vieques are considered significant biogeographic features, although manatees do not use the western offshore islands of Mona and Desecheo (U.S. Fish and Wildlife Service, 2014a). Mona Passage constitutes a migratory barrier to these islands since it is characterized by strong currents and high surf. There have been few sightings in Caja de Muertos on the south and Culebra Island on the east. In contrast, Vieques Island contains extensive seagrass beds and manatees have been observed traveling to and from the east coast of Puerto Rico across the Vieques Sound (i.e., Pasaje de Vieques) (Magor, 1979; Slone et al., 2006). Radio tagging techniques in Puerto Rico have documented the general behavior of manatee populations, where males displayed more extensive movements than females (Slone et al., 2006). Areas of higher manatee use and activity are associated with low wave action, available food, and sources of freshwater for drinking (Collazo et al., 2019; Slone et al., 2006). As such, manatees are sighted more frequently in protected coastal areas having any three of the above characteristics (U.S. Fish and Wildlife Service, 2023g).

Critical habitat is designated at multiple inland rivers and coastal waterways throughout Florida (Figure 3.7-2, Designated Critical Habitat for West Indian Manatees in the Study Area), although the designation does not define any primary constituent elements. The designated critical habitat only overlaps with the Study Area within the St. Johns River (Mayport), Banana River (Port Canaveral), St. Marys River entrance channel (Kings Bay), and a small portion of inland waters encompassed by the South Florida Ocean Measurement Facility Testing Range boundary. However, the Mayport basin and the Trident basin are not considered critical habitat by the USFWS. A petition to revise manatee critical habitat was submitted in 2009. A 12-month finding on the petition, the USFWS stated that revisions should be made, including defining primary constituent elements. However, sufficient funding is not currently available (75 *Federal Register* 1574). In 2012, the USFWS issued a final rule establishing a manatee refuge in Kings Bay, Citrus County, Florida, which includes its tributaries and connected waters (77 *Federal Register* 15617). However, this new refuge does not overlap with the Study Area.

F.5.2.7.3 Population Trends

Demographic analyses indicates that the Florida stock of manatees is increasing or stable throughout much of their Florida range (Runge et al., 2004; Runge et al., 2007; Runge et al., 2017). However, several recent unusual mortality events have been declared for the West Indian manatee due to the high levels of mortalities of this species (Florida Fish and Wildlife Conservation Commission, 2022; U.S. Fish and Wildlife Service, 2023f). In 2021, there were 1,101 manatee mortalities, which is considerably higher compared to the 5-year average from 2016 to 2020 of 625 individuals (FWC 2022c). From 1996 to 2007, a total of 6 unusual mortality events were declared due to red tide (~450 deaths), and in 2010 through 2011 winter cold kills resulted in over 670 deaths. An adult survival rate analysis for the Florida manatee, conducted through the winter of 2005 to 2006, identified a range-wide survival rate of 96 percent (C.A. Langtimm, USGS, pers. Comm., 2011). A more recent analysis from data collected over 10 winters from 2008 to 2017, provided survival rate estimates by age class. Annual survival rate ranges were calculated as 0.85 to 0.89 for manatees less than 1 year old, and for those ages 4+, from 0.88 to 0.96 (Gowan et al., 2021). However, manatee survival rates will likely be revised in consideration of the higher-than-normal mortality rates in recent years.

Population modeling of the Florida manatee by Runge et al., in 2015 and 2017 predicted that, assuming all current threats remain constant, there is less than a 2.5 percent chance that the southeastern U.S. population of Florida manatees will fall below 4,000 individuals over the next 100 years (Runge et al., 2015; Runge et al., 2017). An integrated population model developed in 2021 provided a retrospective analysis estimating the abundance of the West Indian manatee at 2,014 individuals in 1997 (the 95 percent credible interval was 1,861 to 2,229) to 3,019 in 2013 (credible interval: 2,668 to 3,431), and showed a general pattern of slow, but variable, population growth (Hostetler et al., 2021). The red tide event of 2013 contributed to an estimated net drop in the population of 331 (217 to 459) manatees for an annual population growth rate of 0.89 (0.85 to 0.93) (Hostetler et al., 2021). The estimate for 2016 was 2,966 manatees (credible interval: 2,551 to 3,434), with an average growth rate of 2 percent (credible interval: 1 to 3 percent) (Hostetler et al., 2021). Hostetler et al. (2018) estimated that the number of manatees in Florida in 2015 to 2016 was 8,810 (95 percent credible interval 7,520 to 10,280), of which 4,810 (credible interval: 3,820 to 6,010) were on the west coast of Florida and 4,000 (credible interval: 3,240 to 4,910) were on the east coast. The most recent data available from the 2021 to 2022 aerial surveys conducted by the Florida Fish and Wildlife Commission estimated a statewide abundance of 9,790 (credible interval: 8,350 to 11,730) individuals. This estimate is further divided into 2 coast wide estimates. The west coast abundance is 4,630 (credible interval: 3,960 to 5,420) individuals, and the east coast has an estimated 5,160 (credible interval: 3,940 to 6,980) individuals (Gowan et al., 2023).

The USFWS suggests that the Puerto Rico stock of manatees (Antillean subspecies) is at least stable and is possibly exhibiting a slight increase due to increasing numbers detected in annual surveys (U.S. Fish and Wildlife Service, 2014a). However, they caution that information from direct counts cannot be used to determine population trends. Population viability analyses used to predict the likely future status of a given population describes the Antillean manatee population with positive growth, which would continue as long as human-induced mortality does not exceed 5 percent of the population (Castelblanco-Martinez et al., 2012). Antillean manatee population estimates in Puerto Rico were developed from several surveys between June 2010 and March 2014 (Collazo et al., 2019). The average minimum island-wide estimate was 386 ± 89, which was similar to the maximum estimate of 360 suggested in 2005, but fewer than the 700 recently suggested by the Puerto Rico Manatee Conservation Center, and below the 532 average island-wide estimate suggested by the USFWS in 2017 (82 *Federal Register* 16668; Collazo et al. (2019). The USFWS's downlisting of the West Indian manatee from endangered to threatened in May 2017 indicates that populations are improving. Although the ranking of threats to the species have not changed, the impacts of those threats is considered lower due to a better understanding of the resiliency of the population (Runge et al., 2015). High adult survival provides the Florida manatee population with strong resilience to a variety of current and future threats, and a large estimated carrying capacity provides a buffer to short-term reductions in the population from episodic threats (Runge et al., 2017).

F.5.2.7.4 Predator and Prey Interactions

West Indian manatees are herbivorous and are known to consume more than 60 species of plants. They typically feed on bottom vegetation, plants in the water column, and shoreline vegetation, such as hyacinths and marine seagrasses (Reynolds et al., 2009). In some areas, they are known to feed on algae and parts of mangrove trees (Jefferson et al., 2015; Mignucci-Giannoni & Beck, 1998).

Although large sharks, crocodiles, and killer whales are all considered to be potential predators, there is little evidence to confirm this (Weller, 2009).

F.5.2.7.5 Species-Specific Threats

Threats to the Florida manatee include vessel strikes, cold stress, hurricanes, toxic red tide poisoning, habitat destruction (such as loss of seagrass), entrapment and/or crushing in water-control structures (e.g., gates, locks, etc.), entanglement in fishing gear, ingestion of marine debris, and other natural and human-made factors (U.S. Fish and Wildlife Service, 2023f). Vessel strikes are the single greatest cause of death for Florida manatees, which accounts for 20 to 25 percent of reported mortalities and are the leading human related threat to the West Indian manatee (Bassett et al., 2020). A review of research on the effectiveness of laws reducing boat speeds in areas of known manatee habitat indicated that reducing boat speeds in specific areas is an appropriate, reasonable, and defensible management action, although more studies on the effectiveness of boat speed reduction have been recommended (Calleson & Frohlich, 2007). A more recent investigation of manatee behavioral responses to boats in Florida found that boat speed did not affect the occurrence or intensity of manatee response, but that slower boat passes allowed manatees more time to respond, concluding that faster boats likely pose a greater risk of collision than slower boats (Rycyk et al., 2018). Entanglement and ingestion of marine debris is also a concern. In over 6,500 necropsy reports from 1993 to 2012, over 11 percent had ingested or showed evidence of entanglement in marine debris, or both (Reinert et al., 2017). Fifty of those manatees were found to have died as a direct result of entanglement or marine debris ingestion, with fishery related gear involved in 70 percent of those cases.

Expected losses of warm-water habitat are likely to cause a major change in the distribution of the population from the regions where manatees rely heavily on power-plant effluents for warmth in winter (Southwest and Atlantic regions), to the regions where manatees primarily use natural springs in winter (Northwest and Upper St. Johns regions) (Runge et al., 2017). An unusual mortality event was declared in December 2020 in the Atlantic Ocean and Florida for the West Indian manatee due to the high levels of mortalities of this species, and of particular concern are the starvation deaths due to lack of seagrass habitat, especially in the Indian River Lagoon (Florida Fish and Wildlife Conservation Commission, 2022; U.S. Fish and Wildlife Service, 2023f). In recent years, poor water quality in the lagoon has led to harmful algal blooms and widespread seagrass loss (Florida Fish and Wildlife Conservation Commission, 2022).

Unlike the Florida manatee, mass mortalities due to red tide or need for warm water habitats do not pose a threat to the Antillean manatee, given their location in tropical habitats. A single mass mortality (four males and one female) was documented in 2006 when the individuals were impacted by a large vessel in the San Juan Bay (U.S. Fish and Wildlife Service, 2014a). Similar to the Florida manatee, vessel strikes are the leading cause of human-induced mortalities of Antillean manatees (U.S. Fish and Wildlife Service, 2014a, 2023f, 2023g).

F.5.3 SPECIES NOT LISTED UNDER THE ENDANGERED SPECIES ACT

F.5.3.1 Bryde’s Whale (*Balaenoptera brydei/edeni*)

F.5.3.1.1 Status and Management

Bryde’s whales are among the least known of the baleen whales. The species-level taxonomy remains unresolved as well as the number of species or subspecies (Alves et al., 2010; Jefferson et al., 2015; Kato

& Perrin, 2009; Rosel et al., 2021). The Society for Marine Mammalogy's Committee on Taxonomy (2023) recognizes two subspecies of Bryde's whale: (1) *B. edeni* (Eden's whale) and (2) *B. brydei* (offshore Bryde's whale). In addition, the "pygmy form" of Bryde's whales is known as the Omura's whale (Kato & Perrin, 2009; Rice, 1998) has been described. Rosel et al. (2021) determined that Bryde's whales found in the Gulf of Mexico are in fact a distinct species, now designated as Rice's whales (*Balaenoptera ricei*) (Section F.5.2.4.4, Predator and Prey Interactions). The International Whaling Commission continues to use the name *Balaenoptera edeni* for all Bryde's-like whales, although at least three species are recognized.

Current genetic research confirms that gene flow among Bryde's whale populations is low and suggests that management actions treat each as a distinct entity to ensure survival of the species (Kanda et al., 2007).

F.5.3.1.2 Habitat and Geographic Range

Unlike other baleen whale species, Bryde's whales are restricted to tropical and subtropical waters and do not generally occur beyond latitude 40° in either the northern or southern hemisphere (Jefferson et al., 2015; Kato & Perrin, 2009). The primary range of Bryde's whales in the Atlantic is in tropical waters south of the Caribbean, outside the Study Area although may range as far north as Virginia (Kato & Perrin, 2009). Long migrations are not typical of Bryde's whales, although limited shifts in distribution toward and away from the equator in winter and summer have been observed (Best, 1996; Cummings, 1985).

F.5.3.1.3 Population Trends

A trend analysis has not been conducted for this stock.

F.5.3.1.4 Predator and Prey Interactions

Bryde's whales primarily feed on schooling fishes and are lunge feeders. Prey includes anchovy, sardine, mackerel, herring, krill, and pelagic red crab (Baker & Madon, 2007; Jefferson et al., 2015; Nemoto & Kawamura, 1977). Like humpback whales, Bryde's whales were observed using "bubble nets" to herd prey (Jefferson et al., 2015; Kato & Perrin, 2009). Bryde's whales are known to be prey for killer whales, as evidenced by an aerial observation of 15 killer whales attacking a Bryde's whale in the Gulf of California (Weller, 2009).

F.5.3.1.5 Species-Specific Threats

There are no significant species-specific threats to Bryde's whales, though they are susceptible to vessel strike and fishing gear entanglement.

F.5.3.2 Humpback Whale (*Megaptera novaengliae*)

F.5.3.2.1 Status and Management

Humpback whales, as a globally distributed species, are divided into 14 distinct population segments and NMFS revised the listing status of each breeding population in 2016 (81 *Federal Register* 62259). The West Indies distinct population segment that occurs within the Study Area did not warrant listing under the ESA, as they are neither in danger of extinction nor likely to become so in the foreseeable future. All humpback whales feeding in the North Atlantic are considered part of the West Indies distinct population segment (Bettridge et al., 2015), including the Gulf of Maine stock. The West Indies distinct population segment feeding range primarily includes the Gulf of Maine, eastern Canada, as well as western Greenland, and breeding grounds include waters of the Dominican Republic and Puerto Rico (81 *Federal Register* 62259).

For management purposes in U.S. waters, NMFS identified stocks that are based on feeding areas. Although the western North Atlantic population was once treated as a single management stock, the Gulf of Maine stock has been identified as a discrete subpopulation based on strong fidelity of humpbacks feeding in that region (Waring et al., 2016). The Gulf of Maine stock is the only stock of humpbacks in the Atlantic managed under NMFS jurisdiction. However, it should be noted that several other discrete humpback whale subpopulations, based on feeding grounds, are in the western North Atlantic, including the Gulf of St. Lawrence, Newfoundland/Labrador, and western Greenland (Waring et al., 2016), though all belong to the West Indies distinct population segment.

F.5.3.2.2 Habitat and Geographic Range

Humpback whales are distributed worldwide in all major oceans and most seas. Most humpback whale sightings are in nearshore and continental shelf waters; however, humpback whales frequently travel through deep oceanic waters during migration (Calambokidis et al., 2001; Clapham & Mattila, 1990). Humpback whales of the western North Atlantic are typically found in Labrador Current, North Atlantic Gyre and Gulf Stream open-ocean areas during seasonal migrations from northern latitude feeding grounds, occupied during the summer, to southern latitude calving and breeding grounds occupied in the winter (Waring et al., 2016). The Gulf of St. Lawrence, Newfoundland Grand Banks, West Greenland, and Scotian Shelf are summer feeding grounds for humpbacks (Cetacean and Turtle Assessment Program, 1982; Kenney & Winn, 1986; Stevick et al., 2006; Whitehead, 1982). The Gulf of Maine is also one of the principal summer feeding grounds for humpback whales in the North Atlantic. The largest numbers of humpback whales are present from mid-April to mid-November. Other feeding locations in this ecosystem are Stellwagen Bank, Jeffreys Ledge, the Great South Channel, the edges and shoals of Georges Bank, Cashes Ledge, and Grand Manan Banks (Cetacean and Turtle Assessment Program, 1982; Kenney & Winn, 1986; Stevick et al., 2006; Weinrich et al., 1997; Whitehead, 1982). LaBrecque et al. (2015a) delineated a humpback whale feeding area in the Gulf of Maine, Stellwagen Bank, and Great South Channel, substantiated through vessel-and aerial-based survey data, photo-identification data, radio-tracking data, and expert judgment (Figure F.5-7). Humpback whales feed in this area from March through December. Humpback feeding habitats are typically shallow banks or ledges with high seafloor relief (Hamazaki, 2002; Payne et al., 1990).

Additionally, as the West Indies population increases and is no longer considered at risk for extinction, their distribution has expanded outside of their known primary feeding areas into nearshore urban waterways such as the New York-New Jersey Harbor estuary and New York Bight apex (Brown et al., 2018; Brown et al., 2019; King et al., 2021; Smith et al., 2022b; Zoidis et al., 2021), which increases exposure to heavy commercial and recreational vessel presence (Brown et al., 2019; King et al., 2021; Zoidis et al., 2021). Using opportunistic sighting data and known catalogued individuals that feed in the Gulf of Maine, a study (Brown et al., 2022), showed site fidelity of predominately juvenile individuals in the New York Bight apex with a 31.3 percent annual return rate, while the presence of adults and mother-calf pairs were rare. Long-term data for humpback whale presence in this area is lacking, and therefore more research is needed to infer conclusions regarding population trends, sighting increases, and additional factors driving shifts in distribution throughout the northwest Atlantic (Brown et al., 2022).

Individual variability in the timing of migrations may result in the presence of individuals in high-latitude areas throughout the year (Straley, 1990). Records of humpback whales off the United States mid-Atlantic coast (New Jersey to North Carolina) from November through March suggest these waters may represent a supplemental winter-feeding ground used by juvenile and mature humpback whales of U.S. and Canadian North Atlantic stocks (LaBrecque et al., 2015a).

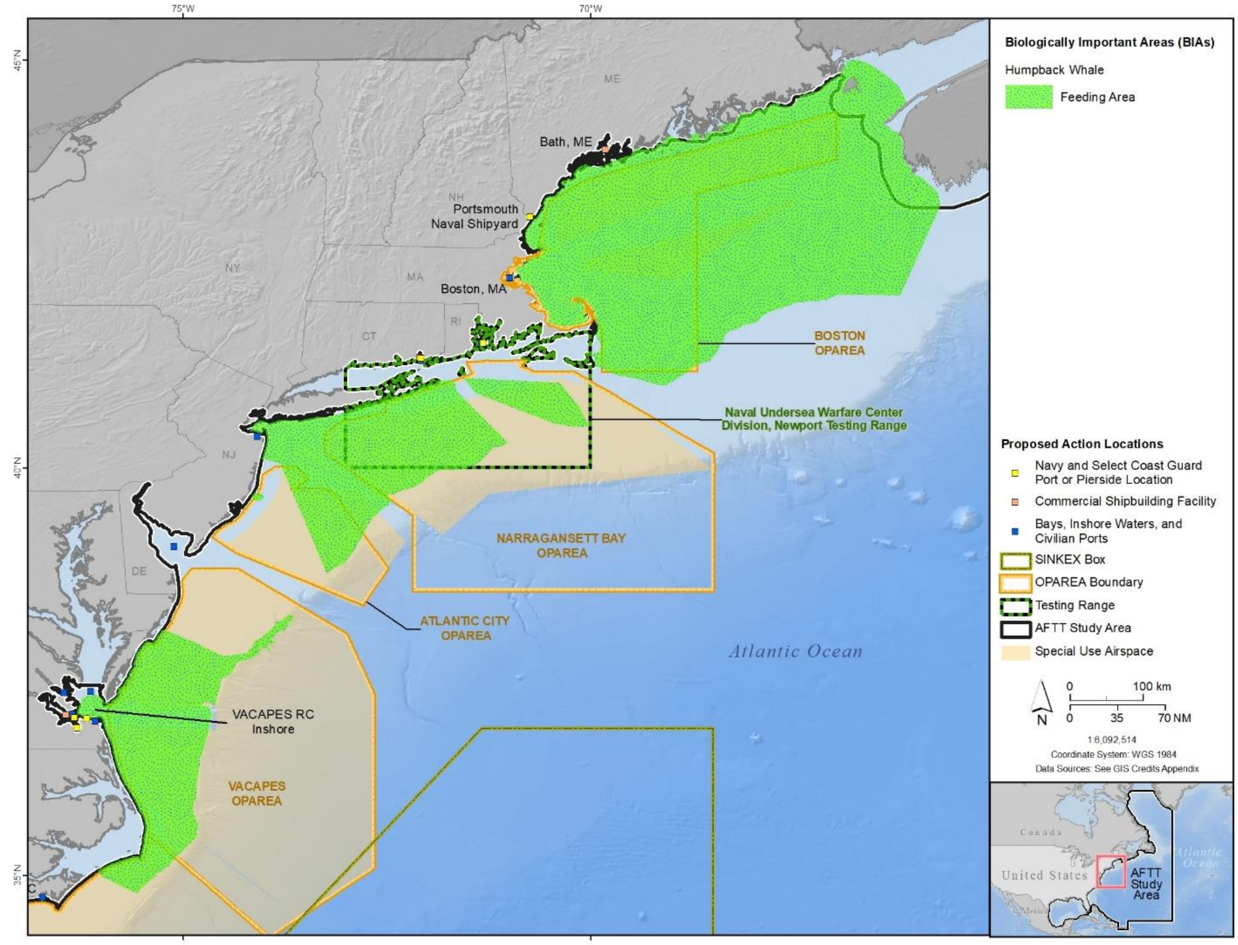


Figure F.5-7: Biologically Important Areas for Humpback Whales in the Study Area

Humpbacks are most likely to occur near the mouth of the Chesapeake Bay and coastal waters of Virginia Beach between November and March; however, they could be found in the area year-round, based on sighting and stranding data in both mid-Atlantic waters and the Chesapeake Bay itself (Aschettino et al., 2022; Barco et al., 2002). Photo-identification data support the repeated use of the mid-Atlantic region by individual humpback whales (Aschettino et al., 2022; Barco et al., 2002). Vessel surveys offshore of Virginia show site fidelity in the winter months for some individuals, and hierarchical state-space modeling of humpback whale tag data shows a high level of occurrence (82 percent of all modeled whale locations) within the shipping channels—an important high-use area by both the Action Proponent and commercial traffic (Aschettino et al., 2020).

Aerial and vessel monitoring conducted offshore of Cape Hatteras, North Carolina, in Onslow Bay, North Carolina, and offshore of Jacksonville, Florida confirmed winter occurrence of humpback whales in these three areas of the Atlantic (Surrey-Marsden et al., 2018; U.S. Department of the Navy, 2013a; Zoodsma et al., 2016), as well as observations in Onslow Bay during the spring months (U.S. Department of the Navy, 2013a).

There are occasional reports of humpback whales in the Gulf of Mexico, but those sightings should be considered extralimital.

F.5.3.2.3 Population Trends

Current data suggest that the Gulf of Maine humpback whale stock is steadily increasing in numbers (Waring et al., 2016). This is consistent with an estimated average growth trend of 3.1 percent (Standard Error =0.005) in the North Atlantic population overall for the period between 1979 through 1993 (Stevick et al., 2003).

F.5.3.2.4 Predator and Prey Interactions

Humpback whales feed on a variety of invertebrates and small schooling fishes. The most common invertebrate prey are krill; the most common fish prey are herring, mackerel, sand lance, sardines, anchovies, and capelin (Clapham & Mead, 1999). Feeding occurs both at the surface and in deeper waters, wherever prey is abundant. The humpback whale is the only species of baleen whale that shows strong evidence of cooperation when feeding in large groups (D'Vincent et al., 1985). Humpback whales were observed using “bubble nets” to herd prey (Jefferson et al., 2015). Bubble nets are a feeding strategy where the whales dive and release bubbles of air that float up in a column and trap prey inside; the humpbacks then lunge through the column of trapped prey to feed at the surface (Hain et al., 1982).

Sensors attached to humpback whales foraging in Stellwagen Bank, Massachusetts, allowed researchers to measure in fine detail the orientation and movement patterns of both humpback whales and their prey at meaningful ecological scales (Friedlaender et al., 2009). Findings indicate that differences between surface and bottom feeding behaviors in humpback whales correlated with vertical changes in the distribution and abundance of their primary prey, sand lance. In addition to prey abundance, other factors relate to humpback whale surface feeding in the Gulf of Maine, such as time of day and tidal height (Hazen et al., 2009). Characteristics of the prey, such as light emitted and the shape of the schools, also relate to humpback whale surface feeding.

This species is known to be attacked by both killer whales and false killer whales, as evidenced by tooth rake scars on their bodies, including pectoral flippers and tail flukes (Jefferson et al., 2015).

F.5.3.2.5 Species-Specific Threats

NMFS has declared an unusual mortality event for humpback whales along the Atlantic coast beginning in January 2016 (National Marine Fisheries Service, 2024b). Increased mortalities have been observed from Maine through North Carolina. As of winter 2023, 178 stranding cases have been reported and about half of those cases have been examined. Of those examined, 40 percent of cases showed evidence of human interaction either in the form of entanglement or vessel strike; however, investigations are still underway to determine the cause of many of the strandings.

F.5.3.3 Minke Whale (*Balaenoptera acutorostrata*)

F.5.3.3.1 Status and Management

Minke whales are the smallest species of mysticete in the Study Area and are classified as a single species with three subspecies recently recognized: *Balaenoptera acutorostrata* in the North Atlantic, *Balaenoptera acutorostrata scammoni* in the North Pacific, and a subspecies that is not formally but generally called the dwarf minke whale, which mainly occurs in the southern hemisphere (Hayes et al., 2022; Jefferson et al., 2015).

There are four recognized populations in the North Atlantic: Canadian east coast, west Greenland, central North Atlantic, and northeastern North Atlantic (Donovan, 1991). As stock structure is still being researched, minke whales off the eastern coast of the United States are considered, for now, to be part of the Canadian east coast stock, which inhabits the area from the western half of the Davis Strait (45° West) to the Gulf of Mexico (Hayes et al., 2022). The relationship between this stock and the other three stocks is uncertain.

F.5.3.3.2 Habitat and Geographic Range

Minke whales have a cosmopolitan distribution in temperate and tropical waters and generally occupy waters over the continental shelf, including inshore bays and even occasionally estuaries (Hayes et al., 2020). There appears to be a strong seasonal component to minke whale distribution on both the continental shelf and in deeper, off-shelf waters. Spring to fall are times of relatively widespread and common acoustic occurrence on the shelf (e.g., Risch et al., 2013)., while September through April is the period of highest acoustic occurrence in deep-ocean waters throughout most of the western North Atlantic. The minke whale is common and widely distributed within the U.S. Exclusive Economic Zone in the Atlantic Ocean (Cetacean and Turtle Assessment Program, 1982).

During summer and early fall, minke whales are found throughout the lower Bay of Fundy (Ingram et al., 2007). Spring and summer are times of relatively widespread and common occurrence and are the seasons when the whales are most abundant in New England waters. In New England waters during fall there are fewer minke whales, while during winter the species appears to be largely absent.

LaBrecque et al. (2015a) delineated two minke whale feeding areas: (1) waters less than 200 m in the southern and southwestern section of the Gulf of Maine, including Georges Bank, the Great South Channel, Cape Cod Bay, Massachusetts Bay, and Stellwagen Bank, and (2) shallow waters around Parker Ridge and Cashes Ledges in the central Gulf of Maine (Figure F.5-8). These feeding areas were substantiated by vessel- and aerial-based surveys, sightings from whale-watching vessels, and expert judgment. Minke whales would be expected in both feeding areas from March through November.

Minke whales occur in the warmer waters of the southern United States during winter. While no minke whale mating or calving grounds have been found in U.S. Atlantic waters LaBrecque et al. (2015a), some data suggest a potential winter breeding area offshore of the southeastern United States and the Caribbean based on seasonal migration patterns, acoustic survey results, calf-stranding records, and sightings of mother-calf pairs in Onslow Bay, North Carolina, and offshore of Jacksonville, Florida (Risch et al., 2014). Acoustic monitoring using marine autonomous recording units deployed between 60 and 150 km offshore of Jacksonville, Florida, revealed continuous vocalizations at the deep-water sites during the winter, while vocalization events were completely absent during the fall suggesting a strong seasonal pattern of occurrence in this area (Oswald et al., 2016). Between 2015 and 2021, 12 minke whales were sighted during Navy visual surveys off the coast of Virginia Beach, Virginia. Ongoing acoustic monitoring efforts offshore of Cape Hatteras since March 2012 in water depths of 950 m resulted in frequent detections of minke whales (Debich et al., 2016; Stanistreet et al., 2013), suggesting spring occurrence in this area as minke whales begin to migrate to northern feeding grounds for the summer months.

(Mitchell, 1991) summarized several winter records of minke whale sightings off the southeast United States, Cuba, Puerto Rico, and the Antilles, hinting at a possible winter distribution in the West Indies, and in the mid-ocean south and east of Bermuda. Although they are not typically expected to occur within the Gulf of Mexico, observation records also exist for mostly immature individuals in the Gulf of Mexico and Florida Keys (Stewart & Leatherwood, 1985; Waring et al., 2013).

F.5.3.3.3 Population Trends

A trend analysis has not been conducted for this stock (Waring et al., 2016).

F.5.3.3.4 Predator and Prey Interactions

This species preys on small invertebrates and schooling fishes, such as capelin, haddock, sand eels, pollock, herring, and cod (Jefferson et al., 2015; Kuker et al., 2005; Lindstrom & Haug, 2001; Reeves et al., 2002b). Similar to other rorquals, minke whales are lunge feeders, often plunging through patches of shoaling fish or krill (Hoelzel et al., 1989; Jefferson et al., 2015).

Minke whales are prey for killer whales (Ford et al., 2005); a common minke was observed under attack by killer whales near British Columbia (Weller, 2009).

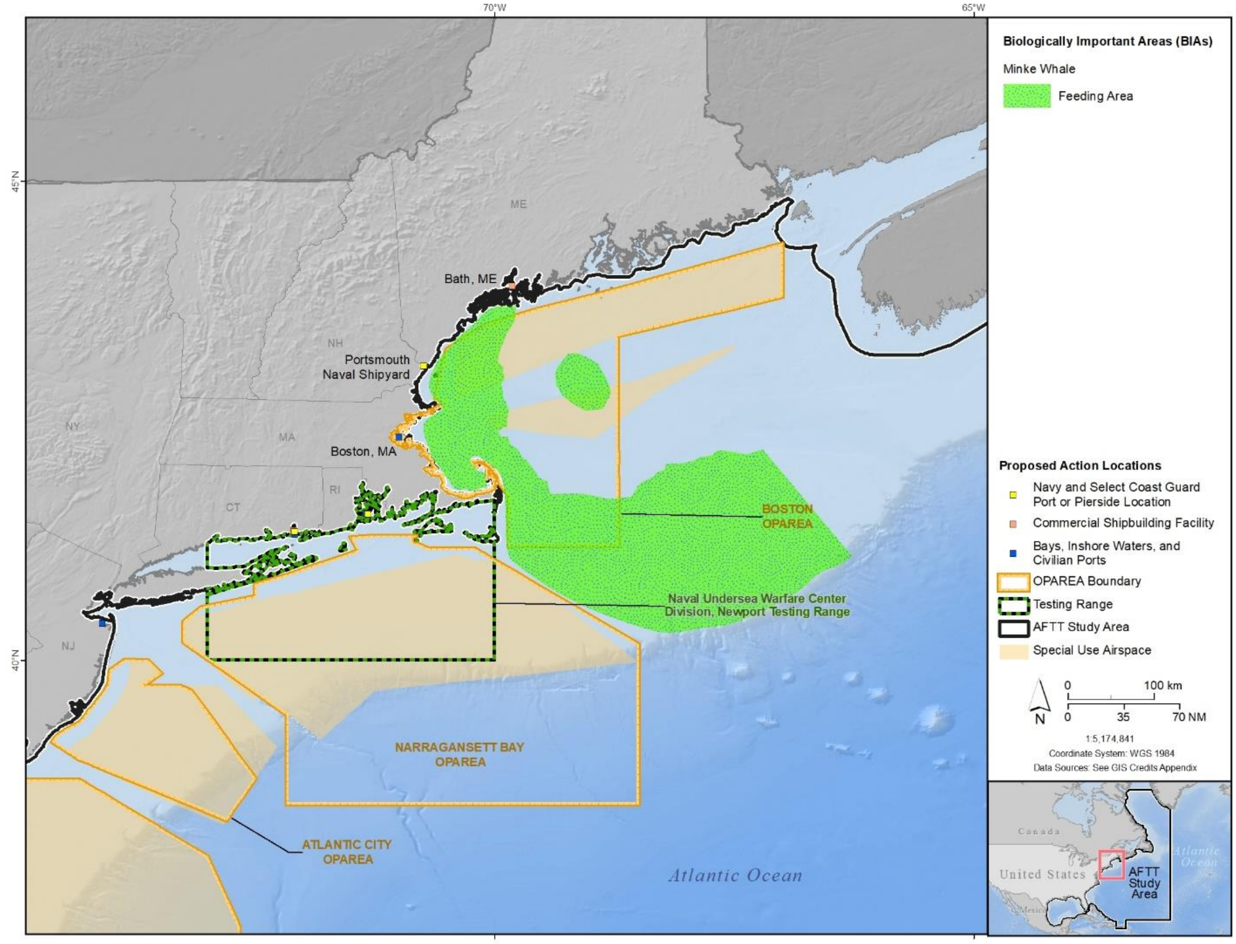


Figure F.5-8: Biologically Important Areas for Minke Whales in the Study Area

F.5.3.3.5 Species-Specific Threats

Minke whales are documented as bycatch in gillnets in the mid-Atlantic and northeast fisheries. This species was also documented as bycatch in pelagic longline fisheries operating in the Atlantic Ocean, Caribbean, and Gulf of Mexico (Zollett, 2009). Minke whale mortality and serious injury has also been documented as a result of interactions with an unknown Canadian fishery.

An unusual mortality event for minke whales was declared from 2017 to 2024 along the Atlantic Coast (National Marine Fisheries Service, 2024c). During that timeframe, 172 minke whale mortalities were observed from Maine to South Carolina. Full or partial necropsy examinations were conducted on more than 60 percent of the whales, with preliminary findings showing evidence of human interactions or infectious diseases in many of the whales. However, these findings are not consistent across all of the whales examined.

F.5.3.4 Dwarf/Pygmy Sperm Whale (*Kogia sima* and *Kogia breviceps*)

F.5.3.4.1 Status and Management

Before 1966, dwarf and pygmy sperm whales were thought to be a single species, until form and structure distinction were shown (Handley, 1966); misidentifications of these two species are still common (Jefferson et al., 2015). Dwarf and pygmy sperm whales are not often observed at sea, but they are among the more frequently stranded cetaceans (Caldwell & Caldwell, 1989; Jefferson et al., 2015; McAlpine, 2009). Rare sightings indicate they may avoid human activity, and they are rarely active at the sea surface. They usually appear slow and sluggish, often resting motionless at the surface with no visible blow (Baird, 2005; Jefferson et al., 2015). Because of the scarcity of biological information available for individual dwarf and pygmy sperm whales, the difficulty of species-level identifications, and the lack of data on individual stock structure and abundance estimates, dwarf and pygmy sperm whales are presented collectively here with species-specific information if available.

Although virtually nothing is known of population status for these species, stranding frequency suggests they may not be as uncommon as sighting records would indicate (Jefferson et al., 2015; Maldini et al., 2005). The western North Atlantic population(s) and the northern Gulf of Mexico population(s) are considered separate stocks for management purposes, but there is no genetic evidence that these two populations differ (Hayes et al., 2021).

F.5.3.4.2 Habitat and Geographic Range

Dwarf and pygmy sperm whales appear to be distributed worldwide from temperate to tropical waters (Caldwell & Caldwell, 1989; McAlpine, 2002). Both species may be found in the Gulf Stream and North Atlantic Gyre open-ocean areas. Most sightings are in the Gulf Stream, perhaps an artifact of survey effort rather than a reflection of actual distribution. Dwarf and pygmy sperm whales can occur close to shore and occasionally over the outer continental shelf. However, several studies show that they may also occur beyond the continental shelf edge (Bloodworth & Odell, 2008; MacLeod et al., 2004). The pygmy sperm whale may frequent more temperate habitats than the dwarf sperm whale, which is more of a tropical species. The dwarf sperm whale may also have a more pelagic distribution, and dive deeper during feeding bouts, than pygmy sperm whales (Barros & Wells, 1998). Hodge and Read (2018) used passive acoustic monitoring to determine that dwarf and pygmy sperm whales are common in deep waters along the continental shelf break and slope between Virginia and Florida, and more common than suggested by visual sightings. Passive acoustics have also been used to estimate density of dwarf/pygmy sperm whales in the Gulf of Mexico, finding that densities derived from acoustic data are substantially higher than those developed from visual sighting data (Hildebrand et al., 2019). At these recording sites, vocalizations were detected more during the day than night hours, and some level of seasonality was evident in the recordings (Hildebrand et al., 2019). A relative lack of oceanic sightings may have more to do with the difficulty of detecting and identifying these animals at sea and lack of effort, in comparison to any real distributional preferences.

In the Study Area, dwarf and pygmy sperm whales are found primarily in the Northeast and Southeast United States Continental Shelf Large Marine Ecosystems, the Gulf of Mexico, and Caribbean Sea (Bloodworth & Odell, 2008; Caldwell & Caldwell, 1989; Cardona-Maldonado & Mignucci-Giannoni, 1999). A stranded pygmy sperm whale on the north shore of the Gulf of St. Lawrence represents the northernmost record for this species in the western Atlantic (Measures et al., 2004).

Despite the difficulty of sighting these species visually, aerial surveys of mid-Atlantic portions of the Study Area (near Norfolk Canyon) in 2018 and 2019 resulted in 10 observations totaling 17 individuals; 4 encounters involved mother-calf pairs (Cotter, 2019).

Pygmy sperm whales were one of the most sighted species in the northern Gulf of Mexico from 1992 to 1994 and from 1996 to 2001 (Mullin & Fulling, 2004). Fulling and Fertl (2003) noted a concentration of sightings in continental slope waters near the Mississippi River Delta. The delta is considered an important area for cetaceans in the northern Gulf of Mexico because of its high levels of productivity associated with oceanographic features. Data from the Gulf of Mexico suggest that dwarf and pygmy sperm whales may associate with frontal regions along the continental shelf break and upper continental slope, where squid densities are higher (Baumgartner et al., 2001; Jefferson et al., 2015).

F.5.3.4.3 Population Trends

Trend analyses have been conducted for dwarf/pygmy sperm whales in both the Gulf of Mexico and Western North Atlantic stocks. However, for both regions, there is high uncertainty in the abundance estimates, and methodological factors make it difficult to compare across years. While there appears to be an increasing trend in the Western North Atlantic stock, this should be interpreted with caution (Hayes et al., 2021).

F.5.3.4.4 Predator and Prey Interactions

Dwarf and pygmy sperm whales feed on cephalopods and, less often, on deep-sea fishes and shrimp (Beatson, 2007; Caldwell & Caldwell, 1989). A study showed cephalopods (squid) were the primary prey of pygmy sperm whales in the Pacific Ocean, making up 78.7 percent of prey abundance and 93.4 percent contribution by mass. Stomach samples revealed an extreme diversity of cephalopod prey, with 38 species from 17 families (West et al., 2009).

Dwarf and pygmy sperm whales are likely subject to occasional killer whale predation, as are other whale species.

F.5.3.4.5 Species-Specific Threats

The northern Gulf of Mexico stocks of dwarf and pygmy sperm whales were among the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill. Injury quantification determined that 15 percent of dwarf and pygmy sperm whales in the Gulf of Mexico were exposed to oil, resulting in 5 percent excess mortality above baseline conditions, 7 percent excess failed pregnancies, and a 6 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the northern Gulf of Mexico dwarf and pygmy sperm whale stocks will take an estimated 11 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.5 Beaked Whales (Various Species)

Six species of beaked whales are known in the western North Atlantic Ocean: goose-beaked whale (formerly Cuvier's beaked whale) (*Ziphius cavirostris*), northern bottlenose whale (*Hyperoodon ampullatus)* discussed in Section F.5.3.6, and four members of the genus *Mesoplodon* — True's (*M. mirus*), Gervais' (*M. europaeus*), Blainville's (*M. densirostris*), and Sowerby's (*M. bidens*) beaked whales. Goose-beaked, Blainville's, and Gervais' beaked whales are also known to regularly occur in the Gulf of Mexico based on stranding or sighting data (Hansen et al., 1995; Würsig et al., 2000). Sowerby's beaked whale in the Gulf of Mexico is considered extralimital because there is only one known stranding of this species (Bonde & O'Shea, 1989) and occurrence is normally in northern temperate waters of the North Atlantic (Mead, 1989a). With the exception of the goose-beaked whale and northern bottlenose whale, beaked whales are nearly indistinguishable at sea (Coles, 2001). Because of the scarcity of biological information available for individual species, the difficulty of species-level identifications for *Mesoplodon*, and the lack of data on individual stock structure and abundance estimates, goose-beaked, True's, Gervais', Blainville's, and Sowerby's beaked whales are presented collectively here with species-specific information if available.

F.5.3.5.1 Status and Management

Stock structure of beaked whales in the Atlantic, Gulf of Mexico, and U.S. Virgin Islands is unknown; however, these are assumed separate for management purposes.

F.5.3.5.2 Habitat and Geographic Range

Goose-beaked, True's, Gervais', Blainville's, and Sowerby's beaked whales are found in Labrador Current, North Atlantic Gyre, and Gulf Stream open-ocean areas and are also known to occur in the Northeast U.S. Continental Shelf, Scotian Shelf, and Newfoundland-Labrador Shelf Large Marine Ecosystems. The continental shelf margins from southern Nova Scotia to Cape Hatteras have been identified as key areas for beaked whales in a global review by MacLeod and Mitchell (2006). Goose-beaked, Gervais', Blainville's, and True's beaked whales may also occur in the Southeast U.S. Continental Shelf Large Marine Ecosystem, while goose-beaked, Gervais' and Blainville's beaked whales may occur in the Gulf of Mexico and Caribbean Sea Large Marine Ecosystems.

Goose-beaked whale is one of the more commonly seen and the best known. Similar to other beaked whale species, this oceanic species generally occurs in waters past the edge of the continental shelf and occupies almost all temperate, subtropical, and tropical waters of the world, as well as subpolar and even polar waters in some areas (Waring et al., 2014). The distribution of goose-beaked whales is poorly known and is based mainly on stranding records (Leatherwood et al., 1976). Strandings were reported from Nova Scotia along the eastern U.S. coast south to Florida, around the Gulf of Mexico, and within the Caribbean (Cetacean and Turtle Assessment Program, 1982; Heyning, 1989; Houston, 1990; Leatherwood et al., 1976; MacLeod, 2006; Mignucci-Giannoni et al., 1999). Goose-beaked whale sightings have occurred principally along the continental shelf edge in the mid-Atlantic region off the northeast U.S. coast (Cetacean and Turtle Assessment Program, 1982; Hamazaki, 2002; Palka, 2006; Waring et al., 1992; Waring et al., 2001) in late spring or summer, although strandings and sightings were reported in the Caribbean Sea and the Gulf of Mexico as well (Dalebout et al., 2006). Goose-beaked whales are generally sighted in waters with a bottom depth greater than 200 m and are frequently recorded in waters with bottom depths greater than 1,000 m (Falcone et al., 2009; Jefferson et al., 2008; Jefferson et al., 2015).

True's beaked whales appear to occur only in temperate waters, and possibly only in warm temperate waters. Most records of occurrence in the northwest Atlantic suggest a probable relation with the Gulf Stream (MacLeod, 2000; Mead, 1989b).

Gervais' beaked whale occurs only in the Atlantic Ocean and Gulf of Mexico, within a range both north and south of the equator to a latitude of 40° (Jefferson et al., 2008; Jefferson et al., 2015; MacLeod, 2006). Although the distribution seems to range across the entire temperate and tropical Atlantic, most records are from the western North Atlantic waters from New York to Texas (more than 40 published records), and they are the most common species of *Mesoplodon* to strand along the U.S. Atlantic coast (Waring et al., 2014).

Sowerby's beaked whales appear to inhabit more temperate waters than many other members of the genus. They are the most northerly distributed of Atlantic species of *Mesoplodon*, and are found in cold temperate waters of the North Atlantic Ocean, generally north of 30˚ N. In the Study Area, they range from Massachusetts to Labrador (MacLeod et al., 2006; Mead, 1989a). There were several at-sea sightings off Nova Scotia and Newfoundland, from New England waters north to the ice pack (MacLeod et al., 2006; Waring et al., 2010). Sowerby's beaked whale occurrence in the Gully Marine Protected Area (east of Nova Scotia) increased during the period from 1988 to 2011 (Whitehead, 2013).

Blainville's beaked whales are one of the most widely distributed of the distinctive toothed whales in the *Mesoplodon* genus (Jefferson et al., 2008; MacLeod et al., 2006). In the Study Area, this species is known to occur in enclosed deep-water seas, such as the Gulf of Mexico and Caribbean Sea. There are records for this species from the eastern coast of the United States and Canada, from as far north as Nova Scotia and south to Florida and the Bahamas (MacLeod & Mitchell, 2006; Mead, 1989a).

Starting January 2015, aerial surveys conducted in the offshore area from Wilmington, North Carolina to near Norfolk Canyon and have resulted in sightings of True's (Number [N] = 2), Sowerby's (N = 1), and Gervais' (N = 8) beaked whales, in addition to sightings only identified to *Mesoplodon* species (N = 27) (Cotter, 2019). Aerial sightings of goose-beaked whale were more common (N = 69). Vessel-based surveys offshore of Norfolk, Virginia, sighted True's (N = 3), Sowerby's (N = 3), and goose-beaked (N = 5) beaked whales, as well as unidentified *Mesoplodons* (N = 4) and unidentified beaked whales (N = 4) between 2016 and 2021 (Halpin et al., 2009).

McLellan et al. (2018) determined that Cape Hatteras is an exceptionally important habitat area for both goose-beaked and *Mesoplodon* species, while (Foley et al., 2021) noted that satellite-tagged goose-beaked whales demonstrate high site fidelity in small core areas in this region. During aerial surveys conducted between May 2011 and December 2014, beaked whales were observed in every month of the year offshore of Cape Hatteras, with goose-beaked whale being the most commonly encountered beaked whale species (McLellan et al., 2015). The highest number of beaked whale sightings occurred between May and August and all sightings occurred along the continental shelf break (McLellan et al., 2015). These results suggest some degree of residency for beaked whales in this area (McLellan et al., 2015). Median water depths at tagging locations ranged from 1,725 to 2,274 m, with a maximum water depth of 3,015 m. Diving data captured by the tags showed a maximum dive depth of 2,800 m suggesting that many of the dives were likely to, or close to, the seafloor (McLellan et al., 2015).

Passive acoustic monitoring conducted between 2007 and 2013 in Onslow Bay, North Carolina resulted in detections of multiple beaked whale vocalization events. Beaked whale detections were documented throughout the monitoring period with no specific diel pattern (Hodge & Read, 2015). Gervais' beaked whales were detected significantly more than any other beaked whale species. Goose-beaked whale clicks were detected in November 2012 and Blainville's beaked whale clicks were detected primarily in April and May 2013 (Hodge & Read, 2015). True's and Sowerby's beaked whales were not detected during this effort, but there were two detections in December 2012 of a click type assigned to an unidentified beaked whale species. Passive acoustic monitoring conducted offshore of Cape Hatteras between March and April 2012 recorded beaked whale clicks on nearly 40 percent of the recording days (Stanistreet et al., 2013). Closer examination of these beaked whale click events suggested they belonged to goose-beaked and Gervais' beaked whales (Stanistreet et al., 2012).

MacLeod and Mitchell (2006) described the northern Gulf of Mexico continental shelf margin as "a key area" for beaked whales. Beaked whales were seen in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) (Hansen et al., 1996; Mullin & Hoggard, 2000). Some of the aerial survey sightings may have included goose-beaked whale, although identification of beaked whale species from aerial surveys is problematic. Beaked whale sightings made during spring and summer vessel surveys were widely distributed in waters greater than 500 m deep. Between 2000 and 2021, vessel surveys in the Gulf of Mexico documented 14 sightings of beaked whales (1 Gervais and 13 goose-beaked) (Halpin et al., 2009). While these survey data include large temporal gaps, they indicate a regular and recurring presence of goose-beaked whales in the Gulf of Mexico.

F.5.3.5.3 Population Trends

A trend analysis has not been conducted for the western North Atlantic goose-beaked whale stock. Additionally, trend analyses have not been conducted for any of the four species of *Mesoplodon* in the western North Atlantic (Waring et al., 2014).

A trend analysis for goose-beaked whale was conducted for data from 2003 to 2018, but the statistical power of this analysis is limited due to the available data (Hayes et al., 2021). Further analysis and additional data are required to determine a true change in abundance versus a distributional shift across the Gulf of Mexico (Hayes et al., 2020). There are insufficient data to determine population trends for Blainville's and Gervais' beaked whales in the northern Gulf of Mexico.

F.5.3.5.4 Predator and Prey Interactions

Beaked whales are generally deep-water feeders and prey on both squid and fish. Examination of stomach contents from stranded *Mesoplodon* indicates that they feed primarily on deep-water cephalopods (MacLeod et al., 2003). Stomach content analyses of captured and stranded *Mesoplodon* suggest that beaked whales are deep divers that feed at or close to the bottom in deep oceanic waters, taking whatever suitable prey they encounter or feeding on whatever species are locally abundant (Ohizumi, 2002). Stomach content analyses from goose-beaked whales show that they feed mostly on deep-sea squid, fish, and crustaceans (Hickmott, 2005; Santos et al., 2007). Data show that goose-beaked whales use suction to ingest prey (Jefferson et al., 2008; Jefferson et al., 2015; Werth, 2006).

F.5.3.5.5 Species-Specific Threats

Impacts from anthropogenic noise have become a serious concern with regard to beaked whales over the past decade. In addition, disturbance by anthropogenic noise may prove to be an important habitat issue in some areas of beaked whales' range, notably in areas of concentrated military activity, oil and gas activity, or shipping. Ongoing studies are being conducted to address this issue and its impact, if any, on this and other marine species.

Gulf of Mexico stocks of Blainville's, goose-beaked, and Gervais' beaked whales were among the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 12 percent of these beaked whale species in the Gulf of Mexico were exposed to oil, resulting in 4 percent excess mortality above baseline conditions, 5 percent excess failed pregnancies, and a 4 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the Gulf of Mexico Blainville's, goose-beaked, and Gervais' beaked whale stocks will take an estimated 10 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.6 Northern Bottlenose Whale (*Hyperoodon ampullatus*)

F.5.3.6.1 Status and Management

There are two populations of northern bottlenose whales in the western North Atlantic: one on the Scotian Shelf in the area referred to as the Gully and a second in Davis Strait off northern Labrador. The Gully is a unique ecosystem that appears to have long provided a stable year-round habitat for a distinct population of bottlenose whales (Dalebout et al., 2006). The Scotian Shelf population of northern bottlenose whales is listed as endangered by the Committee on the Status of Endangered Wildlife in Canada and the Davis Strait-Baffin Bay-Labrador Sea population is designated as a population of special concern (Committee on the Status of Endangered Wildlife in Canada, 2011).

F.5.3.6.2 Habitat and Geographic Range

Northern bottlenose whales are largely a deep-water species and seldom found in waters less than 2,000 m deep (Mead, 1989b). Distribution is concentrated in areas of high relief, including shelf breaks and submarine canyons.

Northern bottlenose whales are commonly found in the Labrador Current and likely occur in the Gulf Stream open-ocean areas. The Gully straddles the Scotian Shelf and Gulf Stream areas.

Northern bottlenose whales are distributed in the North Atlantic primarily from Nova Scotia to about 70° in the Davis Strait, along the east coast of Greenland to 77°, and from England to the west coast of Spitzbergen (Waring et al., 2015). There are two main centers of bottlenose whale distribution in the western North Atlantic: the Scotian Shelf (including the Gully), and Davis Strait off northern Labrador (Reeves et al., 1993). Genetic studies have shown that these two populations are likely distinct from one another (Dalebout et al., 2006). Northern bottlenose whales have been sighted in deep waters off New England but are uncommon in U.S. waters. Strandings have occurred as far south as North Carolina, although that is outside of the natural range or at the edge of the southern range for this more subarctic species (Jefferson et al., 2008; Jefferson et al., 2015; MacLeod et al., 2006).

F.5.3.6.3 Population Trends

There is insufficient data to determine the population trends for this species (Waring et al., 2015).

F.5.3.6.4 Predator and Prey Interactions

This species preys primarily on squid of the genus *Gonatus* but will also take fishes, sea cucumbers, sea stars, and prawns, as confirmed by stomach content analyses (Clarke & Kristensen, 1980; Gowans, 2009). They appear to be more benthic (bottom of the sea) feeders, foraging at depths of between 500 and 1,500 m (Hooker & Whitehead, 2002; Jefferson et al., 2015).

F.5.3.6.5 Species-Specific Threats

There are no significant species-specific threats to northern bottlenose whales in the northwest Atlantic.

F.5.3.7 Atlantic Spotted Dolphin (*Stenella frontalis*)

F.5.3.7.1 Status and Management

The Atlantic spotted dolphin occurs in two forms that may be distinct subspecies (Perrin, 2009a; Perrin et al., 1987; Rice, 1998): the large, heavily spotted form, which inhabits the continental shelf and is usually found inside or near the 200-m isobath; and the smaller, less spotted island and offshore form, which occurs in the Atlantic Ocean but is not known to occur in the Gulf of Mexico (Fulling et al., 2003; Mullin & Fulling, 2003, 2004). The western North Atlantic population is considered a separate stock from the Gulf of Mexico stock(s) for management purposes based on genetic analysis (Waring et al., 2014; Waring et al., 2016). The Puerto Rico and U.S. Virgin Islands population is also considered a separate stock, although there is currently no information to differentiate this stock from the Atlantic Ocean and Gulf of Mexico stocks.

F.5.3.7.2 Habitat and Geographic Range

The Atlantic spotted dolphin is found in tropical to warm-temperate waters, predominantly over the continental shelf and upper slope (Waring et al., 2013, 2014). In the eastern Gulf of Mexico, for instance, the species often occurs over the mid-shelf (Griffin & Griffin, 2003). In the western Atlantic, this species is distributed from New England to Brazil and is found in the Gulf of Mexico as well as the Caribbean Sea (Perrin, 2009a). Atlantic spotted dolphins may occur in the Gulf Stream open-ocean area.

The large, heavily spotted coastal form of the Atlantic spotted dolphin typically occurs over the continental shelf but is often at least several miles offshore (Davis et al., 1998; Perrin, 2002, 2009a). Atlantic spotted dolphin sightings have been concentrated in the slope waters north of Cape Hatteras, but in the shelf waters south of Cape Hatteras, sightings extend into the deeper slope and offshore waters of the mid-Atlantic (Mullin & Fulling, 2003; Waring et al., 2014). Vessel surveys conducted between January 2009 and December 2014 offshore of Cape Hatteras, North Carolina resulted in multiple sightings of Atlantic spotted dolphins annually (Foley et al., 2015). Vessel surveys conducted from 2016 to 2021 offshore of Virginia Beach sighted this species a total of 36 times, with 2 to 12 sightings annually (Ocean Biodiversity Information System – Spatial Ecological Analysis of Megavertebrate Populations, 2024). Aerial surveys in the Norfolk Canyon area detected Atlantic spotted dolphins between April and October of 2016 through 2019, with sightings of both inshore and offshore ecotypes (Cotter, 2019). Aerial and shipboard surveys conducted between 2007 and 2010 in offshore waters of Onslow Bay, North Carolina, indicate that spotted dolphins have a strong preference for waters over the continental shelf and do not typically occur beyond the shelf break (Read et al., 2014). Numerous re-sightings of multiple individuals over several years and across seasons supports the existence of considerable fine-scale population structure and a degree of residency for Atlantic spotted dolphins in Onslow Bay (Swaim et al., 2014).

Photo-identification catalogs of Atlantic spotted dolphins from Cape Hatteras, Onslow Bay, and Jacksonville survey areas have been compared, but no matches have been identified (Foley et al., 2015; Swaim et al., 2014) suggesting a high degree of residency to these areas. Atlantic spotted dolphins were one of the dominant species sighted during vessel surveys conducted along the continental shelf break and pelagic waters offshore of Jacksonville, Florida from July 2009 through December 2013 (Swaim et al., 2014). Sightings were restricted to the relatively shallow shelf waters of the survey area.

Higher numbers of spotted dolphins are reported over the west Florida continental shelf from November to May than during the rest of the year, suggesting that this species may migrate seasonally (Griffin & Griffin, 2003).

In the Gulf of Mexico, Atlantic spotted dolphins occur primarily from continental shelf waters 10-200 m deep to slope waters greater than 500 m deep (Fulling et al., 2003; Maze-Foley & Mullin, 2006; Mullin & Fulling, 2004). Atlantic spotted dolphins were seen in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico from 1992 to 1998 (Hansen et al., 1996; Mullin & Hoggard, 2000).

F.5.3.7.3 Population Trends

A trend analysis has been conducted for the North Atlantic stock of Atlantic spotted dolphins, using data from surveys in 2004, 2011, and 2016 (Hayes et al., 2020). A significant decrease in population size was detected; however, the analysts noted uncertainty in whether interannual abundance changes are related to the population size or changes in spatial distribution due to environmental variation.

There are insufficient data to determine population trends for the Northern Gulf of Mexico stock of Atlantic spotted dolphins (Waring et al., 2013) and for the Puerto Rico and U.S. Virgin Islands stock of Atlantic spotted dolphins (Waring et al., 2012).

F.5.3.7.4 Predator and Prey Interactions

Atlantic spotted dolphins feed on small cephalopods, fishes, and benthic invertebrates. Atlantic spotted dolphins in the Gulf of Mexico were observed feeding cooperatively on clupeid fishes and are known to feed in association with shrimp trawlers (Fertl & Leatherwood, 1997; Fertl & Würsig, 1995). In the

Bahamas, this species was observed to chase and catch flying fish (MacLeod et al., 2004). The diet of the Atlantic spotted dolphin varies depending on its location (Jefferson et al., 2015). This species was documented to be prey for killer whales and sharks (Jefferson et al., 2015).

F.5.3.7.5 Species-Specific Threats

The northern Gulf of Mexico stock of Atlantic spotted dolphins was included as 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification was completed for continental shelf dolphins, which entailed a combination of shelf bottlenose dolphins and Atlantic spotted dolphins. It was determined that 13 percent of continental shelf dolphins, including Atlantic spotted dolphins in the Gulf of Mexico were exposed to oil, resulting in 4 percent excess mortality above baseline conditions, 6 percent excess failed pregnancies, and 5 percent higher likelihood for other adverse health effects. The maximum reduction of combined Atlantic spotted dolphins and bottlenose dolphins was only 3 percent; therefore, it was not possible to calculate the number of years it would take for these stocks to recover (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.8 Atlantic White-Sided Dolphin (*Lagenorhynchus acutus*)

F.5.3.8.1 Status and Management

Three population units of Atlantic white-sided dolphins in the western North Atlantic Ocean are suggested for conservation management of this stock: Gulf of Maine, Gulf of St. Lawrence, and Labrador Sea (Palka et al., 1997; Waring et al., 2004). Evidence for stock differentiation between the Gulf of Maine and Gulf of St. Lawrence comes from reduced summer sightings along the eastern side of Nova Scotia (Hayes et al., 2020). No genetic analysis has been done to confirm this separation. The species is considered abundant in the North Atlantic (Jefferson et al., 2008; Waring et al., 2013).

A proposed taxonomic revision for this species is in progress (Hayes et al., 2020; Vollmer et al., 2019). However, until the new classification is officially accepted, the current species names will be used.

F.5.3.8.2 Habitat and Geographic Range

This species is found primarily in cold temperate to subpolar continental shelf waters to the 328 ft. (100 m) depth contour (Cetacean and Turtle Assessment Program, 1982; Mate et al., 1994; Selzer & Payne, 1988). Occurrence of Atlantic white-sided dolphins in the northeastern United States probably reflects fluctuations in food availability as well as oceanographic conditions (Palka et al., 1997; Selzer & Payne, 1988). Before the 1970s, Atlantic white-sided dolphins were found primarily offshore in waters over the continental slope; however, since then, they occur primarily in waters over the continental shelf, replacing white-beaked dolphins, which were previously sighted in the area. This shift may have been the result of an increase in sand lance and a decline in herring in continental shelf waters (Payne et al., 1990). Areas of feeding importance are around Cape Cod and on the northwest edge of Georges Bank, in an area defined as the Great South Channel-Jeffreys Ledge corridor (Cetacean and Turtle Assessment Program, 1982; Palka et al., 1997). Selzer and Payne (1988) sighted white-sided dolphins more frequently in areas of high seafloor relief and where sea surface temperatures and salinities were low, although these environmental conditions might be only secondarily influencing dolphin distribution; seasonal variation in sea surface temperature and salinity as well as local nutrient upwelling in areas of high seafloor relief may affect preferred prey abundances, which in turn might affect dolphin distribution (Selzer & Payne, 1988).

Atlantic white-sided dolphins would be expected to occur in the Labrador Current and possibly in the northern extent of the Gulf Stream open-ocean area. Atlantic white-sided dolphins are common in waters of the continental slope from New England to southern Greenland (Cipriano, 2009; Jefferson et al., 2008; Jefferson et al., 2015). Along the Canadian and U.S. Atlantic coast, this species is most common from Hudson Canyon north to the Gulf of Maine (Palka et al., 1997). From January to May, low numbers of white-sided dolphins may be found from Georges Bank to Jeffreys Ledge. Even lower numbers are found south of Georges Bank (Palka et al., 1997; Payne et al., 1990; Waring et al., 2004). From June through September, large numbers of white-sided dolphins are found from Georges Bank to the lower Bay of Fundy (Payne et al., 1990; Waring et al., 2004). During this time, strandings occur from New Brunswick to New York (Palka et al., 1997). From October to December, white-sided dolphins occur at intermediate densities from southern Georges Bank to the southern Gulf of Maine. Sightings occur year-round south of Georges Bank, particularly around Hudson Canyon, but in low densities (Cetacean and Turtle Assessment Program, 1982; Palka, 1997; Payne et al., 1990; Waring et al., 2004). A few strandings were collected on Virginia and North Carolina beaches, which appear to represent the southern edge of the range for this species (Cipriano, 2009).

F.5.3.8.3 Population Trends

A trend analysis has not been conducted for the western North Atlantic stock of Atlantic white-sided dolphins (Waring et al., 2015).

F.5.3.8.4 Predator and Prey Interactions

The stomach contents of Atlantic white-sided dolphins caught through fishing bycatch, as well as those stranded off the coast of New England, have included at least 26 fish species and 3 cephalopod species. The most prominent species were the silver hake, spoonarm octopus, and haddock. Sand lances were found in the stomach of one stranded white-sided dolphin (Hayes et al., 2018). There is seasonal variation in the diet; Atlantic herring was found in more dolphins during the summer than in winter (Craddock et al., 2009). This species is known to feed in association with other delphinid (dolphin-like) and large whale species (Jefferson et al., 2015; Palka et al., 1997).

This species was not documented to be prey for any other species (Jefferson et al., 2015).

F.5.3.8.5 Species-Specific Threats

This species is susceptible to mass strandings, as well as fishery-related mortality.

F.5.3.9 Clymene Dolphin (*Stenella clymene*)

F.5.3.9.1 Status and Management

The Clymene dolphin has an extensive range in the tropical Atlantic Ocean. The western North Atlantic and Northern Gulf of Mexico populations are managed as separate stocks.

F.5.3.9.2 Habitat and Geographic Range

Clymene dolphins are a tropical to subtropical species, primarily sighted in deep waters well beyond the edge of the continental shelf (Fertl et al., 2003). Clymene dolphins likely occur in the Gulf Stream open-ocean area.

In the western North Atlantic, Clymene dolphins were observed as far north as New Jersey, although sightings were primarily in offshore waters east of Cape Hatteras over the continental slope and are likely to be strongly influenced by oceanographic features of the Gulf Stream (Fertl et al., 2003; Moreno et al., 2005; Mullin & Fulling, 2003). Vessel and aerial surveys conducted offshore of Cape Hatteras from 2011 through 2017 have resulted in 18 Clymene dolphin sightings during summer and fall, including one sighting of Clymene dolphins in a mixed group of spinner dolphins within the northern offshore waters of the survey area in 2011 (U.S. Department of the Navy, 2013a). Vessel-based surveys offshore of Virginia Beach between 2016 and 2021 did not identify any sightings of this species (Ocean Biodiversity Information System – Spatial Ecological Analysis of Megavertebrate Populations, 2024).

Clymene dolphins in the Gulf of Mexico are observed most frequently on the lower slope and deep-water areas, primarily west of the Mississippi River, in regions of cyclonic or confluent circulation (Davis et al., 2002; Mullin et al., 1994a). Clymene dolphins were seen in the winter, spring, and summer during GulfCet aerial surveys of the northern Gulf of Mexico during 1992 to 1998 (Hansen et al., 1996; Mullin & Hoggard, 2000).

F.5.3.9.3 Population Trends

There are insufficient data to determine population trends for the western North Atlantic stock of Clymene dolphins (Waring et al., 2013, 2014). In the Gulf of Mexico, a trend analysis documented significant differences between abundance estimates in 2004, 2009, and 2017 (Hayes et al., 2021). However, the statistical power in this analysis is low due to lack of annual survey data, and it is not possible to determine whether the results indicate a change in abundance versus a change in distribution of the animals throughout the Gulf of Mexico (Hayes et al., 2021).

F.5.3.9.4 Predator and Prey Interactions

Available information on feeding habits is very limited. This species preys on small fish and squid at moderate depths and feeds primarily at night (Fertl et al., 1997; Jefferson et al., 2015; Perrin et al., 1981).

This species is possibly preyed on by killer whales and large sharks, as evidenced by scars observed on their bodies, although actual predation was not observed (Jefferson, 2009a; Jefferson et al., 2008; Jefferson et al., 2015).

F.5.3.9.5 Species-Specific Threats

The northern Gulf of Mexico stock of Clymene dolphins was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 7 percent of Clymene dolphins in the Gulf of Mexico were exposed to oil, resulting in 2 percent excess mortality above baseline conditions, 3 percent excess failed pregnancies, and 3 percent higher likelihood for other adverse health effects. The maximum reduction of the Clymene dolphin population was only 3 percent; therefore, the trustees were not able to calculate the number of years it would take for this stock to recover (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.10 Common Bottlenose Dolphin (*Tursiops truncatus*)

F.5.3.10.1 Status and Management

There are currently 53 management stocks identified by NMFS in the western North Atlantic and Gulf of Mexico, including oceanic, coastal, and estuarine stocks (Waring et al., 2016). Most stocks in the Study Area are designated as strategic or depleted under the MMPA. For a complete listing of currently identified stocks within the Study Area, see Table 3.7-1 (Marine Mammal Occurrence within the Study Area).

F.5.3.10.2 Habitat and Geographic Range

The bottlenose dolphin occurs in tropical to temperate waters of the Atlantic Ocean as well as inshore, nearshore, and offshore waters of the Gulf of Mexico and U.S. East Coast (Waring et al., 2016). They generally do not range north or south of 45° latitude (Jefferson et al., 2008; Jefferson et al., 2015; Wells & Scott, 2009). They occur in most enclosed or semi-enclosed seas in habitats ranging from shallow, murky, estuarine waters to deep, clear offshore waters in oceanic regions (Jefferson et al., 2008; Jefferson et al., 2015; Wells et al., 2009). Open-ocean populations occur far from land; however, population density appears to be highest in nearshore areas (Scott & Chivers, 1990). Bottlenose dolphins occur in the North Atlantic Gyre and Gulf Stream open-ocean areas.

There are two morphologically and genetically distinct bottlenose dolphin morphotypes (distinguished by physical differences) (Duffield et al., 1983) described as coastal and offshore forms. In a decade-long collaborative study using DNA and morphological data, it has recently been proposed that the coastal form is a separate species than their offshore counterparts, and are more closely related to coastal populations from the Gulf of Mexico and Caribbean. While a definitive distinction as a separate species has yet to be codified, the coastal form are currently being referred to as a Tamanend's bottlenose dolphin (*Tursiops erebennus*) (Costa et al., 2022). Both inhabit waters in the western North Atlantic Ocean and Gulf of Mexico (Curry & Smith, 1997; Hersh & Duffield, 1990; Mead & Potter, 1995) along the U.S. Atlantic coast. The coastal morphotype of bottlenose dolphin is continuously distributed along the Atlantic coast south of Long Island, New York, around the Florida peninsula, and along the Gulf of Mexico coast. The range of the offshore bottlenose dolphin includes waters beyond the continental slope (Kenney, 1990), and offshore bottlenose dolphins may transit between the Gulf of Mexico and the Atlantic (Wells et al., 1999). Dolphins with characteristics of the offshore ecotype have stranded as far south as the Florida Keys.

In Canadian waters, bottlenose dolphins were occasionally sighted on the Scotian Shelf, particularly in the Gully (Gowans & Whitehead, 1995). Seasonally, bottlenose dolphins occur over the outer continental shelf and inner slope as far north as Georges Bank (Cetacean and Turtle Assessment Program, 1982; Kenney, 1990). Sightings occurred along the continental shelf break from Georges Bank to Cape Hatteras during spring and summer (Cetacean and Turtle Assessment Program, 1982; Kenney, 1990).

Several lines of evidence support a distinction between coastal stock dolphins and those present primarily in the inshore waters of the bays, sounds, and estuaries (LaBrecque et al., 2015b). Photo-identification and genetic studies support the existence of more than 40 stock populations in bays, sounds, and estuaries. These populations inhabit estuaries and bays from North Carolina to the Gulf of Mexico coast (Caldwell, 2001; Gubbins, 2002; Gubbins et al., 2003; Litz, 2007; Mazzoil et al., 2005; Zolman, 2002).

LaBrecque et al. (2015a) identified nine small and resident bottlenose dolphin population ranges within estuarine areas along the U.S. east coast (Figure F.5-9 and Figure F.5-10). These ranges include estuarine and nearshore areas extending from Pamlico Sound, North Carolina down to Florida Bay, Florida and were substantiated through vessel- and aerial-based survey data, photo-identification data, genetic analyses, and expert judgment (LaBrecque et al., 2015a). The Northern North Carolina, Southern North Carolina, and Charleston Harbor Populations partially overlap with nearshore portions of the Navy Cherry Point Range Complex, while the Jacksonville Estuarine System Populations partially overlap with nearshore portions of the Jacksonville Range Complex. The Southern Georgia Estuarine System Population also overlap with the Jacksonville Range Complex, specifically within Naval Submarine Base Kings Bay, Georgia and includes estuarine and intracoastal waterways from Altamaha Sound to the Cumberland River (LaBrecque et al., 2015a). The remaining four biologically important areas are outside but adjacent to the Study Area boundaries.

Off the coast of Virginia within the Study Area, three stocks of bottlenose dolphins are common in the waters of Chesapeake Bay and along the state's coastline. Engelhaupt et al. (2022) established dolphin density was highest inshore during the warmer months from June to November, although more than 200 individuals remained present during the winter and spring months, which had not been previously considered or reported (Barco et al., 1999; Cetacean and Turtle Assessment Program, 1982). Photo identification results indicate bottlenose dolphin presence in this area consist of short-term visits, with 82.9 percent of individuals sighted only once, while re-sightings in the Cape Henry region show clear indication of localized site fidelity with overlapping ranges of individual stocks (Engelhaupt et al., 2022).

In the Gulf of Mexico alone, 32 distinct stocks are recognized, although the structure of these stocks is uncertain but appears to be complex. Residency patterns of dolphins in bays, sounds, and estuaries range from transient, seasonally migratory, and stable resident communities, where various stocks may overlap at times. Year-round residency patterns of some individual bottlenose dolphins in bays, sounds, and estuaries have been reported for almost every survey area where photo-identification or tagging studies have been conducted.

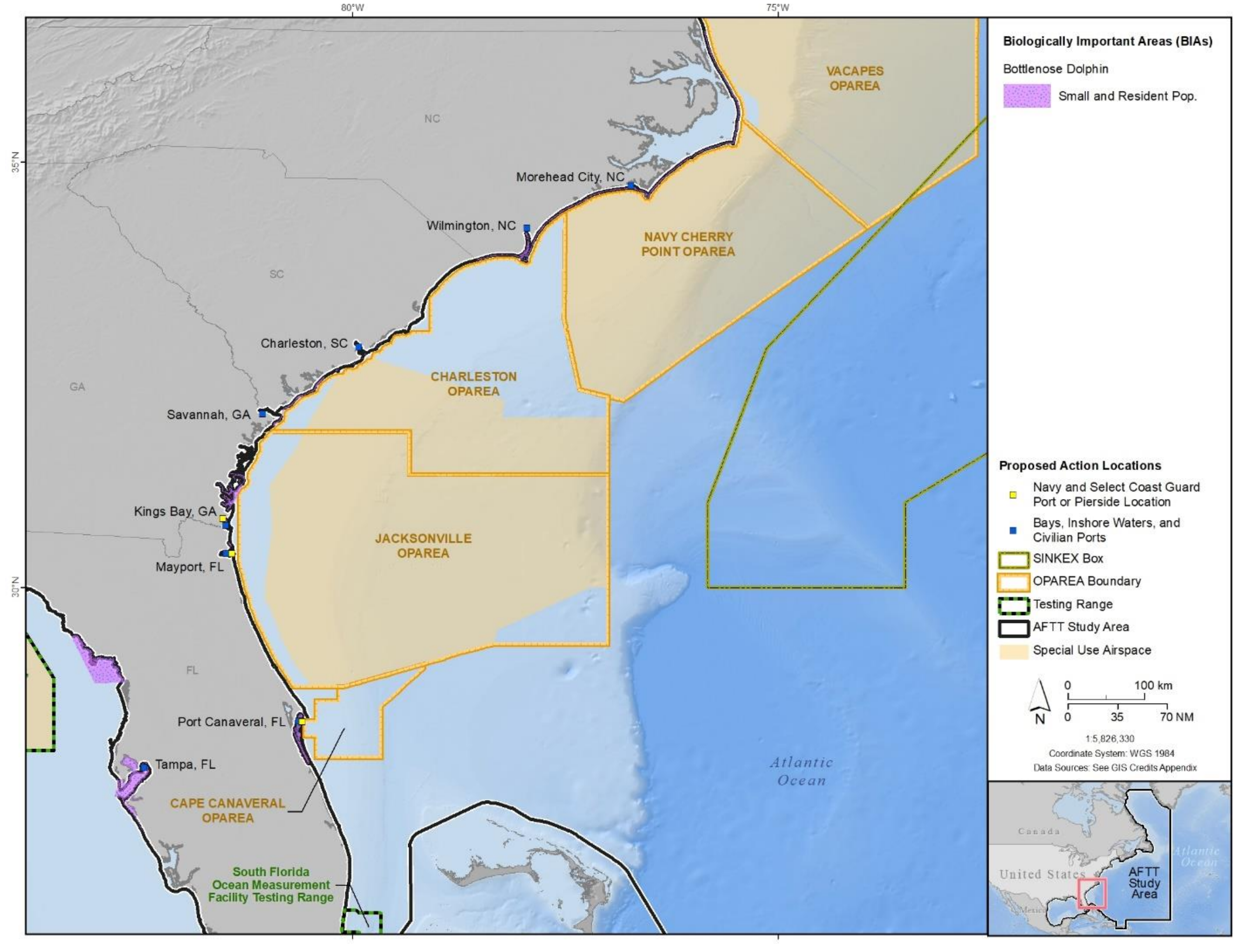


Figure F.5-9: Biologically Important Areas for Bottlenose Dolphins in the Study Area – Southeast

LaBrecque et al. (2015b) delineated 11 small and resident population areas for bottlenose dolphins within the Gulf of Mexico (Figure F.5-11). These areas include bays, sounds, and estuaries ranging from Aransas Pass, Texas to the Florida Keys, Florida and were substantiated through a combination of extensive photo-identification data, genetic analyses, radio-tracking data, and expert knowledge (LaBrecque et al., 2015b). Of the 11 biologically important areas identified for bottlenose dolphins in the Gulf of Mexico, three overlap with the Gulf of Mexico Range Complex (Aransas Pass Area, Texas; Mississippi Sound Area, Mississippi; and St. Joseph Bay Area, Florida) and eight are located adjacent to the Study Area boundaries.

F.5.3.10.3 Population Trends

Trend analyses have been conducted for the Western North Atlantic Northern and Southern Migratory Coastal stocks. While power is limited to detect a trend in either stock separately, a combined analysis indicates a potential decline in population over the last two decades (Hayes et al., 2021).

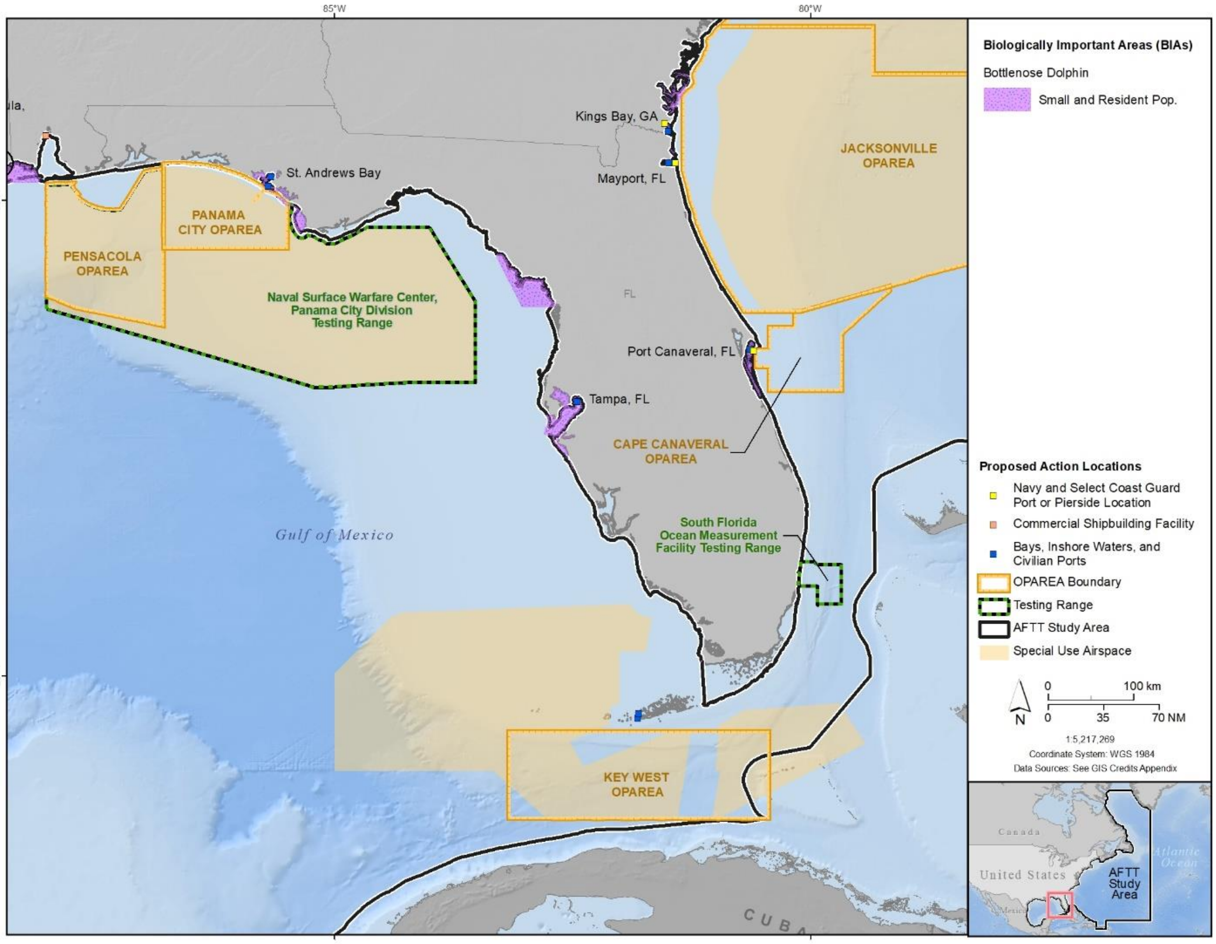


Figure F.5-10: Biologically Important Areas for Bottlenose Dolphns in the Study Area – South Florida and Gulf of Mexico

A trend analysis has not been conducted for the following stocks of bottlenose dolphins: Northern North Carolina Estuarine System stock; Southern North Carolina Estuarine System stock; Western North Atlantic Offshore stock and Northern Gulf of Mexico Oceanic (Waring et al., 2015).

There are insufficient data to determine the population trends for the following stocks of bottlenose dolphins: Northern Gulf of Mexico Continental Shelf stock; Northern South Carolina Estuarine System stock; Charleston Estuarine System stock; Northern Georgia/Southern South Carolina Estuarine System stock; Central Georgia Estuarine System stock; Southern Georgia Estuarine System stock; Jacksonville Estuarine System stock; Indian River Lagoon Estuarine System stock; Biscayne Bay stock; Florida Bay stock; Gulf of Mexico Eastern Coastal stock; Gulf of Mexico Northern Coastal stock; Gulf of Mexico Western Coastal stock; most of the Northern Gulf of Mexico Bay, Sound, and Estuary stocks; Barataria Bay Estuarine System stock; Mississippi Sound stock; Lake Borgne Bay Boudreau stock; St. Joseph Bay stock; Choctawhatchee Bay stock; and Puerto Rico and U.S. Virgin Islands stock (Waring et al., 2012; Waring et al., 2015).

There are limited data available to assess population trends for the following stocks of bottlenose dolphins: Western North Atlantic South Carolina-Georgia Coastal stock, Western North Atlantic

Northern Florida Coastal stock, and Western North Atlantic Central Florida Coastal stock (Waring et al., 2013, 2014).

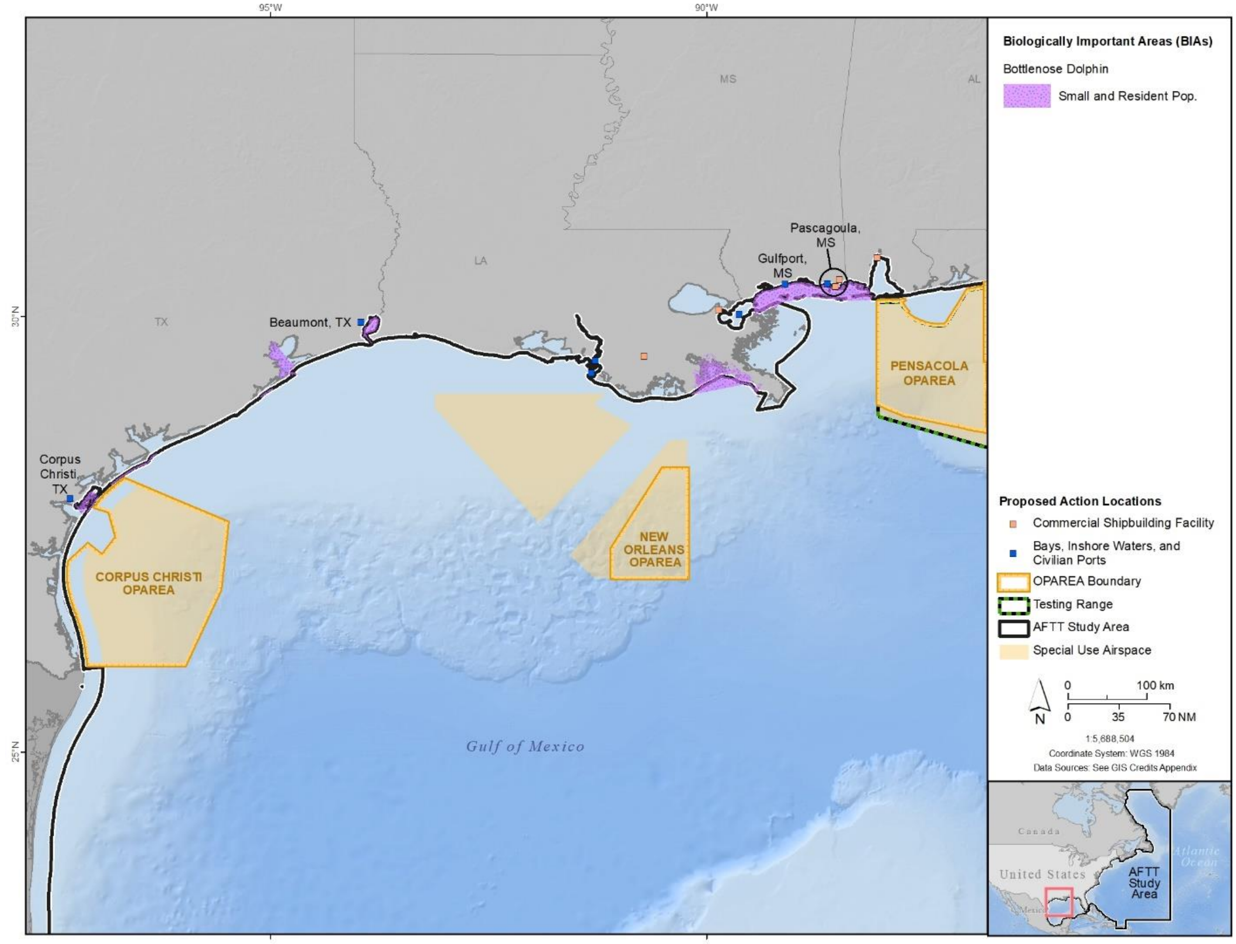


Figure F.5-11: Biologically Important Areas for Bottlenose Dolphins in the Study Area – Gulf of Mexico

F.5.3.10.4 Predator and Prey Interactions

Bottlenose dolphins are opportunistic feeders, taking a variety of fishes, cephalopods, and crustaceans (Wells & Scott, 1999) and using a variety of feeding strategies (Barros & Myrberg, 1987; Barros & Wells, 1998; Shane et al., 1986). Nearshore bottlenose dolphins prey predominantly on coastal fishes and cephalopods, while offshore individuals prey on open-ocean cephalopods and a large variety of near-surface and mid-water fishes (Mead & Potter, 1995).

This species is known to be preyed on by killer whales and sharks (Wells & Scott, 1999). As many as half of the observed bottlenose dolphins in Florida exhibit scars from shark attacks. Primary shark predators are considered to be the bull, tiger, great white, and dusky sharks (Wells & Scott, 1999).

F.5.3.10.5 Species-Specific Threats

Thirteen stocks of bottlenose dolphins in the Gulf of Mexico occur within the footprint of the 2010 *Deepwater Horizon* oil spill. In response to the oil spill, the Deepwater Horizon Natural Resource Damage Assessment Trustees prepared a *Final Programmatic Damage Assessment and Restoration Plan* *and Final Programmatic Environmental Impact Statement* (2016) to present the impacts and injuries sustained by habitats and species within the footprint. The findings from this report are summarized here. Injuries were quantified for four bay, sound, and estuary stocks of bottlenose dolphins: Barataria Bay, Mississippi River Delta, Mississippi Sound, and Mobile Bay. Perdido Bay, Pensacola Bay, Choctawhatchee Bay, and St. Andrew Bay stocks did not show evidence of excess strandings attributed to the oil spill; therefore, they were not included in the injury quantification.

The trustees also quantified injuries for two coastal stocks (Gulf of Mexico Western Coastal stock and the Gulf of Mexico Northern Coastal stock) and the northern Gulf of Mexico Oceanic stock. The northern Gulf of Mexico Continental Shelf stock of bottlenose dolphins was combined with continental shelf Atlantic spotted dolphins in a single continental shelf dolphin category for the injury quantification. In the report, excess mortality was calculated by comparing expected annual mortality rates for each stock based on historical stranding records and annual mortality rates calculated after the oil spill. By this method, excess mortality is considered mortalities attributable to the oil spill. The trustees estimated the Mississippi River Delta stock to have the highest percentage of excess mortality (59 percent), followed by Gulf of Mexico Northern Coastal stock (38 percent), Barataria Bay stock (35 percent), Mississippi Sound stock (22 percent), Mobile Bay stock (12 percent), continental shelf dolphins (including the northern Gulf of Mexico Continental Shelf stock) (4 percent), northern Gulf of Mexico Oceanic stock (3 percent), and Gulf of Mexico Western Coastal stock (1 percent) (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016). It is estimated that the population declined by 45 percent relative to baseline and will take 35 years to recover to 95 percent of baseline numbers (Schwacke et al., 2022). This could be due to a number of overall health impacts, including an increased rate of cardiac abnormalities in dolphins within the oil spill footprint (Linnehan et al., 2021).

An unusual mortality event was active from 2013 to 2015 along the mid-Atlantic coast of the United States (National Marine Fisheries Service, 2024a). During this timeframe, 1,614 mortalities were recorded from New York to Florida due to cetacean morbillivirus infections (92 percent of dolphins tested), or *Brucella* sp. bacterial infections (25 percent of dolphins tested.)

F.5.3.11 Common Dolphin (*Delphinus delphis*)

F.5.3.11.1 Status and Management

One stock of the short-beaked common dolphin (*D. delphis delphis*) is found within the Study Area: the western North Atlantic stock (Jefferson et al., 2009; Waring et al., 2013).

F.5.3.11.2 Habitat and Geographic Range

In the North Atlantic, common dolphins occur over the continental shelf along the 100- to 2,000-meter isobaths and over prominent underwater topography and east to the mid-Atlantic Ridge (29° West) (Doksaeter et al., 2008; Waring et al., 2008). There is a well-studied population of short-beaked common dolphins in the western North Atlantic associated with the Gulf Stream (Jefferson et al., 2009). It occurs mainly in offshore waters, ranging from Canada maritime provinces to the Florida/Georgia border (Waring et al., 2010).

In waters off the northeastern U.S. coast, common dolphins are distributed along the continental slope and are associated with Gulf Stream features (Cetacean and Turtle Assessment Program, 1982; Hamazaki, 2002; Selzer & Payne, 1988). They primarily occur from Cape Hatteras northeast to Georges Bank (35° to 42° North) during mid-January to May (Cetacean and Turtle Assessment Program, 1982; Hain et al., 1981). Common dolphins move onto Georges Bank and the Scotian Shelf from mid-summer to autumn. Selzer and Payne (1988) reported very large aggregations (greater than 3,000 animals) on

Georges Bank in autumn. Common dolphins are occasionally found in the Gulf of Maine (Selzer & Payne, 1988). Migration onto the Scotian Shelf and continental shelf off Newfoundland occurs during summer and autumn when water temperatures exceed 11° C (Gowans & Whitehead, 1995). The species is less common south of Cape Hatteras, although schools were reported as far south as the Georgia/South Carolina border (32° North) (Jefferson et al., 2009).

A single location-only satellite telemetry tag was deployed on a short-beaked common dolphin offshore of Cape Hatteras in June 2014, and location data were obtained over a 40-day period. The individual remained primarily over the continental shelf break and continental slope, and traveled north away from the tagging location to shallower continental shelf waters off New England during the mid-summer (Baird et al., 2015). The median depth of tagged animal locations over the 40-day span was 297 m (Baird et al., 2015).

Vessel-based surveys offshore of Virginia Beach between 2016 and 2021 detected common dolphins frequently; as many as 46 encounters were reported annually (Ocean Biodiversity Information System – Spatial Ecological Analysis of Megavertebrate Populations, 2024). This was the third-most common species encountered during these surveys, with 154 total sightings over the survey period. Aerial surveys of the Norfolk Canyon area detected common dolphins frequently between 2016 and 2019 (Cotter, 2019), with sightings of large groups (greater than 500 individuals) in waters beyond the shelf break.

F.5.3.11.3 Population Trends

A trend analysis has not been conducted for the western North Atlantic stock of common dolphins (Waring et al., 2016).

F.5.3.11.4 Predator and Prey Interactions

Common dolphins feed primarily on organisms in the vertically migrating deep scattering layer, including fish and squid (dos Santos & Haimovici, 2001; Meynier et al., 2008; Overholtz & Waring, 1991; Pusineri et al., 2007). Diel (a 24-hour cycle that often involves a day and the adjoining night) fluctuations in vocal activity, with more vocal activity during late evening and early morning, appear to be linked to feeding in the deep scattering layer, which rises in this same time frame (Goold, 2000). In the western North Atlantic, oceanic dolphins feed more on squid than those in more nearshore waters (Perrin, 2009b).

Short-beaked common dolphins are known to be preyed on by killer whales (Visser, 1999) and large sharks (Leatherwood et al., 1973), although little is known about the impact of this predation on populations.

F.5.3.11.5 Species-Specific Threats

Common dolphins are susceptible to fishery interactions, such as gillnet entanglement or longline fisheries (Hamilton & Baker, 2019; Peltier et al., 2021).

F.5.3.12 False Killer Whale (*Pseudorca crassidens*)

F.5.3.12.1 Status and Management

Little is known of the status of most false killer whale populations around the world. While the species is not considered rare, few areas of high density are known. The population found in the Gulf of Mexico is considered a separate stock from the western North Atlantic stock for management purposes; however, there are no genetic data to differentiate between the two stocks (Waring et al., 2013).

F.5.3.12.2 Habitat and Geographic Range

False killer whales occur worldwide throughout warm temperate and tropical oceans. They are found in deep open-ocean waters and around oceanic islands, and only rarely come into shallow coastal waters

(Baird et al., 2008; Leatherwood & Reeves, 1983; Odell & McClune, 1999). Occasional inshore movements are associated with movements of prey and shoreward flooding of warm ocean currents.

False killer whales have been sighted in U.S. Atlantic waters from southern Florida to Maine (Schmidly, 1981), with periodic records (primarily stranding) from southern Florida to Cape Hatteras dating back to 1920 (Schmidly, 1981). There are 28 records of false killer whale sightings in the western North Atlantic in OBIS-SEAMAP dating back to 1971, with group sizes ranging from 1 to 30 animals. Nine of these sightings occurred between 2000 and 2021 (Ocean Biodiversity Information System – Spatial Ecological Analysis of Megavertebrate Populations, 2024). One additional sighting of 11 animals occurred during a shipboard survey conducted in summer 2011 (Waring et al., 2016). Deployment of high-frequency acoustic recording packages offshore of Cape Hatteras, Onslow Bay, Jacksonville, and the offshore areas near Norfolk Canyon from 2009 through 2022 have resulted in zero false killer whale detections.

Sightings of this species in the northern Gulf of Mexico (i.e., U.S. Gulf of Mexico) occur in oceanic waters, primarily in the eastern Gulf (Maze-Foley & Mullin, 2006; Mullin & Fulling, 2004). False killer whales were seen only in the spring and summer during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al., 1996; Mullin & Hoggard, 2000) and in the spring during vessel surveys (Mullin et al., 2004). There are 17 records of false killer whale sightings in the Gulf of Mexico in OBIS-SEAMAP, dating back to 1987 (Halpin et al., 2009), with group sizes ranging from 3 to 70 individuals. Six of these sightings occurred between 2000 and 2021 (Halpin et al., 2009).

F.5.3.12.3 Population Trends

There are insufficient data to determine population trends for the western North Atlantic stock of false killer whales (Waring et al., 2016). While a trend analysis has been conducted for the northern Gulf of Mexico stock of false killer whales, the confidence is low due to imprecise abundance estimates and long intervals between surveys (Waring et al., 2013). Additionally, a Gulf-wide assessment of false killer whale abundance has not been completed (Waring et al., 2013).

F.5.3.12.4 Predator and Prey Interactions

False killer whales feed primarily on deep-sea cephalopods and fish (Odell & McClune, 1999). They may also prefer large fish species, such as mahi mahi and tuna. Twenty-five false killer whales that stranded off the coast of the Strait of Magellan (outside of the Study Area) were examined and found to feed primarily on cephalopods and fish. Squid beaks were found in nearly half of the stranded animals. The most important prey species were found to be squid, followed by Patagonian grenadier, a coastal species of fish (Koen-Alonso et al., 1999).

False killer whales have been observed attacking dolphins and large whales, such as humpback and sperm whales (Hooker et al., 2009). They are also known to behave aggressively toward small cetaceans during tuna purse seine fishing operations. Unlike other whales or dolphins, false killer whales frequently pass prey back and forth among individuals before consumption, in what appears to be a way of affirming social bonds (Baird et al., 2010). This species is believed to be preyed on by large sharks and killer whales (Baird, 2009a).

F.5.3.12.5 Species-Specific Threats

The northern Gulf of Mexico stock of false killer whales was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 18 percent of false killer whales in the Gulf of Mexico were exposed to oil, resulting in 6 percent excess mortality above baseline conditions, 8 percent excess failed pregnancies, and 7 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the northern Gulf of Mexico false killer whale stock will take an estimated 42 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.13 Fraser's Dolphin (*Lagenodelphis hosei*)

F.5.3.13.1 Status and Management

The Gulf of Mexico population of Fraser's dolphin is provisionally being considered as a separate stock for management purposes, although there are no genetic data to differentiate this stock from the western North Atlantic stock (Waring et al., 2013).

F.5.3.13.2 Habitat and Geographic Range

Fraser's dolphin is a tropical, oceanic species, except where deep-water approaches the coast (Dolar, 2009). Fraser's dolphins likely occur in the Gulf Stream open-ocean area.

This species is assumed to occur in the tropical western North Atlantic, although only a single sighting of approximately 250 individuals was recorded in waters 3,300 m deep off Cape Hatteras during a 1999 vessel survey (Hayes et al., 2019). Monthly aerial surveys offshore of Cape Hatteras from 2011 to 2017 resulted in only one sighting of Fraser's dolphins offshore of the 1,500-m isobaths (U.S. Department of the Navy, 2013a). The first record for the Gulf of Mexico was a mass stranding in the Florida Keys in 1981 (Hersh & Odell, 1986; Leatherwood et al., 1993). Since then, there have been documented strandings on the west coast of Florida and in southern Texas (Yoshida et al., 2010). Sightings of Fraser's dolphin in the northern Gulf of Mexico typically occur in oceanic waters greater than 200 m. This species was observed in the northern Gulf of Mexico during all seasons.

F.5.3.13.3 Population Trends

There are insufficient data to determine population trends for the western North Atlantic stock of Fraser's dolphins (Waring et al., 2007).

There are also insufficient data to determine population trends for the northern Gulf of Mexico stock of Fraser's dolphins. The large relative changes in the total abundances of Fraser's dolphin are probably due to a number of factors. Fraser's dolphin is most certainly a resident species in the Gulf of Mexico but probably occurs in low numbers, and the survey effort is not sufficient to estimate the abundance of uncommon or rare species with precision. In addition, these temporal abundance estimates are difficult to interpret without a Gulf of Mexico-wide understanding of Fraser's dolphin abundance. Studies based on abundance and distribution surveys restricted to U.S. waters are unable to detect temporal shifts in distribution beyond U.S. waters that might account for any changes in abundance (Waring et al., 2013).

F.5.3.13.4 Predator and Prey Interactions

Fraser's dolphins feed on mid-water fishes, squids, and shrimps and has not been documented to be prey to any other species (Jefferson & Leatherwood, 1994; Perrin et al., 1994a). However, this species may be subject to predation by killer whales.

F.5.3.13.5 Species-Specific Threats

There are no significant species-specific threats to Fraser's dolphins in the northwest Atlantic or Gulf of Mexico.

F.5.3.14 Killer Whale (*Orcinus orca*)

F.5.3.14.1 Status and Management

Although some populations of killer whales, particularly in the Pacific Northwest, are extremely well studied, little is known about killer whale populations in most areas including the northwest Atlantic and

Gulf of Mexico. Throughout these regions, killer whales are not highly abundant in specific areas, but are observed in higher concentration in Antarctic waters. For management purposes, the western North Atlantic population and Gulf of Mexico population are considered separate stocks (Waring et al., 2010, 2013; 2016).

F.5.3.14.2 Habitat and Geographic Range

Killer whales are found in all marine habitats, from the coastal zone (including most bays and inshore channels) to deep oceanic basins, and from equatorial regions to the polar pack ice zones of both hemispheres. Although killer whales are also found in tropical waters and the open ocean, they are often most numerous in coastal waters and at higher latitudes (Dahlheim & Heyning, 1999). Killer whales are likely found in Labrador Current, Gulf Stream, and North Atlantic Gyre open-ocean areas.

Killer whales are considered rare and uncommon in waters of the U.S. Exclusive Economic Zone in the Atlantic Ocean (Katona et al., 1988; Waring et al., 2010, 2013). During the 1978 to 1981 Cetacean and Turtle Assessment Program surveys, there were 12 killer whale sightings, which made up 0.1 percent of the 11,156 cetacean sightings in the surveys (Cetacean and Turtle Assessment Program, 1982; Waring et al., 2010, 2013). Nearshore observations are rare. Forty animals were observed in the southern Gulf of Maine in September 1979 and 29 animals in Massachusetts Bay in August 1986 (Katona et al., 1988; Waring et al., 2010).

Sightings of killer whales in the Gulf of Mexico on surveys from 1921 to 1995 were in water depths ranging from 840 to 8,700 ft., with an average of 4,075 ft., and were most frequent in the north-central region of the Gulf of Mexico (Waring et al., 2013). Killer whales were seen only in the summer during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al., 1996; Mullin & Hoggard, 2000), were reported from May through June during vessel surveys (Maze-Foley & Mullin, 2006; Mullin & Fulling, 2004) and recorded in May, August, September, and November by earlier opportunistic ship-based sources (O'Sullivan & Mullin, 1997).

F.5.3.14.3 Population Trends

There are insufficient data to determine population trends for the western North Atlantic and Gulf of Mexico stocks of killer whales (Waring et al., 2013).

F.5.3.14.4 Predator and Prey Interactions

Killer whales are apex predators and feed on a variety of prey, including bony fishes, elasmobranchs (a class of fish composed of sharks, skates, and rays), cephalopods, seabirds, sea turtles, and other marine mammals (Fertl et al., 1996; Jefferson et al., 2015). Some populations are known to specialize in specific types of prey (Jefferson et al., 2015; Wade et al., 2009). The killer whale has no known natural predators; it is considered to be the top predator of the oceans (Ford et al., 2005).

F.5.3.14.5 Species-Specific Threats

There are no significant species-specific threats to killer whales in the northwest Atlantic or Gulf of Mexico.

F.5.3.15 Long-Finned Pilot Whale (*Globicephala melas*)

F.5.3.15.1 Status and Management

The structure of the Western North Atlantic stock of long-finned pilot whales is uncertain (Fullard et al., 2000; International Council of the Exploration of the Sea, 1993). Morphometric (Bloch & Lastein, 1993) and genetic (Fullard et al., 2000) studies have provided little support for stock structure across the

Atlantic (Fullard et al., 2000). However, Fullard et al., (2000) have proposed a stock structure that is related to sea-surface temperature: (1) a cold-water population west of the Labrador/North Atlantic Current and (2) a warm-water population that extends across the Atlantic in the Gulf Stream. The area of overlap between the long-finned and short-finned pilot whales occurs primarily along the shelf break off the coast of New Jersey between 38° North and 40° North latitude (Waring et al., 2016).

F.5.3.15.2 Habitat and Geographic Range

Long-finned pilot whales occur along the continental shelf break, in continental slope waters, and in areas of high topographic relief, inhabiting temperate and subpolar zones from North Carolina to North Africa (and the Mediterranean) and north to Iceland, Greenland, and the Barents Sea (Abend & Smith, 1999; Buckland et al., 1993; Leatherwood et al., 1976). Long-finned pilot whales are likely found in the Gulf Stream and Labrador Current open-ocean areas and may potentially be found in the North Atlantic Gyre.

In U.S. Atlantic waters, pilot whales (*Globicephala* spp.) are distributed principally along the continental shelf edge off the northeastern U.S. coast in winter and early spring, moving onto Georges Bank and into the Gulf of Maine and more northern waters in late spring (Abend & Smith, 1999; Cetacean and Turtle Assessment Program, 1982; Hamazaki, 2002; Payne & Heinemann, 1993). They remain in these areas through late autumn (Cetacean and Turtle Assessment Program, 1982; Payne & Heinemann, 1993). Pilot whales tend to occupy areas of high relief or submerged banks. They are also associated with the Gulf Stream wall and thermal fronts along the continental shelf edge. Long- and short-finned pilot whales overlap spatially along the mid-Atlantic shelf break between Cape Hatteras, North Carolina, and New Jersey (Payne & Heinemann, 1993).

F.5.3.15.3 Population Trends

A trend analysis has not been conducted for the western North Atlantic stock of long-finned pilot whales (Waring et al., 2016).

F.5.3.15.4 Predator and Prey Interactions

Both pilot whale species feed primarily on squid but also eat fish, including mackerel, cod, turbot, herring, hake, and dogfish (Bernard & Reilly, 1999). They are also known to feed on shrimp (Gannon et al., 1997; Jefferson et al., 2015). Feeding generally takes place at depths between 200 and 500 m (Jefferson et al., 2015), but dives may be as deep as 800 m (Heide-Jorgensen et al., 2002). Some accounts of pilot whale attacks on small marine mammals are known, but pilot whales are not generally known to prey on marine mammals (Weller et al., 1996). Killer whales are possible predators of long-finned pilot whales.

F.5.3.15.5 Species-Specific Threats

Long-finned pilot whales were included in the Atlantic Pelagic Longline Take Reduction Plan to reduce bycatch associated with the Atlantic pelagic longline fishery to a level approaching a zero mortality and serious injury rate within 5 years of implementation (74 *Federal Register* 23349). However, in 2021, NMFS proposed removing this species from the scope of the plan. In 2023, a finalized amendment removed the western North Atlantic stock of long-finned pilot whales from the scope of the plan, as the rate of mortality and serious injury has remained below the insignificance threshold since the plan was initially implemented (88 *Federal Register* 36965*).*

F.5.3.16 Melon-Headed Whale (*Peponocephala electra*)

F.5.3.16.1 Status and Management

For management purposes, the western North Atlantic population and Gulf of Mexico population of melon-headed whales are considered separate stocks, although genetic data that differentiate these two stocks is lacking (Waring et al., 2007; Waring et al., 2010, 2013).

F.5.3.16.2 Habitat and Geographic Range

Melon-headed whales are found worldwide in tropical and subtropical waters. They are occasionally reported at higher latitudes, but these movements are considered to be beyond their typical range, as records indicate these movements occurred during incursions of warm-water currents (Perryman et al., 1994). Melon-headed whales are most often found in offshore deep waters and could occur in the southern parts of the Gulf Stream and North Atlantic Gyre open-ocean areas.

Sightings of whales from the Western North Atlantic stock are rare, but a group of 20 whales was sighted during surveys in 1999 offshore of Cape Hatteras, and a group of 80 whales was also sighted off Cape Hatteras in 2002 in waters greater than 2,500 m deep (Waring et al., 2013). Deployment of high-frequency acoustic recording packages offshore of Cape Hatteras, Onslow Bay, Jacksonville, and the offshore areas near Norfolk Canyon from 2009 through 2015 have resulted in zero melon-headed whale detections.

This species was observed in deep waters of the Gulf of Mexico, well beyond the edge of the continental shelf and in waters over the abyssal plain, primarily west of Mobile Bay, Alabama (Davis & Fargion, 1996; Mullin et al., 1994c; Waring et al., 2010, 2013). Sightings of melon-headed whales in the northern Gulf of Mexico were documented in all seasons during GulfCet aerial surveys 1992 and 1998 (Hansen et al., 1996; Mullin & Hoggard, 2000).

F.5.3.16.3 Population Trends

There are insufficient data to determine the population trends for the Western North Atlantic stock of melon-headed whales (Waring et al., 2007).

While abundance estimates for the Gulf of Mexico exist, there were no significant differences between survey years (Garrison et al., 2020), and the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys.

F.5.3.16.4 Predator and Prey Interactions

Little is known on predators of melon-headed whales in the Atlantic, therefore information from other geographic areas is likely applicable to the Study Area. Melon-headed whales are believed to be preyed on by killer whales and were observed fleeing from killer whales in Hawaiian waters (Baird et al., 2006).

Melon-headed whales prey on squid, pelagic fishes, and occasionally crustaceans (Jefferson & Barros, 1997; Perryman, 2009). Most of the fish and squid families preyed upon by this species consist of mid-water forms found in waters up to 1,500 m deep, suggesting that feeding takes place deep in the water column (Jefferson & Barros, 1997).

F.5.3.16.5 Species-Specific Threats

The northern Gulf of Mexico stock of melon-headed whales was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 15 percent of melon-headed whales in the Gulf of Mexico were exposed to oil, resulting in 5 percent excess mortality above baseline conditions, 7 percent excess failed pregnancies, and 6 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the northern

Gulf of Mexico melon-headed whale stock will take an estimated 29 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.17 Pantropical Spotted Dolphin (*Stenella attenuata*)

F.5.3.17.1 Status and Management

The western North Atlantic and northern Gulf of Mexico populations are considered separate stocks for management purposes, although there is currently not enough information to distinguish the two (Waring et al., 2016).

F.5.3.17.2 Habitat and Geographic Range

The pantropical spotted dolphin is distributed in offshore tropical and subtropical waters of the Atlantic Ocean between about 40° North and 40° South (Baldwin et al., 1999; Perrin, 2009c). The species is much more abundant in the lower latitudes of its range. It is primarily found in deeper offshore waters but do approach the coast in some areas (Jefferson et al., 2008; Jefferson et al., 2015; Perrin, 2001). Pantropical spotted dolphins may occur in the Gulf Stream open-ocean area.

The pantropical spotted dolphin is the most sighted species of cetacean in the oceanic waters of the northern Gulf of Mexico. Pantropical spotted dolphins were seen in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al., 1996; Mullin & Hoggard, 2000). Most sightings of this species in the Gulf of Mexico and Caribbean occur over the lower continental slope (Mignucci-Giannoni et al., 2003; Moreno et al., 2005). Pantropical spotted dolphins in the offshore Gulf of Mexico do not appear to have a preference for any one specific habitat type, such as within the Loop Current, inside cold-core eddies, or along the continental slope (Baumgartner et al., 2001). Along the U.S. Atlantic coast, sightings have been concentrated in the slope waters east of New England and Florida (Waring et al., 2014). Sightings during surveys in the Atlantic, north of Cape Hatteras, have been along the continental slope, while sightings in waters south of Cape Hatteras were recorded over the Blake Plateau and in deeper offshore waters of the mid-Atlantic (Hayes et al., 2020).

F.5.3.17.3 Population Trends

There are insufficient data to determine population trends for the western North Atlantic stock of pantropical spotted dolphins. While there are available coast-wide abundance estimates for this species, the high uncertainty in these estimates limits the ability to detect a population trend. In addition, interannual variation in abundance may be caused by either changes in spatial distribution associated with environmental variability or changes in the population size of the stock.

Further analysis of Gulf of Mexico pantropical spotted dolphin survey data from 1991 to 2009 is required to determine whether changes in abundance have occurred (Waring et al., 2015). Additionally, a Gulf-wide assessment of pantropical spotted dolphin abundance has not been made (Waring et al., 2015).

F.5.3.17.4 Predator and Prey Interactions

Pantropical spotted dolphins prey on near-surface fishes, squid, and crustaceans and on some mid-water species (Perrin & Hohn, 1994). Results from various tracking and food habit studies suggest that pantropical spotted dolphins in the eastern tropical Pacific and off Hawaii feed primarily at night on surface and mid-water species that rise after dark with the deep scattering layer (stratified zones in the ocean, usually composed of marine organisms that migrate vertically from depth to surface and back again at different times of day) (Baird et al., 2001; Evans, 1994; Robertson & Chivers, 1997). Pantropical spotted dolphins may be preyed on by killer whales and sharks and were observed fleeing killer whales in Hawaiian waters (Baird et al., 2006). Other predators may include the pygmy killer whale, false killer whale, and occasionally the short-finned pilot whale (Perrin, 2009c).

F.5.3.17.5 Species-Specific Threats

The northern Gulf of Mexico stock of pantropical spotted dolphins was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 20 percent of pantropical spotted dolphins in the Gulf of Mexico were exposed to oil, resulting in 7 percent excess mortality above baseline conditions, 9 percent excess failed pregnancies, and 7 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the northern Gulf of Mexico pantropical spotted dolphin stock will take an estimated 39 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.18 Pygmy Killer Whale (*Feresa attenuata*)

F.5.3.18.1 Status and Management

For management purposes, the Gulf of Mexico population of pygmy killer whale is considered a separate stock, although there is not yet sufficient genetic information to differentiate this stock from the western North Atlantic stocks (Waring et al., 2007; Waring et al., 2013).

F.5.3.18.2 Habitat and Geographic Range

Although the pygmy killer whale has an extensive global distribution, it is not known to occur in high densities in any region and is, therefore, probably one of the least abundant pantropical delphinids (Waring et al., 2013). The pygmy killer whale is primarily an open-ocean deep-water species (Davis et al., 2000; Würsig et al., 2000). This species has a worldwide distribution in tropical and subtropical oceans and generally does not range poleward of 40° N or of 35° S (Donahue & Perryman, 2009; Jefferson et al., 2015). This species occurs in the North Atlantic Gyre and the Gulf Stream, although sightings are rare. Most observations outside the tropics are associated with strong, warm western boundary currents that effectively extend tropical conditions into higher latitudes (Ross & Leatherwood, 1994).

A group of six pygmy killer whales were sighted during a 1992 vessel survey of the western North Atlantic off of Cape Hatteras, North Carolina, in waters greater than 1,500 m deep, but this species was not sighted during subsequent surveys (Waring et al., 2007). Deployment of high-frequency acoustic recording packages offshore of Cape Hatteras, Onslow Bay, Jacksonville, and the offshore areas near Norfolk Canyon from 2007 through 2022 have resulted in zero pygmy killer whale detections. Strandings are recorded from primarily South Carolina and Georgia, with two from North Carolina and one from Massachusetts (Hayes et al., 2020).

In the northern Gulf of Mexico, the pygmy killer whale is found primarily in deeper waters off the continental shelf and in waters over the abyssal plain (Davis et al., 2000; Würsig et al., 2000). The majority of sightings are in the eastern oceanic Gulf of Mexico in waters ranging from 200 to 1,200 m in depth.

F.5.3.18.3 Population Trends

There are insufficient data to determine population trends for the western North Atlantic stock of pygmy killer whales (Waring et al., 2007).

A trend analysis has not been conducted for the northern Gulf of Mexico stock of pygmy killer whales (Waring et al., 2013). Further analysis of northern Gulf of Mexico pygmy killer whale survey data from 1991–2009 is required to determine whether changes in abundance have occurred over this period. Additionally, a Gulf-wide assessment of pygmy killer whale abundance has not been made (Waring et al., 2013).

F.5.3.18.4 Predator and Prey Interactions

The pygmy killer whale has no documented predators in the Atlantic of Gulf of Mexico; however, it may be subject to predation by killer whales and large sharks. Pygmy killer whales feed predominantly on fish and squid (Clarke, 1986; Donahue & Perryman, 2009; dos Santos & Haimovici, 2001). They are known to attack other dolphin species, apparently as prey, although this is not common (Jefferson et al., 2015; Perryman & Foster, 1980; Ross & Leatherwood, 1994).

F.5.3.18.5 Species-Specific Threats

The northern Gulf of Mexico stock of pygmy killer whales was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 15 percent of pygmy killer whales in the Gulf of Mexico were exposed to oil, resulting in 5 percent excess mortality above baseline conditions, 7 percent excess failed pregnancies, and 6 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the northern Gulf of Mexico pygmy killer whale stock will take an estimated 29 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.19 Risso's Dolphin (*Grampus griseus*)

F.5.3.19.1 Status and Management

For management purposes, Risso's dolphins in the Gulf of Mexico and the Atlantic Ocean are currently considered two separate stocks (Waring et al., 2010; Waring et al., 2016).

F.5.3.19.2 Habitat and Geographic Range

Risso's dolphins are distributed worldwide in tropical and temperate waters along the continental shelf break, over the continental slope, and the outer continental shelf (Baumgartner, 1997; Cañadas et al., 2002; Cetacean and Turtle Assessment Program, 1982; Davis et al., 1998; Green et al., 1992; Kruse et al., 1999; Mignucci-Giannoni, 1998). Risso's dolphins were also found in association with submarine canyons (Mussi et al., 2004). The range of the Risso's dolphin distribution in open-ocean waters of the North Atlantic is known to include the Gulf Stream and the southwestern portions of the North Atlantic Gyre.

In the northwest Atlantic, Risso's dolphins occur from Florida to eastern Newfoundland (Baird & Stacey, 1991; Leatherwood et al., 1976). Off the northeast U.S. coast, Risso's dolphins are distributed along the continental shelf edge from Cape Hatteras northward to Georges Bank during spring, summer, and autumn (Cetacean and Turtle Assessment Program, 1982). In winter, the range is in the mid-Atlantic Bight and extends outward into oceanic waters. In general, the population occupies the mid-Atlantic continental shelf edge year-round and is rarely seen in the Gulf of Maine. During 1990, 1991, and 1993, spring/summer surveys conducted along the continental shelf edge and in deeper oceanic waters sighted Risso's dolphins associated with strong bathymetric features, Gulf Stream warm core rings, and the Gulf Stream north wall, and along the shelf break (Hamazaki, 2002; Waring et al., 1992, 1993).

Monthly aerial survey efforts began in January 2015 in the offshore area near Norfolk Canyon and have resulted in 61 Risso's dolphin sightings through 2019, totaling 1,361 individuals (Cotter, 2019) (McAlarney et al., 2017, 2018).

Monthly aerial surveys offshore of Cape Hatteras since May 2011 have documented 24 Risso's dolphin sightings, primarily during the summer months. Risso's dolphins were sighted from inside the 100-m isobath out to 2,000-m water depth (McAlarney et al., 2014). Risso's dolphins were also one of the most commonly encountered pelagic dolphins found during surveys conducted in Onslow Bay, North Carolina and offshore of Jacksonville, Florida (McLellan et al., 2014). Risso's dolphins observed during aerial and vessel surveys conducted monthly between June 2007 and June 2010 offshore of Onslow Bay, North

Carolina were exclusively found over the continental shelf break and in deeper waters of the survey area (Read et al., 2014; U.S. Department of the Navy, 2013a).

Vessel surveys conducted offshore of Jacksonville, Florida have resulted in a few Risso's dolphin sightings (Swaim et al., 2015). Aerial surveys documented higher numbers of Risso's dolphin encounters, with 16 sightings occurring within deeper waters of the survey area (U.S. Department of the Navy, 2013a).

Risso's dolphins in the northern Gulf of Mexico occur throughout oceanic waters but are concentrated in continental slope waters (Baumgartner, 1997; Maze-Foley & Mullin, 2006).

F.5.3.19.3 Population Trends

A trend analysis has not been conducted for the western North Atlantic stock of Risso's dolphins (Waring et al., 2015).

While abundance estimates for the Gulf of Mexico exist, there were no significant differences between survey years (Garrison et al., 2020), and the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys.

F.5.3.19.4 Predator and Prey Interactions

Cephalopods and crustaceans are the primary prey for Risso's dolphins (Clarke, 1996), with feeding occurring predominantly at night (Jefferson et al., 2015).

This species of dolphin may be preyed on by both killer whales and sharks, although there is no documented report of predation by either species (Weller, 2009).

F.5.3.19.5 Species-Specific Threats

Risso's dolphins were included in the Atlantic Pelagic Longline Take Reduction Plan to reduce bycatch associated with the Atlantic pelagic longline fishery to a level approaching a zero mortality and serious injury rate within 5 years of implementation (74 *Federal Register* 23349). However, in 2021, NMFS proposed removing this species from the scope of the plan. In 2023, a finalized amendment removed the western North Atlantic stock of Risso's dolphins from the scope of the plan, as the rate of mortality and serious injury has remained below the insignificance threshold since the plan was initially implemented (88 *Federal Register* 36965*).*

The northern Gulf of Mexico stock of Risso's dolphin was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 8 percent of Risso's dolphins in the Gulf of Mexico were exposed to oil, resulting in 3 percent excess mortality above baseline conditions, 3 percent excess failed pregnancies, and 3 percent higher likelihood for other adverse health effects. The maximum reduction of the Risso's dolphin northern Gulf of Mexico population was only 3 percent; therefore, the trustees were not able to calculate the number of years it would take this stock to recover (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.20 Rough-Toothed Dolphin (*Steno bredanensis*)

F.5.3.20.1 Status and Management

Rough-toothed dolphins are among the most widely distributed species of tropical dolphins, but little information is available on population status (Jefferson, 2009b; Jefferson et al., 2008; Jefferson et al., 2015). The Western North Atlantic and Gulf of Mexico populations of the rough-toothed dolphin are considered two separate stocks for management purposes, but there is insufficient genetic information to differentiate these stocks (Waring et al., 2013; Wimmer & Whitehead, 2004).

F.5.3.20.2 Habitat and Geographic Range

The distribution of the rough-toothed dolphin is poorly understood worldwide. These dolphins are thought to be a tropical to warm-temperate species and have been historically reported in deep oceanic waters in the Atlantic, Pacific, and Indian Oceans as well as the Mediterranean and Caribbean Seas (Gannier & West, 2005; Leatherwood & Reeves, 1983; Perrin & Walker, 1975; Reeves et al., 2003). Rough-toothed dolphins occur in the Gulf Stream and North Atlantic Gyre open-ocean areas.

Rough-toothed dolphins were observed in both shelf and oceanic waters in the northern Gulf of Mexico (Fulling et al., 2003; Mullin & Fulling, 2003) and off the U.S. east coast from North Carolina to Delaware (Waring et al., 2014). In the western North Atlantic, tracking of five rough-toothed dolphins that were rehabilitated and released following a mass stranding on the east coast of Florida in 2005 demonstrated a variety of ranging patterns (Wells et al., 2008). All tagged rough-toothed dolphins moved through a large range of water depths averaging greater than 100 ft. (30 m), though each of the five tagged dolphins transited through very shallow waters at some point, with most of the collective movements recorded over a gently sloping seafloor. Monthly aerial surveys conducted offshore of Cape Hatteras, North Carolina from 2011 through 2017 only resulted in one sighting of four individual rough-toothed dolphins just beyond the 100-m isobaths (U.S. Department of the Navy, 2013a).

Aerial surveys conducted between 2009 and 2017 offshore of Jacksonville, Florida resulted in nine sightings of rough-toothed dolphins, primarily in the summer and fall months. Sightings from aerial surveys have been documented inside the 100-m isobaths in continental shelf waters (Cummings et al., 2016; U.S. Department of the Navy, 2013a).

Rough-toothed dolphins have been observed in all seasons in the Gulf of Mexico (Hansen et al., 1996; Mullin & Hoggard, 2000) but have not been sighted every survey year, attesting to their low density in this region.

F.5.3.20.3 Population Trends

A trend analysis has not been conducted for the Western North Atlantic stock of rough-toothed dolphins (Waring et al., 2014).

Further analysis of Gulf of Mexico rough-toothed dolphin survey data from 2003–2004 and 2009 is required to determine whether changes in abundance have occurred (Waring et al., 2013). Additionally, a Gulf-wide assessment of rough-toothed dolphin abundance has not been made (Waring et al., 2013).

F.5.3.20.4 Predator and Prey Interactions

Prey of rough-toothed dolphins includes fish and cephalopods. They are known to feed on large fishes such as mahi mahi (Miyazaki & Perrin, 1994; Pitman & Stinchcomb, 2002). They also prey on reef fish, and Perkins and Miller (1983) noted that parts of reef fish were found in the stomachs of stranded rough-toothed dolphins in Hawaii. Rough-toothed dolphins also feed during the day on near-surface fishes, including flying fishes (Gannier & West, 2005).

Predation on rough-toothed dolphins has not been documented, but they may be subject to predation by killer whales.

F.5.3.20.5 Species-Specific Threats

The northern Gulf of Mexico stock of rough-toothed dolphins was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that

41 percent of rough-toothed dolphins in the Gulf of Mexico were exposed to oil, resulting in 14 percent excess mortality above baseline conditions, 19 percent excess failed pregnancies, and 15 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the northern Gulf of Mexico rough-toothed dolphin stock will take an estimated 54 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.21 Short-Finned Pilot Whale (*Globicephala macrorhynchus*)

F.5.3.21.1 Status and Management

Studies are currently being conducted at the NMFS Southeast Fisheries Science Center to evaluate genetic population structure in short-finned pilot whales (Waring et al., 2016). The short-finned pilot whale population is managed as three stocks: Western North Atlantic stock, Puerto Rico and U.S. Virgin Islands stock, and the Gulf of Mexico Oceanic stock.

F.5.3.21.2 Habitat and Geographic Range

Short-finned pilot whales range throughout warm, temperate to tropical waters of the world, generally in deep offshore areas (Waring et al., 2016). Thus, the species occupies waters over the continental shelf break, in slope waters, and in areas of high topographic relief (Olson, 2009). While pilot whales are typically distributed along the continental shelf break, movements over the continental shelf are commonly observed in the northeastern United States. Genetic analysis of stranded pilot whales, evaluated as a function of sea surface temperature and water depth, indicated that short-finned pilot whales were not likely to be found at water temperatures less than 22° C and highly likely to occur where water temperatures were greater than 25° C. Probability of a short-finned pilot whale occurrence also increased with increasing water depth. The area of overlap between short-finned and long-finned pilot whales occurs primarily along the shelf break off the coast of New Jersey between 38° North and 40° North latitude (Waring et al., 2014). Short-finned pilot whales are likely found in the Gulf Stream open-ocean area.

Sightings of pilot whales (*Globicephala* spp.) in the western North Atlantic occur primarily near the continental shelf break ranging from Florida to the Nova Scotian Shelf (Mullin & Fulling, 2003). Long-finned and short-finned pilot whales overlap spatially along the mid-Atlantic shelf break between Cape Hatteras, North Carolina, and New Jersey (Payne & Heinemann, 1993). Long-finned pilot whales have occasionally been observed stranded as far south as Florida, while short-finned pilot whales have occasionally been observed stranded as far north as Massachusetts (Pugliares et al., 2016).

Pilot whales are one of the most common cetacean species observed off Cape Hatteras during aerial surveys, specifically from the 100-m isobaths out to water depths greater than 2,000 m (U.S. Department of the Navy, 2013a). While photo-identification work suggests that short-finned pilot whales display a high degree of residence off Cape Hatteras, satellite tagging demonstrates that these animals cover a significant range up and down the continental slope, from Georges Bank in the north, down to Cape Lookout Shoals in the south, with movements at least occasionally into waters beyond the U.S. Exclusive Economic Zone (Baird et al., 2015, 2016). Thirty-nine satellite telemetry tags were deployed on short-finned pilot whales off the coast of Cape Hatteras during the summers of 2014 and 2015. This study provided the first information on long-term and long-distance movements of short-finned pilot whales in the area, other than information obtained from tags on previously stranded and rehabilitated individuals.

Deployment of high-frequency acoustic recording packages offshore of Cape Hatteras, Onslow Bay, Jacksonville, and the offshore areas near Norfolk Canyon from 2007 through 2022 has resulted in zero short-finned pilot whale detections. Passive acoustic data were collected from marine autonomous recording units deployed on the continental shelf, just beyond the shelf, and offshore from the shelf break off Jacksonville, Florida in late 2009 and early 2010. These deployments resulted in detections of the blackfish group of cetaceans, which includes short-finned pilot whales, along with melon-headed whales, pygmy killer whales, false killer whales, and killer whales. Blackfish were detected every day during deployments but there were no obvious or consistent differences in the occurrence of blackfish vocalizations relative to water depth or time of day (Oswald et al., 2016). The combination of five species into the blackfish category may have masked any patterns in vocal behaviors (Oswald et al., 2016).

Short-finned pilot whales are also documented along the continental shelf and continental slope in the northern Gulf of Mexico (Hansen et al., 1996; Mullin & Fulling, 2003; Mullin & Hoggard, 2000), and in the Caribbean. Short-finned pilot whales were seen in all seasons during GulfCet aerial surveys of the northern Gulf of Mexico between 1992 and 1998 (Hansen et al., 1996; Mullin & Hoggard, 2000).

F.5.3.21.3 Population Trends

A trend analysis has not been conducted for the western North Atlantic stock of short-finned pilot whales (Waring et al., 2016).

While abundance estimates for the Gulf of Mexico exist, there were no significant differences between survey years (Garrison et al., 2020), and the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys.

F.5.3.21.4 Predator and Prey Interactions

Pilot whales feed primarily on squid, to which they are generally well adapted (Jefferson et al., 2008; Werth, 2006), but they also forage on fish (Bernard & Reilly, 1999). Pilot whales are not generally known to prey on other marine mammals, but records from the eastern tropical Pacific suggest that the short-finned pilot whale does occasionally chase, attack, and even predate dolphins during fishery operations (Olson, 2009; Perryman & Foster, 1980). They were also observed harassing sperm whales in the Gulf of Mexico (Weller et al., 1996). This species is not known to have any predators (Weller, 2009), but it may be subject to predation by killer whales.

F.5.3.21.5 Species-Specific Threats

Short-finned pilot whales were included in the Atlantic Pelagic Longline Take Reduction Plan to reduce bycatch associated with the Atlantic pelagic longline fishery to a level approaching a zero mortality and serious injury rate within 5 years of implementation. Pursuit of this goal also takes into account the economic status of the fishery, technological availability, as well as current fishery management plans on both state and regional levels (88 *Federal Register* 36965).

The northern Gulf of Mexico stock of short-finned pilot whales was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 6 percent of short-finned pilot whales in the Gulf of Mexico were exposed to oil, resulting in 2 percent excess mortality above baseline conditions, 3 percent excess failed pregnancies, and 2 percent higher likelihood for other adverse health effects. The maximum reduction of the short-finned pilot whale Gulf of Mexico population was only 3 percent; therefore, the trustees were not able to calculate the number of years it would take for this stock to recover (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.22 Spinner Dolphin (*Stenella longirostris*)

F.5.3.22.1 Status and Management

For management purposes, the western North Atlantic and Gulf of Mexico populations of spinner dolphins are considered separate stocks, although there is currently insufficient data to differentiate them (Waring et al., 2014).

F.5.3.22.2 Habitat and Geographic Range

This is presumably an offshore, deep-water species (Perrin & Gilpatrick, 1994; Schmidly, 1981), although its distribution in the Atlantic is poorly known. Spinner dolphins likely occur in the Gulf Stream and North Atlantic Gyre open-ocean areas, based on their preference for waters greater than 2,000 m deep.

In the western North Atlantic, these dolphins occur in deep water along most of the U.S. coast south to the West Indies and Venezuela, including the Gulf of Mexico (Waring et al., 2014). Spinner dolphin sightings have occurred exclusively in deeper (greater than 2,000 m) oceanic waters of the northeast U.S. coast (Cetacean and Turtle Assessment Program, 1982; Waring et al., 1992). Stranding records exist from North Carolina, South Carolina, Florida, and Puerto Rico in the Atlantic as well as in Texas and Florida in the Gulf of Mexico, while there was one recent sighting during summer 2011 in oceanic waters off North Carolina. Monthly aerial surveys offshore of Cape Hatteras conducted from 2011 to 2019 have only resulted in one sighting of spinner dolphins in a mixed group of Clymene dolphins within the northern offshore waters of the survey area (U.S. Department of the Navy, 2013a). Although spinner dolphins were sighted and stranded off the southeastern U.S. coast, they are not common in those waters, except perhaps off southern Florida (Waring et al., 2010). In the northern Gulf of Mexico, spinner dolphins are found mostly in offshore waters beyond the edge of the continental shelf and primarily east of the Mississippi River (Waring et al., 2013). This species was seen during all seasons in the northern Gulf of Mexico during aerial surveys between 1992 and 1998 (Waring et al., 2013).

F.5.3.22.3 Population Trends

Due to imprecise abundance estimates and long periods of time between surveys, a trend analysis has not been conducted for the western North Atlantic stock of spinner dolphins (Waring et al., 2014).

While abundance estimates for the Gulf of Mexico exist, there were no significant differences between survey years (Garrison et al., 2020), and the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys.

There are insufficient data to determine the population trends for the Puerto Rico and U.S. Virgin Islands stock of spinner dolphins (Waring et al., 2012).

F.5.3.22.4 Predator and Prey Interactions

Spinner dolphins feed primarily on small mid-water fishes, squids, and shrimp and dive to at least 1,300 ft. (400 m) (Perrin & Gilpatrick, 1994). Studies of spinner dolphins in the Pacific suggest they forage primarily at night, when the mid-water community migrates toward the surface and the shore (Benoit-Bird, 2004; Benoit-Bird et al., 2001). Spinner dolphins track the horizontal migrations of their prey (Benoit-Bird & Au, 2003), allowing foraging efficiencies (Benoit-Bird, 2004; Benoit-Bird & Au, 2004). Foraging behavior was also linked to lunar phases in scattering layers off the island of Hawaii (Benoit-Bird & Au, 2004). Similar foraging behavior is expected for spinner dolphins that occur in the Study Area.

Spinner dolphins may be preyed on by sharks, killer whales, pygmy killer whales, and short-finned pilot whales (Perrin, 2009d).

F.5.3.22.5 Species-Specific Threats

The northern Gulf of Mexico stock of spinner dolphins was 1 of the 31 cetacean stocks impacted by the 2010 *Deepwater Horizon* oil spill in the Gulf of Mexico. Injury quantification determined that 47 percent of spinner dolphins in the Gulf of Mexico were exposed to oil, resulting in 16 percent excess mortality above baseline conditions, 21 percent excess failed pregnancies, and 17 percent higher likelihood for other adverse health effects. Spinner dolphins were determined to take the longest to recover compared to all cetacean stocks impacted by the oil spill, as this species resulted in the highest maximum reduction in population size. Without active restoration efforts, recovery of the northern Gulf of Mexico spinner dolphin stock will take an estimated 105 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.23 Striped Dolphin (*Stenella coeruleoalba*)

F.5.3.23.1 Status and Management

For management purposes, the Gulf of Mexico population of striped dolphin is provisionally considered as a separate stock, although there are not sufficient genetic data to differentiate the Gulf of Mexico stock from the western North Atlantic stock (Waring et al., 2010). There is very little information on stock structure in the western North Atlantic (Hayes et al., 2020).

F.5.3.23.2 Habitat and Geographic Range

The striped dolphin is one of the most common and abundant dolphin species, with a worldwide range that includes both tropical and temperate waters (Waring et al., 2014). Although primarily a warm-water species, the range of the striped dolphin extends higher into temperate regions than those of any other species in the genus *Stenella* (spotted, spinner, Clymene, and striped dolphins). Striped dolphins are found in the western North Atlantic from Nova Scotia south to at least Jamaica, as well as in the Gulf of Mexico. In general, striped dolphins appear to prefer continental slope waters offshore to the Gulf Stream (Leatherwood et al., 1976; Perrin et al., 1994b; Schmidly, 1981).

Striped dolphins are relatively common in the cooler offshore waters of the U.S. east coast. Along the mid-Atlantic ridge in oceanic waters of the North Atlantic Ocean, striped dolphins are sighted in significant numbers south of 50° North (Waring et al., 2010). In waters off the northeastern U.S. coast, striped dolphins are distributed along the continental shelf edge from Cape Hatteras to the southern margin of Georges Bank and also occur offshore over the continental slope and rise in the mid-Atlantic region (Cetacean and Turtle Assessment Program, 1982; Mullin & Fulling, 2003). Continental shelf edge sightings in the Cetacean and Turtle Assessment Program (1982) were generally centered along the 1,000-m depth contour in all seasons. During 1990 and 1991 cetacean habitat-use surveys, striped dolphins were associated with the Gulf Stream north wall and warm-core ring features (Waring et al., 1992). Striped dolphins seen in a survey of the New England Sea Mounts (Palka, 1997) were in waters that were between 20 and 27°C and deeper than about 3,000 ft. (900 m).

Regular periodic aerial surveys in the offshore area near Norfolk Canyon from 2015 to 2019 resulted in six striped dolphin sightings (McAlarney et al., 2016). Aerial surveys offshore of Cape Hatteras from 2011 to 2017 have resulted in five striped dolphin sightings, primarily in late winter and early spring.

Striped dolphins are also found throughout the deep, offshore waters of the northern Gulf of Mexico. Sightings of striped dolphins in the northern Gulf of Mexico typically occur in oceanic waters and during all seasons (Waring et al., 2010).

F.5.3.23.3 Population Trends

A trend analysis has not been conducted for the western North Atlantic stock of striped dolphins (Waring et al., 2014).

While abundance estimates for the Gulf of Mexico exist, there were no significant differences between survey years (Garrison et al., 2020), and the statistical power to detect a trend in abundance for this stock is poor due to the relatively imprecise abundance estimates and long intervals between surveys.

F.5.3.23.4 Predator and Prey Interactions

Striped dolphins often feed in open sea or sea bottom zones along the continental slope, or just beyond it, in oceanic waters. Most of their prey possess light-emitting organs, suggesting that striped dolphins may be feeding at great depths, possibly diving from 655 to 2,295 ft. (200 to 700 m) (Archer & Perrin, 1999). Striped dolphins may feed at night to take advantage of the deep scattering layer's diurnal vertical movements. Small mid-water fishes (in particular lanternfishes) and squids are the predominant prey (Perrin & Gilpatrick, 1994).

This species was documented to be preyed upon by sharks (Ross & Bass, 1971). They may also be subject to predation by killer whales.

F.5.3.23.5 Species-Specific Threats

The northern Gulf of Mexico stock of striped dolphins was 1 of the 31 cetacean stocks impacted by the 2010 Deepwater Horizon oil spill in the Gulf of Mexico. Injury quantification determined that 13 percent of striped dolphins in the Gulf of Mexico were exposed to oil, resulting in 5 percent excess mortality above baseline conditions, 6 percent excess failed pregnancies, and 5 percent higher likelihood for other adverse health effects. Without active restoration efforts, recovery of the northern Gulf of Mexico striped dolphin stock will take an estimated 14 years (Deepwater Horizon Natural Resource Damage Assessment Trustees, 2016).

F.5.3.24 White-Beaked Dolphin (*Lagenorhynchus albirostris*)

F.5.3.24.1 Status and Management

There are at least two separate stocks of the white-beaked dolphin in the North Atlantic: one in the eastern and another in the western North Atlantic, although the genus *Lagenorhynchus* is currently proposed to be revised (Vollmer et al., 2019).

F.5.3.24.2 Habitat and Geographic Range

White-beaked dolphins are found in cold-temperate and subarctic waters of the North Atlantic (Waring et al., 2007). In the western North Atlantic Ocean, the white-beaked dolphin occurs throughout northern waters of the Atlantic of the United States and eastern Canada, from eastern Greenland through the Davis Strait and south to Massachusetts (Lien et al., 2001). White-beaked dolphins would be expected to occur in the Labrador Current.

Within the Study Area, white-beaked dolphins are concentrated in the western Gulf of Maine and around Cape Cod (Cetacean and Turtle Assessment Program, 1982; Palka et al., 1997). Before the 1970s, these dolphins were found primarily in waters over the continental shelf of the Gulf of Maine and Georges Bank. Since then, they have been replaced by large numbers of Atlantic white-sided dolphins and now occur mainly in waters over the continental slope (Katona et al., 1993; Palka et al., 1997). This habitat shift may be a result of an increase in sand lance and a decline of herring in continental shelf waters (Payne et al., 1990). Sightings are common in nearshore waters of Newfoundland and Labrador (Lien et al., 2001). They also occur in the Gulf of St. Lawrence (Waring et al., 2010). During Cetacean and Turtle Assessment Program (1982) surveys, white-beaked dolphins were typically sighted in shallow coastal waters near Cape Cod and along Stellwagen Bank, with a bottom depth ranging from 43 to 2,454 ft. (Palka et al., 1997).

F.5.3.24.3 Population Trends

Abundance has declined in some areas, such as the Gulf of Maine, but this may be more closely related to habitat shifts than to direct changes in population size. However, there are insufficient data to determine population trends for this species (Waring et al., 2007).

F.5.3.24.4 Predator and Prey Interactions

This species preys on small mid-water and schooling fish, such as herring and haddock, as well as squid and crustaceans, with observations of cooperative feeding behavior (Jefferson et al., 2008).

Killer whales and sharks are considered possible predators to the white-beaked dolphin. Although no attacks were documented, groups of white-beaked dolphin were observed fleeing from killer whales (Kinze, 2009).

F.5.3.24.5 Species-Specific Threats

There are no significant species-specific threats to white-beaked dolphins in the northwest Atlantic.

F.5.3.25 Harbor Porpoise (*Phocoena phocoena*)

F.5.3.25.1 Status and Management

The Gulf of Maine–Bay of Fundy stock is the only stock of harbor porpoise under NMFS management within the Study Area. There are three additional harbor porpoise populations that also occur within the Study Area: Gulf of St. Lawrence, Newfoundland, and Greenland (Gaskin, 1992).

F.5.3.25.2 Habitat and Geographic Range

Harbor porpoises inhabit cool temperate-to-subpolar waters, often where prey aggregations are concentrated (Watts & Gaskin, 1985). Thus, they are frequently found in shallow waters, most often near shore, but they sometimes move into deeper offshore waters. Harbor porpoises are rarely found in waters warmer than 63°F (17°C) (Read, 1999) and closely follow the movements of their primary prey, Atlantic herring (Gaskin, 1992).

Harbor porpoises would likely be found only in the Labrador Current open-ocean area. In the western North Atlantic, harbor porpoises range from Cumberland Sound on the east coast of Baffin Island, southeast along the eastern coast of Labrador to Newfoundland and the Gulf of St. Lawrence, and southwest to about 34° North on the coast of North Carolina (Waring et al., 2016). Harbor porpoises are also found in waters off southwest Greenland. During summer (July to September), harbor porpoises are concentrated in the northern Gulf of Maine and southern Bay of Fundy region, generally in waters less than 150 ft. deep (Gaskin, 1977; Kraus et al., 1983; Palka, 1995a; Palka, 1995b), with a few sightings in the upper Bay of Fundy and on the northern edge of Georges Bank (Palka, 2000).

During winter (January to March), intermediate densities of harbor porpoises can be found in waters off New Jersey to North Carolina, while lower densities are found in waters off New York to New Brunswick, Canada (Waring et al., 2016). Harbor porpoises sighted off the mid-Atlantic states during winter include porpoises from other western North Atlantic populations (Rosel et al., 1999). There does not appear to be a temporally coordinated migration or a specific migratory route to and from the Bay of Fundy region (Waring et al., 2016).

LaBrecque et al. (2015a) identified a small and resident population area for harbor porpoise in the Gulf of Maine (Figure F.5-12) based on sightings documented by the National Oceanic and Atmospheric Administration Fisheries ship and aerial surveys, strandings, and animals taken incidental to fishing reported by National Oceanic and Atmospheric Administration Fisheries observers. From July to September, harbor porpoises are concentrated in waters less than 150 m deep in the northern Gulf of Maine and southern Bay of Fundy. During fall (October to December) and spring (April to June), harbor porpoises are widely dispersed from New Jersey to Maine, with lower densities farther north and south (LaBrecque et al., 2015a).

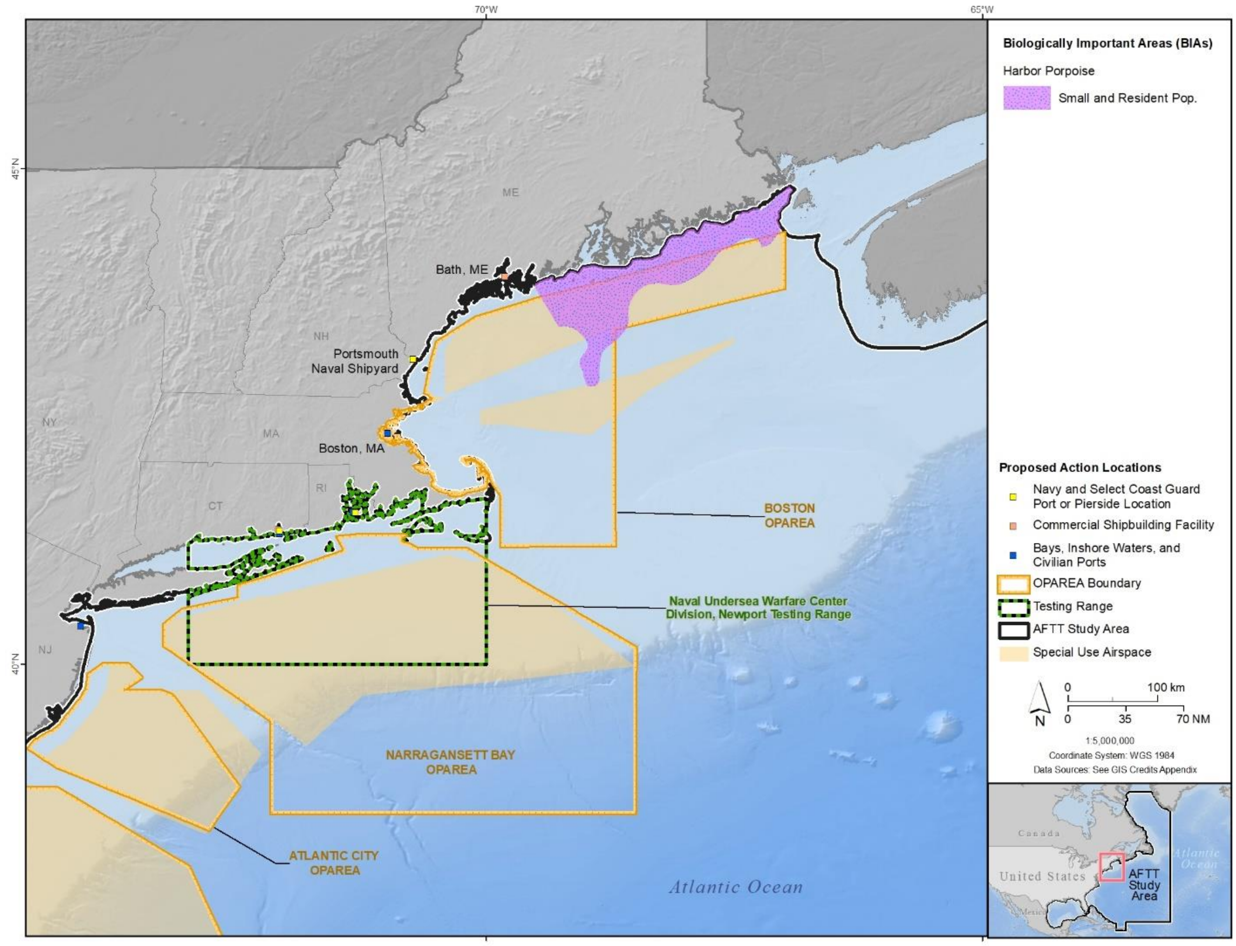


Figure F.5-12: Biologically Important Areas for Harbor Porpoises in the Study Area

F.5.3.25.3 Population Trends

A trend analysis has not been conducted for the Gulf of Maine–Bay of Fundy stock of harbor porpoises (Palka, 2012). Since there are no population estimates available for the Gulf of St. Lawrence, Newfoundland, or Greenland stocks, trend analyses have not been conducted for these populations either (Waring et al., 2016).

F.5.3.25.4 Predator and Prey Interactions

This species preys on a variety of fish, especially high-fat pelagic species such as herring, sprat, anchovy, and cephalopods (Berrow & Rogan, 1996; Bjorge & Tolley, 2009; Santos & Pierce, 2003). The harbor porpoise is known to be attacked and killed by killer whales and common bottlenose dolphins (Jefferson et al., 2015).

F.5.3.25.5 Species-Specific Threats

Harbor porpoises have been documented as bycatch in a variety of fisheries, including sink and drift gillnets, herring weirs, and pelagic longlines (Hayes et al., 2018; Zollett, 2009). The total annual estimated human-caused mortality and serious injury is 307 harbor porpoises per year (CV = 0.16) from U.S. fisheries (Hayes et al., 2018).

F.5.3.26 Gray Seal (*Halichoerus grypus*)

F.5.3.26.1 Status and Management

There are three main populations of gray seal in the North Atlantic, including the Northeast Atlantic, Northwest Atlantic, and the Baltic Sea (Katona et al., 1993; Waring et al., 2010; Waring et al., 2016). These stocks are separated by geography, different breeding seasons, and genetic variation (Waring et al., 2010). Genetic research indicates that gray seals found in U.S. waters along the coasts of Maine and Massachusetts are descended from the Canadian population and are members of the Northeast Atlantic stock (Hayes et al., 2021). The percentage of time that individuals are resident in U.S. waters is unknown (Hayes et al., 2021).

F.5.3.26.2 Habitat and Geographic Range

The Western North Atlantic management stock corresponds to the eastern Canada population, primarily ranging from Labrador to New Jersey (Hammill et al., 1998; Waring et al., 2004). This gray seal population is centered in the Canadian Maritimes, including the Gulf of St. Lawrence and the Atlantic coasts of Nova Scotia, Newfoundland, and Labrador. In the Study Area, the primary range of this species includes the northwestern waters of the Newfoundland-Labrador Shelf, the Scotian Shelf, and the Northeast U.S. Continental Shelf (Davies, 1957; Hall & Thompson, 2009).

The gray seal is considered a coastal species and may forage far from shore but does not appear to leave the continental shelf regions (Lesage & Hammill, 2001). Gray seals haul-out on land-fast ice, exposed reefs, or beaches of undisturbed islands (Hall & Thompson, 2009; Lesage & Hammill, 2001). Remote uninhabited islands tend to have the largest gray seal haul-outs (Reeves et al., 1992).

The Canadian population is divided into three groups for management purposes: Sable Island, Gulf of St. Lawrence, and Coastal Nova Scotia (Hammill et al., 2014a). The largest pupping site of gray seals in the world is located at Sable Island (Bowen et al., 2007). In the Gulf of St. Lawrence, gray seals pup on the pack ice (Davies, 1957; Hammill & Gosselin, 1995; Hammill et al., 1998), which is second-largest breeding colony in eastern Canada (Hammill et al., 2014a). Smaller numbers of seals pup on islands along the coast of Nova Scotia (Hammill et al., 2014a).

Gray seals range south into the northeastern U.S., with strandings and sightings as far south as North Carolina (Hammill et al., 1998; Waring et al., 2004). Gray seal distribution along the U.S. Atlantic coast has shifted in recent years, with an increased number of seals reported in southern New England (Kenney, 2014; Waring et al., 2016). Surveys in coastal Virginia since 2014 indicate that gray seals are occasional visitors to this area with multiple observations at the surveyed haul-out sites since 2014 (Guins et al., 2023; Jones & Rees, 2023).

Along the coast of the United States, gray seals are known to pup at three or more colonies, including Muskeget Island, Massachusetts, which is the southernmost breeding site (Rough, 1995; Waring et al., 2004), and Green and Seal Islands, Maine (Waring et al., 2016). Pupping has also been reported at Matinicus Rock and Mount Desert Rock in Maine (Waring et al., 2016). Gray seals are observed in New England outside of the pupping season on Muskeget Island and Monomoy as well as locations along the shoreline between southern Maine and Woods Hole, Massachusetts.

F.5.3.26.3 Population Trends

Gray seal abundance is likely increasing in U.S. waters, but the rate of increase is unknown (Waring et al., 2016). Single-day pup counts at three U.S. established colonies detected an increase from the 2001 to 2002 through the 2007 to 2008 pupping season (Wood LaFond, 2009). However, no recent surveys or modeling of gray seal abundance in U.S. Atlantic waters are available (Waring et al., 2016).

F.5.3.26.4 Predator and Prey Interactions

Gray seals prey on a variety of demersal and bottom-dwelling organisms, as well as schooling fish, cephalopods, other mollusks, and occasionally sea birds (Jefferson et al., 2015). Atlantic cod, Atlantic herring, sand lance, mackerel, flatfish, and white hake were the most prominent types of fish in the diet of gray seals off Nova Scotia, Canada (Hammill et al., 2014b). They also likely prey on harbor porpoises (Haelters et al., 2012; Leopold et al., 2015) and harbor seals (van Neer et al., 2015). Feeding during the breeding season is minimal (Hauksson & Bogason, 1997). This species is preyed upon by sharks (Jefferson et al., 2015), and are likely to be preyed upon by killer whales (Weller, 2009).

F.5.3.26.5 Species-Specific Threats

A review of 405 cases of marine mammal mortalities on Cape Cod and southeastern Massachusetts from 2000 to 2006 concluded that gray seals are highly susceptible to human interaction; 45 percent of gray seal deaths were due to interactions with humans (Bogomolni et al., 2010). Stranding and bycatch data from Cape Cod, Connecticut, Rhode Island, and New York coasts between 1990 through 2012 were collected and analyzed to identify changes in stranding and bycatch trends. The analysis suggests that gray seal strandings and bycatch are increasing at rates between 18 and 22 percent since the early 1990s in the southern New England region (Johnston et al., 2015). However, the researchers note that beach counts of gray seals are also increasing in this area, and it is possible the increase in stranding and bycatch rates is attributable to the growth in population.

An unusual mortality event for both gray seals and harbor seals along the Northeast Coast was declared from 2018 to 2020 due to elevated mortalities in both species. The likely cause of this unusual mortality event was infectious disease, with the main pathogen being phocine distemper virus (National Marine Fisheries Service, 2022a). Seals of both species were showing clinical signs of the virus; stranding as far south as Virginia, although not in elevated numbers. This unusual mortality event was non-active as of 2021 and closure is pending. A second unusual mortality event was declared in June 2022 for both species in the northeast due to the pathogen avian influenza (HPAI) H5N1 was closed in July 2022 (National Marine Fisheries Service, 2024e).

F.5.3.27 Harbor Seal (*Phoca vitulina*)

F.5.3.27.1 Status and Management

Western Atlantic harbor seals that occur along the coast of the eastern United States and Canada represent a single population (Temte et al., 1991; Waring et al., 2010; Waring et al., 2016), though there is some uncertainty in the stock structure in the Atlantic Ocean.

F.5.3.27.2 Habitat and Geographic Range

The harbor seal is one of the most widely distributed seals, found in temperate to polar coastal waters of the northern hemisphere (Jefferson et al., 2008; Jefferson et al., 2015). Harbor seals occur in nearshore waters and are rarely found more than 20 km from shore; frequently occupying bays, estuaries, and inlets (Baird, 2001). Individual seals have been observed several kilometers upstream in coastal rivers (Baird, 2001). Haul-out sites vary but include intertidal and subtidal rock outcrops, sandbars, sandy beaches, and even peat banks in salt marshes (Burns, 2009; Gilbert & Guldager, 1998; Prescott, 1982; Schneider & Payne, 1983; Wilson, 1978). Harbor seals occur in the cold and temperate nearshore waters of the northwest Atlantic, typically north of 35° North (Waring et al., 2016). In the Study Area, their approximate range includes the Gulf of St. Lawrence, Scotian Shelf, Gulf of Maine, Bay of Fundy, and northeast U.S. continental shelf down to the Virginia/North Carolina border.

Harbor seals are found year-round in the coastal waters of eastern Canada and Maine; from September to May they also occur from southern New England to North Carolina although there have been rare sightings and strandings recorded as far south as Florida (Katona et al., 1993; Waring et al., 2010). A general southward movement from the Bay of Fundy to southern New England waters occurs in autumn and early winter (Barlas, 1999; Jacobs & Terhune, 2000; Rosenfeld et al., 1988; Whitman & Payne, 1990). A northward movement from southern New England to Maine and eastern Canada occurs before the pupping season, which takes place from mid-May through June along the Maine coast (DeHart, 2002; Kenney, 1994; Richardson et al., 1995; Whitman & Payne, 1990; Wilson, 1978). In the northeastern United States, breeding and pupping normally occur north of the New Hampshire and Maine borders, although breeding has been recorded historically as far south as Cape Cod (Katona et al., 1993; Waring et al., 2010). Several thousand seals overwinter between New Hampshire and Massachusetts (Waring et al., 2010).

Harbor seal distribution along the U.S. Atlantic coast has shifted in recent years, with an increased number of seals reported in southern New England to the mid-Atlantic region. Harbor seals have been consistently detected in the mid-Atlantic region from November through March, with as many as 45 individuals observed during a single day at haul-out sites in Virginia (Jones & Rees, 2022).

F.5.3.27.3 Population Trends

The number of harbor seals in U.S. Atlantic waters increased from the 1980s to 2010 (Waring et al., 2010). There is some evidence that the population may be declining and a trend analysis for the North Atlantic stock is currently underway, however it is not possible at this time to discriminate between population decline and geographic redistribution (Hayes et al., 2021).

F.5.3.27.4 Predator and Prey Interactions

The main prey species of the harbor seal are cod, hake, mackerel, herring, salmon, sardines, smelt, shad, capelin, sand eels, sculpins, and flatfish (Burns, 2009). Sand eels are the main prey for individuals foraging in the southern portion of their range, while cod is the main prey in other geographic areas. Harbor seals are also known to feed on cephalopods and crustaceans (Burns, 2009). Shrimp appears to be important in the diet of newly weaned pups (Burns, 2009). Off Massachusetts, harbor seals are known to depredate monkfish, skate, and flounder from gillnets (Rafferty et al., 2012). There is no seasonal variation in prey species, but capelin and herring are more numerous in the fall and winter (Hauksson & Bogason, 1997; Jefferson et al., 2015; Reeves et al., 1992). Killer whales and sharks are known to prey on adult harbor seals and pups may be preyed on by eagles, ravens, gulls, and coyotes (Burns, 2009; Weller, 2009).

F.5.3.27.5 Species-Specific Threats

Harbor seals in the western North Atlantic are common bycatch in commercial fisheries (Hammill et al., 2010).

An unusual mortality event for both gray seals and harbor seals along the Northeast Coast was declared from 2018 to 2020 due to elevated mortalities in both species. The likely cause of this unusual mortality event was infectious disease, with the main pathogen being phocine distemper virus (National Marine Fisheries Service, 2022a). Seals of both species were showing clinical signs of the virus; stranding as far south as Virginia, although not in elevated numbers. This unusual mortality event was non-active as of 2021 and closure is pending. A second unusual mortality event for both species in the northeast was declared in June 2022 and closed in July 2022 due to the pathogen avian influenza (HPAI) H5N1 (National Marine Fisheries Service, 2024e).

F.5.3.28 Harp Seal (*Pagophilus groenlandicus*)

F.5.3.28.1 Status and Management

Three distinct populations or stocks of harp seals are recognized, including one in the Barents Sea that breeds on the “East Ice” in the White Sea, a population off eastern Greenland that breeds on the “West

Ice" near Jan Mayen, and a third population in the northwest Atlantic off eastern Canada (Lavigne, 2009). The Western North Atlantic stock is the largest and is divided into two breeding herds: the Front herd, which breeds off the coast of Newfoundland and Labrador, and the Gulf herd, which breeds near the Magdalen Islands in the Gulf of St. Lawrence (Reeves et al., 2002b; Waring et al., 2014; Waring et al., 2004).

F.5.3.28.2 Habitat and Geographic Range

The primary range of harp seals spans throughout the Arctic, with a secondary range that includes the western waters of the Scotian Shelf and the Northeast U.S. Continental Shelf. Harp seals are closely associated with drifting pack ice, where they breed, molt, and forage in the surrounding waters (Lydersen & Kovacs, 1993; Ronald & Healey, 1981). Harp seals make extensive movements over much of the continental shelf within their winter range in the waters off Newfoundland (Bowen & Siniff, 1999).

Typically, harp seals are distributed in the pack ice of the North Atlantic segment of the Arctic Ocean and through Newfoundland and the Gulf of St. Lawrence (Reeves et al., 2002b). Most western North Atlantic harp seals congregate off the east coast of Newfoundland-Labrador (the Front herd) to pup and breed; the remainder (the Gulf herd) gathers to pup near the Magdalen Islands in the Gulf of St. Lawrence (Morissette et al., 2006; Ronald & Dougan, 1982).

The number of sightings and strandings of harp seals off the northeastern U.S. has been increasing since the 1990s, based on records from Maine to New Jersey, primarily during the months of January to May (Harris et al., 2002; McAlpine & Walker, 1999). A few sightings and strandings are also reported annually for Virginia and North Carolina (Lloyd, 2015; Soulen et al., 2013; Swingle et al., 2016). Most recently, two young harp seals stranded separately in Norfolk, Virginia in early 2022 (Epple et al., 2023). An increase in strandings along the U.S. east coast has been correlated with poor ice conditions in the Gulf of St. Lawrence whelping area (Soulen et al., 2013).

F.5.3.28.3 Population Trends

Currently available data are insufficient to determine a minimum population estimate for U.S. waters (Waring et al., 2013); thus, population trends are also unknown. Outside of U.S. waters in the Northwest Atlantic between Canada and Greenland, population models of harp seals indicate the population has been stable since the 1990's, and appears to be increasing due to an increase in reproductive rates and low removal. In 2017, the estimated population size in this area was 6.8 million individuals with a projected increase to 7.6 million harp seals by 2019 (Hammill et al., 2021).

F.5.3.28.4 Predator and Prey Interactions

Harp seals feed on a variety of prey, which vary with age class, season, location, and year (Lavigne, 2009). Prey preference studies have revealed that harp seals prefer small fish to crustaceans (Lindstrom et al., 1998). The main prey species of harp seals are capelin, Greenland halibut, and Arctic and polar cod (Hauksson & Bogason, 1997; Lavigne, 2009; Morissette et al., 2006). Harp seals rarely eat commercially important Atlantic cod (Lavigne, 2009). Most foraging occurs at depths of less than 90 m, although dives as deep as 568 m have been recorded (Folkow et al., 2004; Lydersen & Kovacs, 1993). Harp seals feed intensively during the winter and summer and less so during the spring and fall migrations, or during pupping and molting (Ronald & Healey, 1981). Some overlap and competition exist for prey between hooded and harp seals. This species is preyed upon by polar bears, killer whales, and sharks (Lavigne, 2009).

F.5.3.28.5 Species-Specific Threats

Although harp seals are documented to be taken incidentally in commercial fishing gear, the level of take is small compared to the size of the population. Harp seals are also hunted commercially in Canada and Greenland. Climate change may also threaten whelping areas (Bajzak et al., 2011). Although the population in the Northwest Atlantic between Canada and Greenland has been increasing in recent years, climate-induced reductions in sea ice cover throughout their range will become an important factor of their population dynamics as the Gulf of St. Lawrence and Northeast Newfoundland area is projected to be ice free by the end of the century (Hammill et al., 2021; Han et al., 2015; Han et al., 2019).

F.5.3.29 Hooded Seal (*Cystophora cristata*)

F.5.3.29.1 Status and Management

The International Council for the Exploration of the Sea/Northwest Atlantic Fisheries Organization Working Group on Harp and Hooded Seals currently recognizes three separate stocks of hooded seals: the Northwest Atlantic, Greenland Sea, and White Sea stocks (International Council for the Exploration of the Sea, 2014). None of these stocks falls under U.S. jurisdiction, but individuals are found in U.S. waters. The western North Atlantic stock (synonymous with the Northwest Atlantic stock) pups off the coast of eastern Canada; the whelping area for the Greenland Sea stock is in the “West Ice” near Jan Mayen Island, east of Greenland (Kovacs, 2009a); the White Sea stock is in the “East Ice” near the Barents Sea/Russia. The Western North Atlantic stock is divided further into three whelping areas: Newfoundland-Labrador, Gulf of St. Lawrence, and David Strait.

F.5.3.29.2 Habitat and Geographic Range

Hooded seals are distributed in the Arctic and the cold temperate North Atlantic Ocean (Bellido et al., 2007). At sea, hooded seals stay primarily near continental coastlines but are known to wander widely. This species follows the seasonal movement of pack ice, on which it breeds. In the Study Area, its primary range is around the Newfoundland-Labrador, West Greenland, and Scotian Shelf.

Most hooded seals occur in the western Atlantic (Stenson et al., 1996). They migrate between winter/spring pupping areas along the Canadian coast to summer and molting areas off Greenland. The western North Atlantic stock breeds and pups at three main areas around Canada, including the Gulf of St. Lawrence, north of Newfoundland in an area that is known as the Front, and Davis Strait (Hammill et al., 1997; Jefferson et al., 2008; Kovacs, 2009b). Based on satellite relay data loggers deployed on hooded seals during 2004 to 2008, males appeared to prefer areas with complex seabed relief such as Davis Strait and the Flemish cap, whereas females preferred the Labrador Shelf (Andersen et al., 2013).

Hooded seals are highly migratory and may wander as far south as Puerto Rico (Mignucci-Giannoni & Odell, 2001), with more frequent occurrences from Maine to Florida in winter-spring and summer-fall, respectively (Harris et al., 2001; McAlpine et al., 1999; Mignucci-Giannoni & Odell, 2001).

F.5.3.29.3 Population Trends

The number of hooded seals in the western North Atlantic is relatively well known, and the total Northwest Atlantic population size is reported to have increased from 1965 to 2005 (Hammill & Stenson, 2006). However, uncertainty about the relationship among whelping areas and lack of reproductive and mortality data makes it difficult to reliably assess the current population trend.

F.5.3.29.4 Predator and Prey Interactions

The main prey species of hooded seals are redfish and cod, but they forage on squid and Greenland halibut as well (Hammill et al., 1997; Hauksson & Bogason, 1997). Some overlap and competition exist for prey between hooded and harp seals (Tucker et al., 2009). This species is preyed on by polar bears and killer whales (Kovacs, 2009b).

F.5.3.29.5 Species-Specific Threats

Although hooded seals are documented to be taken incidentally in commercial fishing gear, the level of take is very small compared to the size of the population. Hooded seals are also hunted commercially in Canada. The hooded seal is likely one of the most sensitive arctic marine mammal species to climate change due to its dependence on pack ice and specialized feeding habits (Laidre et al., 2008).

F.6 REPTILES

The following information was updated since the 2018 Final EIS/OEIS (Section 3.8, Reptiles):

- In 2022, the Naval Undersea Warfare Center Division Newport, Rhode Island, provided updated density models for green, Kemp's ridley, loggerhead, and leatherback sea turtles in the Atlantic Ocean spanning from the northern Florida Keys to the Gulf of Maine and out to the United States Exclusive Economic Zone. The density distribution and abundance estimates provided by these models are incorporated into their respective sections and provide updates from the 2018 Final EIS/OEIS.
- In 2022, density models were produced by NOAA Southeast Fisheries Science Center for green, Kemp's ridley, leatherback, and loggerhead sea turtle populations in the Gulf of Mexico.
- In 2023, NMFS and the USFWS performed a 5-year review on the conservation status of the loggerhead sea turtle in the Northwest Atlantic Ocean, and the updated findings through August 2021 are reflected in this appendix.
- In 2022, the USFWS released a 5-year review on the American crocodile.
- NMFS and USFWS initiated a status review of leatherback sea turtles in December of 2017 to gather and review best available scientific and commercial data on the species, apply the Distinct Population Segment (DPS) Policy, and evaluate extinction risk of any potential DPSs, following NMFS' Listing Guidance. In 2020, NMFS and USFWS announced a petition to identify the Northwest Atlantic population as a DPS in addition to six other DPSs based on data collected. Although the best available data indicates the populations meet the criteria for significance and discreteness, they found that it would not further the purposes of the ESA to recognize and list seven distinct population segments separately as endangered under the ESA. The current global listing of the species remains in effect (85 *Federal Register* 48332).
- Updated nest counts for green, leatherback, and loggerhead sea turtles are provided by Florida Fish and Wildlife Commission.
- The USFWS published a proposed rule to designate new areas of critical habitat and modify existing critical habitat for five DPSs of green sea turtle (North Atlantic, South Atlantic, Central North Pacific DPSs, Central South Pacific, and Central West Pacific DPSs).

F.6.1 GENERAL BACKGROUND

All reptiles are ectotherms, commonly referred to as "cold-blooded" animals, that have adopted different strategies to use external sources of heat to regulate body temperature. In the Study Area, sea turtles, crocodilians, and diamondback terrapins are analyzed for potential impacts.

The sea turtles considered in this analysis are found in coastal waters and on nesting beaches of the United States Atlantic Coast, Gulf of Mexico, Caribbean Sea, and in open-ocean areas. The American crocodile inhabits coastal areas of south Florida where they are at the northern extreme of their range. American alligators are found throughout the southeastern United States, in estuaries and freshwater habitats along rivers and lakes. The diamondback terrapin is also found in nearshore and inshore waters along the Atlantic and Gulf coasts. All sea turtles analyzed in this document are ESA listed, along with the American crocodile. The American alligator is listed under the ESA classification of "threatened due to similarity of appearance" to the American crocodile. The diamondback terrapin is not ESA listed.

Sea turtles are highly migratory, long-lived reptiles that occur throughout the open-ocean and coastal regions of the Study Area. Generally, sea turtles are distributed throughout tropical to subtropical latitudes, with some species extending into temperate seasonal foraging grounds. In general, sea turtles spend most of their time at sea, with female sea turtles returning to land to nest. Habitat and distribution vary depending on species and life stages and is discussed further in the species profiles and summarized in the following sections.

Crocodilians are also long-lived reptiles whose life spans can be up to 70 years in the wild (Schubiger, 2023). Crocodilians control their body temperature by basking in the sun or moving to areas with warmer or cooler air and water temperatures. The American crocodile inhabits freshwater wetland habitats, including rivers, lakes, and reservoirs, and can also be found in brackish environments such as estuaries and swamps (Fishman et al., 2009). It occurs in the Study Area in coastal portions of the Caribbean and in Florida. The American alligator is found throughout the southeastern United States, from the Carolinas to Texas. Unlike American crocodiles, American alligators lack lingual salt glands and are therefore unable to remove excess salt from their bodies (Nifong & Silliman, 2017). Gardner et al. (2016) predictively modeled alligator occurrence in North Carolina and found a strong negative relationship between water salinity and alligator occurrence and abundance. Throughout their range, American alligators are usually found in freshwater wetland habitats, in slow-moving rivers, or in the brackish waters of swamps, marshes, and lakes. Neither species occurs in offshore oceanic waters.

Diamondback terrapins are a species of turtle that can be found along the eastern and gulf coasts of the United States, from Cape Cod (Massachusetts) to Texas. They are most common in salt marshes and shallow bays. They are usually found in brackish water and occasionally travel out into the open ocean, though access to fresh water needs to be somewhat regularly available. They cannot tolerate high-salinity water for long periods of time, as they may dehydrate. Diamondback terrapins can secrete excess salt from their bloodstream through salt glands around their eyes. Terrapins have a diet consisting of mussels, crabs, snails, and sometimes insects, worms, and algae. They begin mating in early spring, and females will come ashore to sandy beaches or dunes to lay their eggs about 4 to 8 in. under the sand, where her eggs will continue to develop until around August or September. This species is listed as vulnerable to extinction by the International Union for Conservation of Nature; anthropogenic threats such as coastal development, commercial fisheries, and sea level rise due to climate change all contribute to making terrapins an "at-risk" species (National Geographic, 2024).

Additional species profiles and information on the biology, life history, species distribution, and conservation of reptile species can also be found on the following organizations' websites:

- NMFS Office of Protected Resources (includes sea turtle species distribution maps)
- USFWS Ecological Services Field Office and Region Offices (for sea turtle nesting habitat and general locations of nesting beaches)
- OBIS-SEAMAP species profiles
- International Union for Conservation of Nature, Marine Turtle Specialist Group
- State resource agencies (for sea turtle nesting information, status and management for American alligators, American crocodiles, and diamondback terrapins)

F.6.1.1 Group Size

A sea turtle's group size varies from being a solitary animal to being among groups for foraging, mating, and nesting/hatching. Research finds unique patterns in group size and structure throughout the course of a sea turtle's life (Clabough et al., 2022). Nesting and hatching activity for each sea turtle species is described in their respective sections.

After nesting, sea turtles head back, individually, to their preferred foraging grounds. An individual sea turtle will remain in an area of a few hundred square kilometers during these inter-nesting periods, where they will forage until it is time to nest again. Some sea turtles spend their time in one foraging area during the summer and a different foraging area during the winter (Foley et al., 2014). However, sea turtles typically do not show territoriality, so individuals of the same species or differing species can have foraging areas that overlap during certain times of the year. For example, multispecies communities (green, Kemp's ridley, and loggerhead sea turtles) are commonly observed foraging in the northern Gulf of Mexico, specifically near Florida (Hart et al., 2018; Wildermann et al., 2019).

Crocodiles and alligators are territorial, but will gather in groups as juveniles (as a defense against predators), and as adults when exhibiting courtship behavior and feeding (Hidalgo-Ruz et al., 2012; National Park Service, 2012). For both American crocodiles and American alligators, courtship and mating take place during the spring warming period (typically April and May), and nesting and egg-laying is initiated during the early summer (Briggs-Gonzalez et al., 2017; Vliet, 2001).

Limited information is known about hibernation in diamondback terrapins, but they have been observed hibernating during the colder months of the year. During this time, they submerge themselves in the mud of creeks and marshes, where they remain completely inactive until warmer seasonal temperatures return (National Aquarium, 2024). Diamondback terrapins may hibernate individually or together in large groups (Sheridan et al., 2010). Pfau and Roosenburg (2010) used harvesting records in the Chesapeake Bay to estimate that large hibernating groups may number as many as 200 individual diamondback terrapins.

F.6.1.2 Habitat Use

Sea turtles are dependent on sandy beaches for nesting habitats, specifically in locations that have sand deposits that are, ideally, not inundated with tides or storm events prior to hatching. In water, sea turtle habitat use is dependent on species and life stage. Life stages greatly influence behavior which range from diving, foraging, mating, and migration strategies (Hart et al., 2016; Wildermann et al., 2019). Sea turtles spend 1 to 15 years (depending on species) in the open ocean with sparse observations during their oceanic dispersal. These years are termed "lost years" (Putman et al., 2019). Once they reach sexual maturity, some individuals return to natal beaches to reproduce, while others go to nearshore foraging grounds (Mansfield et al., 2017; Mansfield et al., 2021; Putman et al., 2019).

Alligators depend on brackish and fresh water estuarine wetland types, typically inhabiting swamps, lakes, and slow-moving rivers. Crocodiles also inhabit both freshwater and brackish water environments, and sometimes even marine environments, as they have a higher tolerance to salt water than alligators. Both depend on habitats with sufficient water to use as concealment for hunting and stalking of prey. Nesting habitats are on dry land along the edges of wetlands, with eggs being deposited in nests that they construct from soft mud and vegetation in both wetland and upland habitats (Britton, 2009).

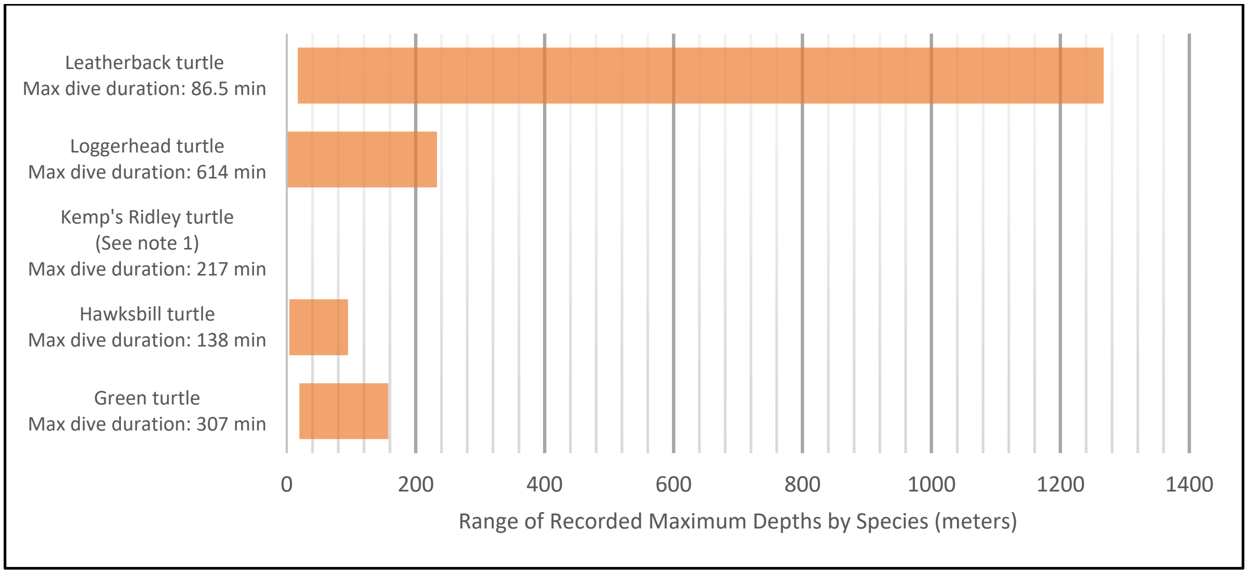
Although the diamondback terrapin is an aquatic turtle and they spend the majority of their life in water, they do leave the water to bask and lay eggs. One biological advantage these turtles have acquired over time is the ability to survive in salt waters of variable salinities. The typical habitat of the diamondback terrapin includes coastal swamps, estuaries, lagoons, tidal creeks, mangroves, and salt marshes with salinities ranging from 0 to 35 ppt (Pfau & Roosenburg, 2010).

F.6.1.3 Dive Behavior

While the American crocodile, American alligator, and diamondback terrapin do submerge themselves fully in water, they do not dive to depth and for extended periods of time in the way that sea turtles do; thus, these species are not discussed in this section.

Sea turtle dive depth and duration varies by species, the age of the animal, the location of the animal, and the activity (e.g., foraging, resting, and migrating). Roberts et al. (2022) analyzed dive and surface behavior of green, Kemp's ridley, and loggerhead sea turtles that were satellite tagged and tracked in the Gulf of Mexico from 2010 to 2019. Results showed that loggerheads spent a higher proportion of time at the surface while in warmer waters (between 25 and 30˚C) and at shallow to intermediate depth ranges. Kemp's ridley sea turtles occupied a wide temperature range at the surface (between 20 and 35˚C) with close proximity to the continental shelf and in very shallow waters (i.e., below 30 m). Green sea turtles spent more time at the surface during spring months and within shallow waters. A study by Rogers et al. (2024) tracked leatherback sea turtles to understand their surface behavior, and results showed that the species typically spend about 40 percent to 60 percent of the day at the surface of the water column, thus making them especially vulnerable to vessel strikes.

Dive durations are often a function of sea turtle size, with larger sea turtles being capable of diving to greater depths and for longer periods. The diving behavior of a particular species or individual has implications for mitigation, monitoring, and developing sound conservation strategies (Wildermann et al., 2019). In addition, their relative distribution throughout the water column is an important consideration when conducting acoustic exposure analyses. Methods of collecting dive behavior data over the years has varied in study design, configuration of electronic tags, parameters collected in the field, and data analyses. Hochscheid (2014) collected published data from 57 independent studies between 1986 and 2013, summarizing datasets that included depth and duration of dives from an overall total of 538 sea turtles. From this work, maximum dive depths (and duration in minutes) for each sea turtle species found in the Study Area is shown in Figure F.6-1 below.

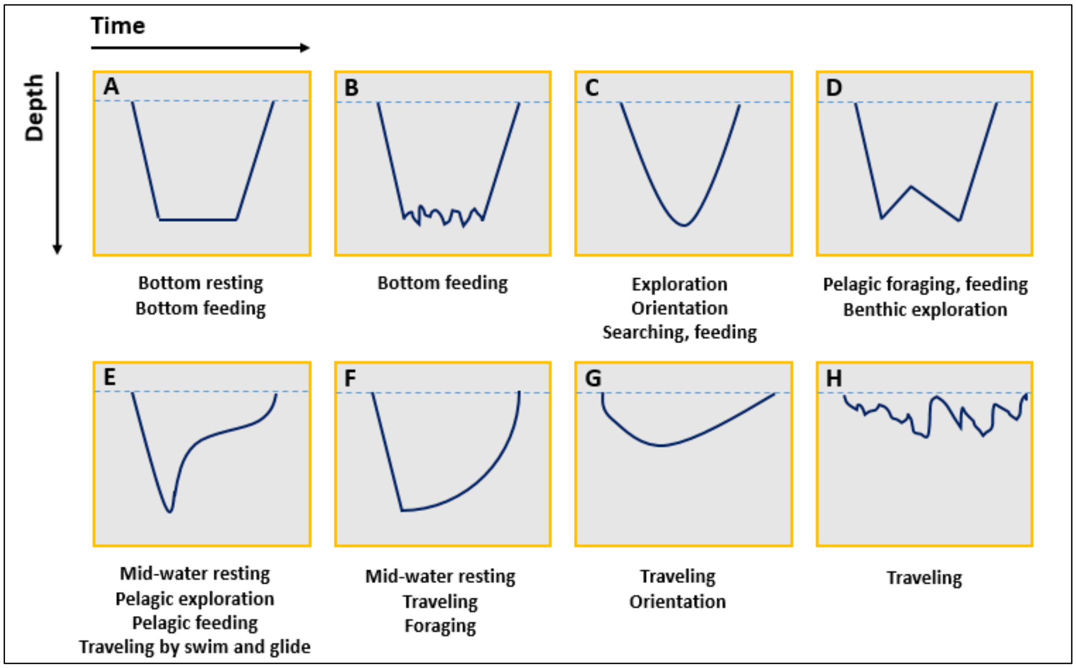


Sources: Hochscheid (2014); Sakamoto et al. (1993); Rice and Balazs (2008); Gitschlag (1996); Salmon et al. (2004).

Note: This figure shows the ranges of maximum dive depths and durations reported in the literature for the sea turtle species included in this analysis. Only one study was reviewed for Kemp's ridley sea turtle, which recorded depths of one juvenile Kemp's ridley sea turtle, and was not comparable to other data collected on other species.

Figure F.6-1: Dive Depth and Duration Summaries for Sea Turtle Species

Hochscheid (2014) also collected information on generalized dive profiles, with correlations to specific activities, such as bottom resting, bottom feeding, orientation and exploration, pelagic foraging and feeding, mid-water resting, and traveling during migrations. Generalized dive profiles compiled from 11 different studies by Hochscheid (2014) show eight distinct profiles tied to specific activities. These profiles and activities are shown in Figure F.6-2. Iverson et al. (2019) studied loggerhead dive behavior in the Gulf of Mexico, testing for associations between dive behavior (total daily dive frequency, frequency of dives to the bottom, frequency of long dives and time at depth), oceanographic conditions (sea surface temperature, net primary productivity), and behavioral mode (inter-nesting, migration, or foraging). This study found that loggerhead sea turtles dove to the bottom 95 percent of the days during inter-nesting and foraging. Other findings included that the majority of dives were less than or equal to 25 m, that longer dives occurred more frequently during migration, and most dives reached the sea floor when sea surface temperatures and net primary productivity were higher. Temperature was a key variable in this study, and results found that it has a strong influence on dive behavior. Migrating to deeper waters will keep the sea turtles cooler, allow exposure to high primary productivity with increased food availability and thus more foraging behavior, which occurs at depth during dives (Iverson et al., 2019).



Sources : Hochscheid (2014); Rice and Balazs (2008) ; Sakamoto et al. (1993) ; Houghton et al. (2003) ; Fossette et al. (2007) ; Salmon et al. (2004) ; Hays et al. (2004); Southwood et al. (1999)

Note : Profiles A-H, as reported in the literature and compiled by Hochscheid (2014). The depth and time arrows indicate the axis variables, but the figure does not represent true proportions of depths and durations for the various profiles. In other words, the depths can vary greatly, but behavioral activity seems to dictate the shape of the profile. Profiles G and H have only been described for shallow dives (less than 5 m).

Figure F.6-2: Generalized Dive Profiles and Activities Described for Sea Turtles

A similar study conducted by Wildermann et al. (2019) near Crystal River, Florida, observed similar dive activity patterns in maximum dive depth for each species were on average similar during the daytime and nighttime. Maximum dive durations for green and loggerhead sea turtles were longer during the daytime compared to nighttime but Kemp's ridley sea turtle remained similar throughout the day and night. Okuyama et al. (2021) analyzed almost 50,000 dive profiles of tracked leatherback sea turtles and found that leatherbacks migrating across the North Pacific changed their dive behavior regionally, likely in response to local environmental conditions, (e.g., sea surface temperature and prey abundance). They generally conducted deeper dives when sea surface temperature was warmer. The dive patterns are assumed to assist with thermoregulation, saving energy, and finding prey. Dive type classification exhibited a V-shape (see Figure F.6-2 for generalized dive behavior) presumably diving into deep cold waters to avoid overheating. The deepest dive recorded during this study was just over 1,000 m (Okuyama et al., 2021). The study concluded that leatherbacks are able to spend more time foraging in cool-temperature areas (i.e., high latitude areas) due to mesothermy (i.e., use of metabolic heat to raise one's body temperature), but there is a trade-off between energy gain by foraging activity and migration cost, which in turn appears to reduce their reproductive output (Okuyama et al., 2021).

F.6.1.4 Hearing and Vocalization

Information on hearing and vocalization in reptiles is provided in Appendix D (Acoustic and Explosive Impacts Supporting Information), Section D.7-1(Hearing and Vocalization).

F.6.1.5 General Threats

F.6.1.5.1 Water Quality

Sea Turtles

Water quality in sea turtle habitats can be affected by a wide range of anthropogenic activities. The potential for energy exploration and extraction activities to degrade nearshore and offshore habitats are discussed in Section F.6.1.5.2 (Commercial and Recreational Industries). Marine debris in sea turtle habitats is discussed in Section F.6.1.5.6 (Marine Debris). Chemical pollution and impacts on water quality are also of great concern, although its effects are just starting to be explored and understood on marine reptiles (Law et al., 2014; Ortmann et al., 2012). Specific ocean contaminations such as oil and chemical spills have been documented to have damaging effects in most marine reptile species through direct exposure to oil or chemicals as well as impacts on prey availability (McDonald et al., 2017). Ingested plastics, discussed in more detail in Section F.6.1.5.6 (Marine Debris), can also release toxins, such as bisphenol-A (commonly known as "BPA") and phthalates, and organisms may absorb heavy metals from the ocean and release those into tissues (Fukuoka et al., 2016; Teuten et al., 2007). Plastic size, polymer type, shape, and chemical composition of the debris ingested by sea turtles or their prey all influence impacts to these species (Tuuri & Leterme, 2023). Life stage, geographic location relative to concentrations of pollutants, and feeding preference affect the severity of impacts on sea turtles associated with chemical pollution in the marine environment.

Crocodilians

For the American crocodile, the increase in salinity levels from freshwater input reductions may influence community or population distributions in southern Florida (Mazzotti et al., 2016; Mazzotti et al., 2019). One of the goals of the Comprehensive Everglades Restoration Plan is to restore historic freshwater flows through portions of the Everglades. Green et al. (2014) modeled potential effects of restoring freshwater flows to the Everglades, predicting crocodile populations across south Florida would decrease approximately 3 percent after the restoration of historical flows compared to future conditions without restoration. The model also predicted that the freshwater influx would result in local decreases of up to 30 percent in the vicinity of Buttonwood Canal; however, expected local increases of up to 30 percent would result in the Joe Bay area by Taylor Slough (Green et al., 2014).

American alligators are often cited as indicators for water quality, in particular, for heavy metal pollution. Alligators avoid polluted, low-quality waters; therefore, a lack of alligators in one area can be an indicator that the waters there are polluted. Alligators are considered a keystone species within ecosystems. Keystone species are ones that strongly interact with other species, and thus their removal may significantly affect community composition (Brandt et al., 2016; Hodge, 2011; Nilsen et al., 2019). Fluctuations in water levels are a primary driver for alligator presence in inland freshwater systems (Brandt et al., 2016; Hidalgo-Ruz et al., 2012; National Park Service, 2012), along with lower salinities (Gardner et al., 2016; Nifong & Silliman, 2017). The population status and/or behaviors of the alligator can be used to represent the health of the ecosystem as a whole, which can largely assist in monitoring current restoration efforts and understanding of how to guide future environmental policies (Brandt et al., 2016).

Terrapins

Diamondback terrapins are also considered to be an indicator species for water quality (Pfau & Roosenburg, 2010). Terrapins, like alligators, are also commonly found in higher abundances in higher quality waters, even within the same bay system. Although it is unclear how pollutants impact terrapin individuals, populations and their habitats, studies indicate that terrapins uptake pollutants into tissues. Basile et al. (2011) measured fat content in diamondback terrapins for a number of contaminants, including persistent organic pollutants (e.g., polychlorinated biphenyls, polybrominated diphenyl ethers, chlorinated pesticides, and methyl-triclosan). This study was conducted by collecting fat biopsies on terrapins in Barnegat Bay in New Jersey, covering industrial areas and outfalls, as well as less polluted areas of the bay (e.g., Forsythe National Wildlife Refuge). Basile et al. (2011) found that terrapins closer to the industrial area had higher persistent organic pollutants in fat stores than terrapins further from sources of industrial pollution. Male terrapins had higher concentrations of pollutants in fat stores than females, while females had higher concentrations of persistent organic pollutants in plasma than males. Monitoring of chronic, genetic, and reproductive effects of contaminant exposure also will be important in assessing the health of this species in the future, as well as estimating the expected impacts of other potential threats to terrapins and their habitats (Basile et al., 2011).

With respect to other water quality parameters such as dissolved oxygen, a study conducted by Farrell (2021) observed that terrapins in Chesapeake Bay do not prefer either higher or lower dissolved oxygen levels and were observed throughout the middle dissolved oxygen ranges. Low dissolved oxygen levels are an indicator of poor water quality that suggests a lack of primary production occurring in the area, and therefore a lack of organisms and overall productivity in a certain region. In this case, it was concluded that terrapins do not prefer higher water quality over lower water quality, as a function of dissolved oxygen composition (Farrell, 2021).

F.6.1.5.2 Commercial and Recreational Industries

Sea Turtles

In offshore regions of the Study Area, bycatch from commercial fisheries is a primary threat to sea turtles. Population declines of loggerhead and leatherback sea turtles in the Atlantic have been associated with longline fisheries (Swimmer et al., 2017). In U.S. fisheries, Finkbeiner et al. (2011) estimated that bycatch resulted in 71,000 sea turtle deaths per year prior to effective regulations that protect sea turtles (e.g., regulations adopted since the mid-1990s in different U.S. fisheries for sea turtle exclusion devices). Mortality estimates are currently 94 percent lower (4,600 deaths) than pre-regulation estimates (Finkbeiner et al., 2011). One comprehensive study estimates that worldwide, 447,000 sea turtles are killed each year from bycatch in commercial fisheries around the world (Wallace et al., 2010a; Wallace et al., 2010b). Lewison et al. (2014) compared bycatch using three different gear types (longline, gillnet, and trawling nets) for sea turtles, marine mammals, and seabirds. Sea turtles were most susceptible to bycatch, with the Mediterranean and waters off the South American Atlantic coast as the two highest fisheries reporting sea turtle mortalities (primarily through trawling) (Lewison et al., 2014). Offshore energy development, including oil, natural gas extraction, and renewable energy projects in coastal and deep waters on the continental shelf can also effect sea turtles through degradation of habitats during pre-construction, construction, and operational phases (Bergström et al., 2014; Finkbeiner et al., 2011; Wright & Kyhn, 2015).

In nearshore areas, recreational fishing has surpassed commercial fisheries in terms of bycatch of Kemp's ridley and green sea turtles according to a modeling study on juvenile sea turtle bycatch risk between 1996 and 2017 (Putman et al., 2023). In addition, this study found that bycatch rates of Kemp's ridley sea turtles were two orders of magnitude higher than for green sea turtles and is likely attributed to their nearshore distribution where shore-based recreational fishing is concentrated. Kemp's ridley sea turtles are the most abundant sea turtle species in the western Gulf of Mexico, which is also where the shrimping effort is the highest. Although green sea turtles are also abundant in the Gulf of Mexico, they would not spend much time within shrimp fishing areas as their diet is of seagrass and algae (Putman et al., 2023). Aside from risks of bycatch, sea turtles have been observed changing their behavior, avoiding areas or reducing their home range where recreational fisheries occur (Wildermann et al., 2018).

Large-scale commercial exploitation within the nearshore areas also contributes to global decline in marine sea turtle populations. There are approximately 42 countries and territories that allow direct take of sea turtles and collectively take in excess of 42,000 sea turtles per year, the majority of which (greater than 80 percent) are green sea turtles (Humber et al., 2014). Illegal fishing for sea turtles and nest harvesting also continues to be a major cause of sea turtle mortality, both in countries that allow sea turtle take and in countries that outlaw the practice (Lam et al., 2011; Maison et al., 2010). For example, Humber et al. (2014) estimated that in Mexico, 65,000 sea turtles have been illegally harvested since 2000. The authors, however, noted a downward trend of legal and illegal direct takes of sea turtles over the past three decades—citing a greater than 40 percent decline in green sea turtle take since the 1980s, a greater than 60 percent decline in hawksbill and leatherback take, and a greater than 30 percent decline in loggerhead take (Humber et al., 2014).

Increasing offshore energy development activities globally (U.S. Department of Energy, 2022) have likely led to negative consequences for sea turtle populations in the Study Area. The *Deepwater Horizon* spill in 2010, releasing 200 million gallons of crude oil into the Gulf of Mexico (Putman et al., 2015a), is anticipated to have long-term effects that persist for decades (National Marine Fisheries Service, 2011c, 2014). Vulnerability from such an oil spill can impact nesting beaches through toxicity of embryos from oil deposited along the shorelines by wind and currents, thereby increasing potential of developmental deformities and/or successful hatch rate. In the marine environment, oil spills can cause fouling of hatchlings and oceanic juveniles, as well as degradation and contamination of habitat and prey (Wallace et al., 2020). Surveys in 2013 and 2014 showed continued reductions in the number of Kemp's ridley nests since the 2010 spill (Gallaway et al., 2016). High levels of Kemp's ridley sea turtle strandings on the northern Gulf of Mexico beaches are a likely result of the spill and is presumably tied to cold winter/spring temperatures at the time of the spill. These cold temperatures had caused a delay to the nesting season, and resulted in exposure to the spill (Gallaway et al., 2016). It is further suspected that a decline in prey (blue crab and shrimp) requires sea turtles (e.g., Kemp's ridleys) to expend more energy foraging, which may have an adverse impact on their migrations and reproductive output (Gallaway et al., 2016).

Prior to drilling operations, vessel traffic and seismic disturbances through exploration activities can degrade sea turtle coastal and open-ocean foraging habitats. The United States has a federal goal of installing 30 gigawatt of offshore wind power capacity by 2030 with U.S. East Coast states currently having approximately 25 gigawatt of offshore wind projects under various stages of development. As of 2023, wind power growth is planned along the east coast between 2027 and 2040, with most development occurring off the coasts of Massachusetts, New Jersey, Virginia, and North Carolina (S&P Global

Commodity Insights, 2023; Smith et al., 2015). In February 2023, the Interior Department announced offshore wind lease sale for three proposed lease areas in the Gulf of Mexico off the coasts of Galveston, Texas, and Lake Charles, Louisiana (S&P Global Commodity Insights, 2023). Construction of offshore wind energy facilities in the mid-Atlantic is likely to occur in warmer months, and sea turtles will be present during these periods (Williams et al., 2015). Onshore development can also lead to nesting habitat loss or habitat degradation. Construction activities can facilitate erosion or inhibit natural sediment deposition to form beaches. Once facilities are operational, artificial lighting, noise, and other stressors can further degrade nesting habitats (CSA Ocean Sciences Inc., 2023; Seminoff et al., 2015).

Vessel strikes have been identified as an important mortality factor in nearshore sea turtle habitats, worldwide. Increased recreational fishing effort in Florida has shown that mean annual estimates of mortality to sea turtles from vessel strikes are similar to those reported throughout the entire southeastern United States between 2007 and 2016 (Putman et al., 2023). Foley et al. (2019) studied stranded sea turtles in Florida between 1986 and 2014. A third of stranded loggerhead, green, and leatherback sea turtles had a vessel strike injury. Vessel strike injuries were also observed in Kemp's ridley and hawksbill sea turtles, but to a lower extent compared to the other sea turtle species. Scientists in Hawaii reported that 2.5 percent of green sea turtles found dead on the beaches between 1982 and 2003 had been killed by vessel strikes (Chaloupka et al., 2008), and in the Canary Islands, 23 percent of stranded sea turtles showed lesions from vessel strikes or fishing gear (Oros et al., 2005). Denkinger et al. (2013) reports that vessel strikes in the Galapagos Islands were most frequent at foraging sites close to a commercial and tourism port. Barco et al. (2016) found that loggerheads were healthy at the time of vessel strikes, suggesting healthy individuals are not any likely to be injured or killed any more than unhealthy sea turtles.

Crocodilians

American crocodiles and American alligators were widely hunted for their skins from 1920 to 1970, which led to substantial population declines across all parts of the species ranges. Country-specific (e.g., the listing of the American crocodile as endangered in 1979 [44 *Federal Register* 17710] under the ESA) and international trade restrictions, along with the availability of legally obtained skins from other crocodilians, have significantly reduced commercial hunting in recent decades (Brandt et al., 2016; National Park Service, 2012; Thorbjarnarson et al., 2006). Regulated commercial use of captive reared crocodilians has relieved commercial exploitation for wild crocodilians. The American alligator population has expanded greatly throughout its historic range in wetlands of the southeastern United States. The Florida population of the American crocodile has increased, and its distribution has expanded, since it was listed as endangered (Brandt et al., 2016; National Park Service, 2012).

Oil spills that impact freshwater and estuarine habitats will alter important wetland ecological functions, such as removing sediments, nutrients, pesticides, metals, and other pollutants; and provide essential foundations for food chains for wildlife (Corn, 2010), including crocodilians. Oil spills that occur in or wash into these wetlands could reduce prey availability for both the American alligator and the American crocodile. For the American alligator, coastal oil pollution likely has only limited impacts because the highest abundance of alligators are found in inland freshwater systems (Corn, 2010). For American crocodiles, oil spills would have to occur within, or wash into, crocodile habitats in southern Florida for impacts to occur, and would likely be a substantial and persistent inhibiting factor in American crocodile recovery.

Terrapins

Commercial activities that threaten diamondback terrapins include commercial harvesting, bycatch mortality in crab pots, and pollution. Up until the beginning of the 20th century, diamondback terrapins were in great demand by gourmet restaurants and the pet trade in major metropolitan areas of the United States (Pfau & Roosenburg, 2010). Dredging of shallow water habitats and scraping of hibernacula where terrapins congregate during the winter were the most effective forms of commercial harvesting. Commercial harvesting, as determined by test dredging, tended to capture more females than males, which likely severely reduced the reproductive potential for populations in terrapin fisheries. The commercial demand for terrapins generally subsided through the 20th century. However, there was an increase in terrapin exports to China from the United States in the late 1980s, but by 2007, all of the states within the diamondback terrapin range had prohibited commercial harvest of terrapins (Pfau & Roosenburg, 2010).

Roosenburg et al. (1997) studied crab pot use in the Chesapeake Bay and estimated that 15 to 78 percent of the local terrapin population can be captured in crab pots in a single year. Crab pots are designed with small entrances, which tend to capture smaller males rather than larger females. Because of the selective mortality of males in crab pots, Pfau and Roosenburg (2010) estimated that the terrapin sex ratio in the Chesapeake Bay at one male to two, possibly three females. New crab traps with terrapin exclusions have greatly reduced terrapin bycatch (Lester, 2012; Pfau & Roosenburg, 2010; University of Georgia, 2023).

Oil spills in coastal areas directly impact diamondback terrapins by oiling and drowning the animals and indirectly by contaminating their nesting beaches (Pfau & Roosenburg, 2010). The short-term impacts of the oil spill from a leak in an underground oil pipeline near Chalk Point, Maryland, showed direct impacts on adult terrapins and decline in hatchling survivability where the oil leak polluted sand in a nesting location (Michel et al., 2001).

Residential and urban development restricts freshwater flow into swamps and estuaries, which may limit diamondback terrapin growth, survival, and abundance, and potentially impact diamondback terrapin habitats if spills reach estuaries and riverine areas (Basile et al., 2011).

F.6.1.5.3 Disease and Parasites

Fibropapillomatosis is a disease of sea turtles that typically results in the production of tumors, both external and internal, that are considered benign, but may obstruct crucial functions, such as swimming, feeding, sight, and buoyancy, and can lead to death (Balazs, 1986; Patrício et al., 2016). The disease was first noticed in 1936 (Smith & Coates, 1938), but did not gain international attention until the early 1980s, when it was most commonly reported in green sea turtles. Though more frequent among green sea turtles, Fibropapillomatosis has been reported in all species of sea turtles (Patrício et al., 2016). Throughout the 1980’s, rapid spreading of fibropapillomatosis was recorded, quickly becoming an endemic in Florida and Hawaii green sea turtle populations (Day et al., 2016; Work & Balazs, 2013). By 1995 the concentration of disease in the population reached its climax and has showed a decline in prevalence since (Patrício et al., 2016). A study conducted by Kelley et al. (2022) at a coastal foraging site in Florida found that 49 percent of recaptured resident juvenile green sea turtle in the Indian River Lagoon had developed fibropapillomatosis, but also that they were capable of full recoveries. It is important to note, however, that this study was limited to only visible tumors, and had a small (~10 percent) recapture rate.

Edmonds et al. (2016) lists 16 parasites known to occur in sea turtles, with the most common and significant (in terms of impacts on health) being blood flukes and flatworms (Watson et al., 2017). Some of the common external parasites found on sea turtles include leeches and a number of different species that reside on the shell called epibiota (Glandon & Miller, 2016). Leeches and parasitic isopods (sea lice) are usually found on the soft skin tissues around the base of flippers, neck, tail, eyes, and the mouth (Foster & Gilmour, 2016). More recently, sea turtle egg fusariosis is a fungal disease that has become an emerging problem for sea turtle eggs and neonates. Linked to the fungi *Fusarium solani* species complex, this fungal disease causes decreased egg hatch success rate in all 7 extant sea turtle species, while also causing cutaneous and subcutaneous infections, and skin lesions (Kuschke et al., 2023).

The type and severity of disease in crocodilians and terrapins is poorly understood and is not considered as a significant threat to species recovery (Florida Fish and Wildlife Conservation Commission, 2009; Hackney, 2010; National Park Service, 2012; Savannah River Ecology Laboratory & Herpetology Program, 2012).

F.6.1.5.4 Invasive Species

Invasive species have been shown to have both harmful and beneficial impacts on sea turtles. Impacts on sea turtles associated with invasive species primarily concern nest predation and prey base. For example, feral hogs (*Sus scrofa*) have been known to destroy several sea turtle nests during a season on certain nesting beaches in Florida (Engeman et al., 2016) and South Carolina (Engeman et al., 2019). Engeman et al. (2016) noted nesting success after a successful implementation of a feral hog control program in Florida. In foraging grounds, sea turtles have been shown to adapt their foraging preferences for invasive seagrass and algae. Becking et al. (2014) showed green sea turtle foraging behavior shift to consumption of *Halophila stipulacea*, a rapidly spreading seagrass in the Caribbean. In Hawaii, green sea turtles in Kaneohe Bay have modified their diets over several decades to include seven non-native species (Spiny Seaweed, *Acanthophora spicifera; Hypnea musciformis, Gracilaria salicornia, Eucheuma denticulatum,* Graceful Red Weed*, Gracilaria tikvahiae;* Agar-agar*, Kappaphycus striatum;* and Elkhorn Sea Moss, *Kappaphycus alvarezii*), with non-native algae accounting for over 60 percent of sea turtle diet (Russell & Balazs, 2015).

Non-native Burmese pythons (*Python bivitattus*) are large generalist predators that have established an expanding breeding population in Florida (Walters et al., 2016). Introduced pythons present a direct threat to the American alligator and American crocodile through predation, where predation of alligators up to 2 m in length have been reported (Dorcas et al., 2012). Originally thought to be restricted to freshwater habitats in Florida, Hart et al. (2012) has demonstrated salt water tolerance of python neonates, which could increase predation rates to American crocodiles, American alligators, and terrapins. Dorcas et al. (2012) noted that increased python predation on mammalian populations also reduced prey availability for alligators and crocodiles.

Terrapin nests and hatchlings are vulnerable to predation from non-native rats and ants, along with other native terrestrial and avian predators (Draud et al., 2004; Pfau & Roosenburg, 2010). In addition, invasive vegetation can severely impact wetlands when made vulnerable by high amounts of disturbance. *Phragmites australis*, an invasive emergent marsh reed, is rapidly expanding in coastal wetlands of the United States, particularly brackish wetlands, which likely degrades terrapin nesting areas. Cook (2016) found that *Phragmites australis* can alter vegetation structure, soil temperature, and moisture in nesting locations, which may limit preferred nesting habitats (replacing sparsely vegetated sandy locations with thick stands of *Phragmites australis*), potentially skew sex ratios towards males, and reduce nesting success through the encroachment of root systems into nests.

F.6.1.5.5 Climate Change

Sea turtles, crocodilians, and terrapins are particularly susceptible to climate change effects because their life history, physiology, and behavior are extremely sensitive to environmental temperatures (Donaton et al., 2019; Fuentes et al., 2013; Green et al., 2014; Hart & Lee, 2006; Lockley & Eizaguirre, 2021; Patrício et al., 2021; University of Georgia, 2023; Wheatley et al., 2012). Climate change models predict sea level rise and increased intensity of storms and hurricanes in tropical sea turtle nesting areas (Patino-Martinez et al., 2008), as well as coastal areas of the United States where crocodilians and terrapins may nest (Frost et al., 2017). These factors could significantly increase beach inundation and erosion, thus affecting overall quality of sea turtle, crocodilian, and terrapin nesting beaches. Inundation of nests can have detrimental effects to the eggs as vital gas exchange occurs between an egg and its environment through the egg shell, so being impeded by water can cause the egg to drown (Pike et al., 2015).

Climate change may negatively impact reptiles in multiple ways and at all life stages. These impacts may include the potential loss of nesting beaches due to sea level rise and increasingly intense storm surge (Patino-Martinez et al., 2008), shifts in benthic prey community due to warming temperatures (Donaton et al., 2019), feminization of populations from elevated nest temperatures(Jensen et al., 2018; Laloë & Hays, 2023; Patrício et al., 2021; Pfau & Roosenburg, 2010; Reneker & Kamel, 2016), decreased reproductive success (Hawkes et al., 2006; Laloë et al., 2016; Pike, 2014), shifts in reproductive periodicity and latitudinal ranges (Pike, 2014), disruption of hatchling production, dispersal and migration (Montero et al., 2019), and indirect effects to food availability (Donaton et al., 2019; Witt et al., 2010). Short-term effects on aquatic reptiles and their habitat also include the potential impacts caused by increased hurricane occurrence and intensity (Elsey et al., 2006; Elsey & Woodward, 2010). American alligators are less likely to be affected by coastal impacts associated with climate change because they occur in freshwater systems further inland (Eversole et al., 2015).

Adaptation strategies to protect coastal infrastructure are an anticipated response to rising sea levels. These activities may include shoreline stabilization projects and infrastructure hardening, which could contribute to the loss of nesting habitat. Shoreline stabilization may hold beach sediments in place; however, the disruption of onshore currents can reduce the beach replenishment of sediments further away (Boyer et al., 1999; Fish et al., 2008).

F.6.1.5.6 Marine Debris

Marine debris is defined as “any persistent solid material that is manufactured or processed and directly or indirectly, intentionally or unintentionally, disposed of or abandoned into the marine environment or Great Lakes” (33 U.S.C 1956). There are three main types of marine debris that impact wildlife: (1) plastics, (2) derelict fishing gear, and (3) abandoned and derelict vessels (Guertin, 2019). Debris in offshore and inshore waters present ingestion and entanglement risks for sea turtles, crocodilians, and terrapins. Ingestion of marine debris can cause mortality or injury to sea turtles (Wilcox et al., 2018). Plastic is the primary type of debris found in marine and coastal environments. NOAA estimates there are between 20 million and 1.8 billion pieces of plastic along the coastline of the United States and at least 8 million tons of plastic end up in our oceans every year from surface waters to deep-sea sediments (Guertin, 2019).

Plastics are the most common type of marine debris ingested by sea turtles (Schuyler et al., 2014). Life stage and feeding preference affects the likelihood of ingestion. Sea turtles living in oceanic or coastal environments and feeding in the open-ocean or on the seafloor may encounter different types and densities of debris and may therefore have different probabilities of ingesting debris. In 2014, Schuyler et al. (2014) reviewed 37 studies of debris ingestion by sea turtles, showing that young oceanic sea turtles are more likely to ingest debris (particularly plastic), and that green and loggerhead sea turtles were significantly more likely to ingest debris than other sea turtle species. Sea turtle prey can ingest microplastics (commonly associated with cosmetic products) as they have been found in various invertebrate species. Since some of these species are sea turtle prey, sea turtles are in turn ingesting microplastics (Duncan et al., 2018). Marine debris also poses a risk to nesting sea turtles and hatchlings that are moving to/from the water and beaches. Debris can cause obstructions, increase exposure to toxins, and can affect hatchling survival and movement to the ocean (Aguilera et al., 2018). Sea turtles can mistake debris for prey; one study found 37 percent of dead leatherback sea turtles to have ingested various types of plastic (Mrosovsky et al., 2009), and Narazaki et al. (2013) noted an observation of a juvenile or sub-adult loggerhead exhibiting hunting behavior on approach to a plastic bag, possibly mistaking the bag for a jellyfish. Even small amounts of plastic ingestion at all life stages can cause an obstruction in a sea turtle's digestive tract and lead to mortality (Balazs et al., 1994; Bjorndal, 1997). Ingested plastics can release toxins, such as bisphenol-A (commonly known as "BPA") and phthalates, and cause heavy metals from the ocean to absorb into the individual's tissues (Fukuoka et al., 2016; Teuten et al., 2007).

Ribic et al. (2010) documented regional differences in amounts and long-term trends of marine debris (land-based and ocean-based) along the U.S. Atlantic coast, while indexing debris amounts with population growth and fisheries activity. Based on their analysis, Ribic et al. (2010) concluded that the vast majority of marine debris was either land-based (38 percent), general-source debris (42 percent), or ocean-based (20 percent) recreational and commercial sources (Ribic et al., 2010); no items of military origin were differentiated. The inland portions along the southeast Atlantic coast contributed the lowest amounts of debris despite a 19 percent increase in coastal population from 1997 through 2007. The Northeast Atlantic coast also contributed low amounts of marine debris, although the coastal population increased by 8 percent. Most of the marine debris inputs along the U.S. Atlantic coast were sourced from inland portions of the mid-Atlantic. With a 10 percent population increase in the mid-Atlantic, the types of debris included heavy land-based and general-source debris loads. Where fisheries were stable, ocean-based debris either stayed steady or declined.

Because of the limited overlap of crocodilian habitats and marine debris, marine debris as an entanglement or ingestion hazard for the American crocodile and American alligator is not likely as pressing of a concern for crocodilian conservation as it is for many other species populations. There is only one reported mortality of an estuarine crocodile (*Crocodylus porosus*) in Australia entangled by plastic marine debris (Ceccarelli, 2009); however, Platt and Thorbjarnarson et al. (2006) suggested that accidental drowning in monofilament fishing nets was likely a significant source of mortality for American crocodiles in Belize in conservation areas where poaching is not likely to occur. Outside of conservation areas in Belize, the authors found that poaching was a major cause of crocodile deaths, in addition to drownings in derelict and active fishing nets. Terrapin drowning events are most often associated with bycatch in crab pots (Roosenburg et al., 1997) as well as derelict crab traps (Bilkovic et al., 2014); however, marine debris in estuarine environments likely pose an entanglement hazard for diamondback terrapins.

F.6.2 ENDANGERED SPECIES ACT-LISTED SPECIES

F.6.2.1 Green Sea Turtle (*Chelonia mydas*)

F.6.2.1.1 Status and Management

The green sea turtle was first listed under the ESA in 1978. In 2016, NMFS and USFWS reclassified the species into 11 "distinct population segments," which maintains federal protections while providing a more tailored approach for managers to address specific threats facing different populations (see 81 *Federal Register* 20058). The geographic areas that include these distinct population segments are (1) North Atlantic Ocean, (2) Mediterranean Sea, (3) South Atlantic Ocean, (4) Southwest Indian Ocean, (5) North Indian Ocean, (6) East Indian Ocean – West Pacific Ocean, (7) Central West Pacific Ocean, (8) Southwest Pacific Ocean, (9) Central South Pacific Ocean, (10) Central North Pacific Ocean, and (11) East Pacific Ocean.

Only the North Atlantic distinct population segment (which is listed as threatened) is in the Study Area and is discussed further in the document. It should be noted, however, that North Atlantic green sea turtle populations have minimal mixing (gene flow) with the South Atlantic regions and no mixing with the Mediterranean region, and juvenile sea turtles from the North Atlantic may occasionally use South Atlantic or Mediterranean foraging grounds (Seminoff et al., 2015).

Critical habitat is currently designated in the Study Area (63 *Federal Register* 46693) and proposed for designation in three other portions of the Study Area (88 *Federal Register* 46572) (see Section 3.8, Reptiles, Figure 3.8-1 through Figure 3.8-4). In 1998, critical habitat was designated for green sea turtles in coastal waters around Culebra Island, Puerto Rico, from the mean high-water line seaward to 3 NM to include Culebra's outlying Keys (63 *Federal Register* 46693). The essential physical and biological features of this critical habitat include (1) seagrass beds, which provide valuable foraging habitat; (2) coastal waters of Culebra, which serve as a developmental habitat and support juvenile, subadult, and adult green sea turtle populations; and (3) coral reefs and other topographic features that provide shelter (63 *Federal Register* 46693). Puerto Rico's Culebra Island, where the NMFS and USFWS designated critical habitat for green sea turtles, supports important habitat for juveniles, subadults, and a small population of adults. Green sea turtles are most abundant at Culebrita, Mosquito Bay, Puerto Manglar, and Tamarindo Grande, probably due to the presence of dense seagrass beds in those areas (Collazo et al., 1992; Patrício et al., 2016; Patrício et al., 2014).

Higher concentrations and abundance in other locations throughout the green sea turtle range also support dense marine vegetation used as foraging grounds (Patrício et al., 2014; Seminoff et al., 2015). In 2023, NMFS proposed to designate occupied critical habitat, encompassing 1,047,564 km^2 of *Sargassum* habitat and 96,349 km^2 of nearshore waters (from the mean high water line to 20 m depth) in Florida, Texas (from the Mexico border to and including Galveston Bay), North Carolina (from the South Carolina border to but not including Albemarle Sound), and Puerto Rico (Culebra Island, Maunabo, Guayama, and northern Puerto Rico Island, southern Mona Island, eastern and southern Vieques Island) (*88 Federal Register 46572*) (see Section 3.8, Reptiles). The essential features of this proposed critical habitat include (1) *Sargassum* habitat for surface-pelagic foraging and resting; (2) Florida's nearshore waters that contain migratory, and benthic foraging/resting habitat; (3) Texas (from the Mexico border to and including Galveston Bay) and North Carolina's (from the South Carolina border to but not including Albemarle Sound) nearshore waters that contain benthic foraging/resting habitat; and (4) nearshore waters off Maunabo, Guayama, southern Mona Island, and eastern and southern Vieques Island that also contain reproductive essential features.

F.6.2.1.2 Habitat and Geographic Range

The green sea turtle is distributed worldwide across tropical and subtropical coastal waters generally between 45° North and 40° South. After emerging from the nest, green sea turtle hatchlings swim to offshore areas where they float passively in major current systems; however, laboratory and modeling studies suggest that dispersal trajectories might also be shaped by active swimming (Christiansen et al., 2016; Putman & Mansfield, 2015). Post-hatchling green sea turtles forage and develop in floating *Sargassum* habitats of the open-ocean. At the juvenile stage (estimated at five to six years), they leave the open-ocean habitat and retreat to protected lagoons and open coastal areas that are rich in seagrass or marine algae (Bresette et al., 2006), where they will spend most of their lives (Bjorndal & Bolten, 1988). The optimal developmental habitats for late juveniles and foraging habitats for adults are warm shallow waters (3 to 5 m), with abundant submerged aquatic vegetation and close to nearshore reefs or rocky areas (Holloway-Adkins, 2006; Seminoff et al., 2015; Seminoff et al., 2002). Climate change and ocean warming trends may impact the habitat and range of this species over time (Fuentes et al., 2013; Lockley & Eizaguirre, 2021). These impacts apply to all sea turtle species and are discussed in Section F.6.1.5.5 (Climate Change).

Four regions within the North Atlantic distinct population segment support nesting concentrations: Costa Rica (Tortuguero), Mexico (Campeche, Yucatán, and Quintana Roo), the United States (Florida), and Cuba. The highest concentration of nesting is in Tortuguero, and in Mexico, where nesting occurs primarily along the Yucatán Peninsula. Most green sea turtle nesting occurs along the Atlantic coast of eastern central Florida, with smaller concentrations along the Gulf Coast and Florida Keys (Naval Undersea Warfare Center Division Newport, 2023). In Cuba, nesting primarily occurs on the extreme western tip of the country and on islands off the southern shore of Cuba. Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras, Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and United States (North Carolina, South Carolina, Georgia, Texas, and Virginia).

Green sea turtles are known to live in the open-ocean waters of the Gulf Stream and North Atlantic Gyre during the first five to six years of life. Habitat use, movement patterns, and general distribution during this life phase was previously not well known. A tracking study conducted by Mansfield et al. (2021) recorded oceanic-stage juvenile green sea turtles (less than 1 year old). The tagged green sea turtles were observed within continental shelf waters for only 93 out of 1,379 days, the remainder of the time, the sea turtles spent in deeper, offshore waters. Only 1 of 21 tagged sea turtles remained in the continental shelf the entire time. Turtles were typically tracked within and departing over time, from the Gulf Stream and Gyre currents and moving south of or near Cape Hatteras, North Carolina (Mansfield et al., 2021). Juvenile green sea turtles have the ability to migrate independently of ocean currents (directional and active swimming) to access productive foraging grounds (Christiansen et al., 2016; Putman & Mansfield, 2015; Ribic et al., 2010). Tracking studies by Sloan et al. (2022) and Lamont et al. (2023) studied movement of green sea turtles from nesting locations in the Gulf of Mexico to record their inter-nesting and migration patterns. Movement of tracked sea turtles overlapped with 23 Marine Protected Areas (Lamont et al., 2023) with migration patterns recorded between U.S. nesting sites and foraging areas in Mexico.

Green sea turtles start to move into mid-Atlantic foraging grounds in late spring and early summer (Barco et al., 2018b). Green sea turtles have also been sighted (in low numbers) in the past within Chesapeake Bay and coastal Virginia waters during spring, summer, and fall but surveys have shown in increase in numbers over the past twenty years (Barco et al., 2018b). Per stranding and sighting data, green sea turtles typically do not occur north of Cape Cod as there was only one sighting in July of 2017 in Cape Cod Bay (Sea Turtle Sightings Hotline, 2023).

As ocean temperatures increase in the spring, juvenile and adult green sea turtles migrate from southeastern U.S. waters to the estuarine habitats of Long Island Sound, Peconic Bay, Chesapeake Bay, and possibly Nantucket Sound, where an abundance of algae and eelgrass occurs. Peak occurrence in the Northeast U.S. Continental Shelf Large Marine Ecosystem is likely in September (Berry et al., 2000). During nonbreeding periods, adult and juvenile distributions may overlap in coastal feeding areas (Hirth, 1997; Weishampel et al., 2006).

Juveniles use the estuarine and nearshore waters of the panhandle of Florida throughout the year, including Indian River Lagoon, Pensacola Bay, St. Joseph Bay, Charlotte Harbor, Cedar Keys, Homosassa Springs, Crystal River, Tampa Bay, and St. Andrews Bay (Lamont et al., 2015; Lamont & Iverson, 2018; Langhamer et al., 2016; Renaud et al., 1995; Seminoff et al., 2015). Wildermann et al. (2019) studied habitat use and behavior of multiple sea turtle species, including green sea turtles in the Gulf of Mexico and identified foraging hotspots along the nearshore and coastal habitats of eastern and northern portion of the Gulf of Mexico. In the northern Gulf of Mexico, green sea turtles prefer the coastal habitats of southern Texas (e.g., lagoons, channels, inlets, bays) where seagrass beds and macroalgae are abundant, including Texas' Laguna Madre (Renaud et al., 1995; Wildermann et al., 2019).

As water temperatures rise from April to June, green sea turtle numbers increase in the continental shelf waters of the Gulf of Mexico Large Marine Ecosystem, off Galveston Bay, and in those waters associated with the continental shelf break northeast of Corpus Christi. Green sea turtles found in these deeper waters have been documented migrating from resident foraging grounds to distant nesting grounds (Lamont et al., 2023; Meylan, 1995; Sloan et al., 2022). The sparse sighting records in Louisiana and Texas waters, as well as nesting records on the southern Texas coast, indicate that green sea turtles are found in the northwestern Gulf of Mexico during spring but in far fewer numbers than in the northeastern Gulf.

F.6.2.1.3 Population Trends

Green sea turtle nesting has shown an exponential increase over the past 29 years, with nests reported along the Florida panhandle, Florida Gulf Coast, Florida Atlantic coast, Georgia, Alabama, South Carolina, North Carolina, southeastern Virginia, and Texas, along with the wider Caribbean, Yucatán Coast of Mexico, Suriname, and Isla Trindade (Brazil) (Florida Fish and Wildlife Conservation Commission, 2018; Seminoff et al., 2015; Shaver et al., 2020b; Virginia Department of Wildlife Resources, 2024). A green sea turtle nested at Cape Henlopen State Park in Delaware in August 2011, which was the first green sea turtle nesting ever observed north of Virginia (Murray, 2011). Data on green sea turtle nesting along the northwestern Gulf of Mexico had previously been limited but a study along the Texas coast from 1987 through 2019 documented 111 confirmed green sea turtle nests (Shaver et al., 2020b).

The Marine Turtle Specialist Group (under the International Union for Conservation of Nature's Species Survival Commission) conducted a worldwide analysis of the green sea turtle population based on 32 index nesting sites around the world (Seminoff & Marine Turtle Specialist Group Green Turtle Task Force, 2004). The analysis concluded there has been a 48 to 65 percent decline in the number of females nesting annually over the past 100 to 150 years. About 80 percent of nesting in the Western Atlantic Ocean occurs at Tortuguero, Costa Rica (Seminoff et al., 2015).

Generally, nesting trends in the Western Atlantic Ocean are stable to increasing and are increasing in Florida, as shown by annual total nest counts for green sea turtles on Florida's index beaches (27 out of 229 nesting beaches selected to monitor long-term nesting trends). In 2023, green sea turtle nest counts recorded on the 27 core index beaches reached more than 61,000 nests. Nest count data shows a mostly biennial pattern of fluctuation with record highs set in 2011, 2013, 2015, 2017, 2019, and 2023. Since standardized nest counts began in 1989, green turtle nest counts have increased 120-fold. (Florida Fish and Wildlife Conservation Commission, 2024).

Although these data appear to present an encouraging global outlook, Casale and Ceriani (2020) found that current methods for estimating population abundance using nesting data may greatly overestimate the abundance of sea turtle populations, especially in situations with a low detection probability, including temporary emigration. They conducted a study using 15-year beach monitoring data to simulate a virtual population of adult females. Some recommendations that came out of the study include but are not limited to: using datasets with at least 20 remigration intervals (including multiple values from the same sea turtle); interpreting models used with caution as very large errors are possible; and having a consistent and complete spatio-temporal coverage of the nesting activity in a nesting area (Casale & Ceriani, 2020).

A recent Navy-funded study provided the first broad scale models of sea turtle in-water abundance, density, and distribution for the four most common sea turtle species in the U.S. East Coast region in over a decade. For green sea turtles, this is the first time that a density spatial model has been published for this region. The green sea turtle model predicted a mean annual abundance of 64,674 individuals with monthly predicted abundance ranging from a high of 96,935 in July to a low of 49,720 in January. Abundance was generally higher in warmer months (June–August) and lower in non-summer months due to the animal's predicted preference for warm, shallow, productive waters. Overall, predicted density was high off the coasts of Georgia and Florida year-round. Green sea turtles were predicted to occur in the mid-Atlantic from May to October when waters are warmer, generally from the Chesapeake Bay to Long Island. Green sea turtles were predicted to move south again, starting in October (DiMatteo et al., 2024).

NOAA Southeast Fisheries Science Center (Garrison et al., 2023) also developed density spatial models to estimate sea turtle in-water abundance, density, and distribution for the four most common sea turtle species in the Gulf of Mexico. The model predictions were limited to the continental shelf waters and based on aerial surveys that were conducted in 2011-2012 and 2017-2018. Average monthly predictions were generated for the period 2015-2019. Monthly average abundance ranged from 1,909 green turtles in January to 10,159 green turtles in July. Overall, green turtle density was predicted to be highest in waters close to shore and in warmer waters in the eastern region of the Gulf (Garrison et al., 2023; Rappucci et al., 2023).

F.6.2.1.4 Predator and Prey Interactions

While primarily herbivorous, a green sea turtle's diet changes substantially throughout its life, varying by lifestage (hatching, post-hatching, juvenile, sub-adult, adult). Salmon et al. (2004) reported that post-hatchling green sea turtles were found to feed near the surface on seagrasses or at shallow depths on comb jellies and unidentified gelatinous eggs off the coast of southeastern Florida. Very young green sea turtles are omnivorous (Bjorndal, 1997). Long-term assessments of green sea turtle diets between 1987 and 2014 along the Texas coast found that small (16.2 cm straight carapace length), oceanic life stages consume primarily macroalgae along the nearshore whereas larger (30 cm straight carapace length) green sea turtles present within the nearshore waters forage on seagrasses (Howell & Shaver, 2021). Nagaoka et al. (2012) analyzed 50 incidentally caught juvenile green sea turtles in Brazil and determined that juveniles consumed an omnivorous diet, including terrestrial plants (floating in the water), algae, invertebrates, and seagrass. Black mangrove leaves were of the greatest importance to diet at this location (adjacent to a black mangrove forest). Sampson and Giraldo (2014) observed opportunistic foraging of tunicates (a type of filter-feeding marine invertebrate) by green sea turtles in the eastern tropical Pacific. Pelagic juveniles smaller than 8 to 10 in. in length eat worms, young crustaceans, aquatic insects, grasses, and algae (Bjorndal, 1997). After settling in coastal juvenile developmental habitat at 8 to 10 in. in length, they eat mostly mangrove leaves, seagrass, and algae (Balazs et al., 1994; Nagaoka et al., 2012). Research indicates that green sea turtles in the open-ocean environment, and even in coastal waters, also consume jellyfish, sponges, and sea pens (Hatase et al., 2006; Seminoff et al., 2015). Fukuoka et al. (2016) also noted that juvenile green sea turtles were at higher risk to marine debris ingestion, likely due to the resemblance of small pieces of debris to omnivorous dietary items. The green sea turtle is the only species of sea turtle that, as an adult, primarily consumes plants and other types of vegetation (Mortimer, 1995; Nagaoka et al., 2012).

The loss of eggs to land-based predators such as mammals, snakes, crabs, and ants occurs on most nesting beaches. As with other sea turtles, hatchlings may be preyed on by birds and fish. Sharks are the primary nonhuman predators of juvenile and adult green sea turtles at sea (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1991; Seminoff et al., 2015). Within the Northwest Atlantic Ocean and Gulf of Mexico, tiger sharks preyed predominantly on green sea turtles with evidence of bull, dusky, and great white shark attacks also recorded (Aoki et al., 2023).

F.6.2.1.5 Species-Specific Threats

In addition to the general threats described previously in Section F.6.1.5 (General Threats), damage to seagrass beds and declines in seagrass distribution can reduce foraging habitat for green sea turtles as well as noted decreases in mass and size of green sea turtles (Meylan et al., 2022; National Marine

Fisheries Service & U.S. Fish and Wildlife Service, 1991; Seminoff et al., 2015). Green sea turtles are susceptible to the disease fibropapillomatosis, which causes tumor-like growths (fibropapillomas). These tumors result in reduced vision, disorientation, blindness, physical obstruction to swimming and feeding, increased susceptibility to parasites, and increased susceptibility to entanglement (Balazs, 1986; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1991; Patrício et al., 2016; Work & Balazs, 2013). Some populations (e.g., the Florida population) have begun to show resistance to the disease (Kelley et al., 2022), but it remains an issue for others, such as Pacific populations, and Hawaii's green sea turtles in particular (Chaloupka et al., 2009; Seminoff et al., 2015). Patrício et al. (2016) noted that fibropapillomatosis recovery was likely in a resident population in Puerto Rico, with tumor regression occurring within three years of formation. Other factors, such as increased stressors and selection of healthy sea turtles during illegal poaching activities may increase susceptibility of sea turtles (Patrício et al., 2016).

F.6.2.2 Hawksbill Sea Turtle (*Eretmochelys imbricata*)

F.6.2.2.1 Status and Management

The hawksbill sea turtle is listed as endangered under the ESA (35 *Federal Register* 8491). While the current listing as a single global population remains valid, data may support separating populations at least by ocean basin under the distinct population segment policy (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2007a). The most recent status review document was released in 2013 by NMFS and USFWS (National Marine Fisheries Service, 2013a).

Critical habitat for hawksbill terrestrial nesting areas was designated in Puerto Rico in 1982. This designation includes portions of Mona Island, Culebra Island, Cayo Norte, and Island Culebrita, from the mean high tide line to a point 150 meters (m) from shore. Critical marine habitat was also designated in 1998 for the coastal waters surrounding Mona and Monito Islands, Puerto Rico, from the mean high water line seaward to 3 NM (see Section 3.8, Reptiles, Figure 3.8-5) (National Marine Fisheries Service, 2013a). Key components in making this location a critical habitat include (1) coral reefs for food and shelter and (2) nesting beaches. The essential physical and biological features of coral reefs support a large, long-term juvenile hawksbill population, in addition to subadults and adults. The types of sponges that hawksbills prefer for food are found on the reefs around these islands. Reef ledges and caves also provide resting areas and protection from predators. Nesting beaches on Mona Island support the largest population of nesting hawksbill sea turtles in the U.S. Caribbean (National Marine Fisheries Service, 2013a).

F.6.2.2.2 Habitat and Geographic Range

The hawksbill is the most tropical of the world's sea turtles, with its range in western North Atlantic also extending into subtropical areas of the Gulf of Mexico and Atlantic coasts (Avens et al., 2021). While hawksbills are known to occasionally migrate long distances in the open ocean, they are primarily found in coastal habitats and use nearshore areas more exclusively than other sea turtles. Adults can be associated with the same foraging sites for 10 years at a time, and migrate shorter distances to breeding locations than other species (Valverde & Holzwart, 2017). Hatchlings in the Study Area are believed to occupy open-ocean waters, associating themselves with surface algal mats in the Atlantic Ocean (Parker, 1995; Witherington & Hirama, 2006; Witzell, 1983). Juveniles leave the open-ocean habitat after 1 to 3 years and settle in coastal foraging areas, typically coral reefs but occasionally seagrass beds, algal beds, mangrove bays, and creeks (Avens et al., 2021; Mortimer & Donnelly, 2008). Hawksbill distribution in the mainland United States is primarily obtained through stranding records of individual hawksbills washing ashore. From these, hawksbills have regularly been observed along the coasts of Texas and Florida and to a lesser extent along other Gulf of Mexico and Atlantic states (Avens et al., 2021; Gorham et al., 2014). In Florida, hawksbills regularly occur in the nearshore waters off the southeastern coast, in the Florida Keys (including the Marquesas and Dry Tortugas). Juvenile hawksbills have been observed along the jetties near Port Aransas, Texas and within the coral reefs at the Flower Garden Banks National Marine Sanctuary in the western Gulf of Mexico (Avens et al., 2021).

Less is known about the hawksbill's oceanic stage, but it is thought that neonates live in the oceanic zone where water depths are greater than 200 m. Distribution in the oceanic zone may be influenced by surface gyres (Leon & Bjorndal, 2002; National Marine Fisheries Service, 2013a).

Foraging and reproductive aggregations of hawksbills are extensive in Puerto Rico, U.S. Virgin Islands, and the Caribbean Sea (Avens et al., 2021). Juveniles and adults share the same foraging areas, including tropical nearshore waters associated with coral reefs, hard bottoms, or estuaries with mangroves (Musick & Limpus, 1997). In nearshore habitats, resting areas for late juvenile and adult hawksbills are typically in deeper waters, such as sandy bottoms at the base of a reef flat (Houghton et al., 2003). As they mature into adults, hawksbills move to deeper habitats and may forage to depths greater than 90 m. During this stage, hawksbills are seldom found in waters beyond the continental or insular shelf unless they are in transit between distant foraging and nesting grounds (Renaud et al., 1996). Ledges and caves of coral reefs provide shelter for resting hawksbills during both day and night, where an individual often inhabits the same resting spot. Hawksbills are also found around rocky outcrops and high-energy shoals, where sponges are abundant, and in mangrove-fringed bays and estuaries. Female hawksbills return to their natal beach every 2 to 3 years to nest at night, every 14 to 16 days during the nesting season (Renaud et al., 1996).

In the Caribbean Sea and Gulf of Mexico Large Marine Ecosystems, the principal nesting season is from June to November (Hillis, 1990), with only rare nesting activity in Florida, which is restricted to Volusia, Martin, Palm Beach, Broward, Miami-Dade, and Monroe Counties per genetic testing (Meylan et al., 2006; National Marine Fisheries Service, 2013a). Throughout their range, hawksbill sea turtles typically nest in low densities; aggregations of nesting activity that usually include approximately 20 nests, but can exceed a few hundred nests in some locations (National Marine Fisheries Service, 2013a). These locations with up to 100 nests include Mona Island, Puerto Rico, and Buck Island Reef off St. Croix.

The greatest hawksbill sea turtle numbers in the southeastern United States are found off the coast of southern Florida, but their typical patterns of habitat are not well understood (Wood et al., 2017a). There, hawksbills are documented from winter to summer from Palm Beach, Broward, and Dade Counties to the Florida Keys, and to coastal waters just northwest of Tampa Bay, where the northernmost stranding records typically occur. Foraging juveniles and adults settle on coral reef and hard-bottom habitats off southern Florida throughout the year (Musick & Limpus, 1997). Hawksbill sea turtle sightings in waters off the Florida panhandle, Alabama, Mississippi, Louisiana, and Texas (Rester & Condrey, 1996; Witzell, 1983), though rare, are likely of early juveniles born on nesting beaches in Mexico that have drifted north with the dominant currents (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1993).

F.6.2.2.3 Population Trends

Within the continental United States, nesting is rare in Florida with up to five nests recorded per year from 1979 to 2019 (Avens et al., 2021). An estimated 22,004 to 29,035 hawksbill sea turtles nest each year at sites among the Atlantic, Indian, and Pacific oceans; of these individuals, 3,626 to 6,108 nests occur among the Atlantic nesting sites alone. Historical population trends showed overall declines for the 20- to 100-year period of evaluation (National Marine Fisheries Service, 2013a). The last 5-year status review for hawksbill sea turtles states that among the 88 sites worldwide for which historic trends could be assessed, 63 (72 percent) showed a decline. Shorter-term population trends, however, show more increases at some nesting sites, particularly in the North Atlantic and Pacific Oceans with 10 (24 percent) increasing, 3 (7 percent) stable, and 28 (68 percent) decreasing (National Marine Fisheries Service, 2013a). Gulick et al. (2022) looked at annual counts of hawksbill females and nesting trends at Buck Island Reef National Monument, St. Croix, and U.S. Virgin Islands from 1998 to 2017. Overall, female abundance was found to have stabilized, but nest abundance declined. New females were recorded recruiting to the area between 1988 and 2017, with remigrants recorded from 2007 and 2017 and counts stabilizing. Temporal trends were not observed in annual mean hatch success, emergence success, and hatchling production during the survey period, but declines in clutch size and female body size were detected (Gulick et al., 2022).

F.6.2.2.4 Predator and Prey Interactions

Hawksbill sea turtles have a varying diet and feeding habitat preference throughout different life stages. Post-hatchling hawksbills feed on floating habitats (e.g., *Sargassum*) in the open ocean (Bresette et al., 1998; Plotkin & Amos, 1998; Van Houtan et al., 2016). During the later juvenile stage, hawksbills are considered omnivorous, feeding on sponges, sea squirts, algae, molluscs, crustaceans, jellyfish, and other aquatic invertebrates (Bjorndal, 1997). Older juveniles and adults are more specialized, feeding primarily on sponges, which compose as much as 95 percent of their diet in some locations (Meylan, 1988; Witzell, 1983). As adults, hawksbill sea turtles fill a unique ecological niche in marine and coastal ecosystems, supporting the natural functions of coral reefs by keeping sponge populations in check (Hill, 1998; Leon & Bjorndal, 2002; Wood et al., 2017b). Feeding on sponges helps to control populations of sponges that may otherwise compete for space with reef-building corals (Hill, 1998; Leon & Bjorndal, 2002).

The loss of hawksbill eggs to predators such as rats, snakes, crabs, and ants is a severe problem on some nesting beaches within the United States. Outside of the U.S., predators such as mongoose and feral pigs can pose as a threat to the eggs. As with other sea turtles, hatchlings may be preyed on by birds and fish. Sharks are the primary nonhuman predators of juvenile and adult hawksbills at sea (National Ocean Service, 2016; Southern California Marine Institute, 2016).

F.6.2.2.5 Species-Specific Threats

In addition to the general threats described in Section F.6.1.5 (General Threats), the greatest threat to hawksbills is harvest for commercial and subsistence use. Direct harvest of eggs and nesting adult females from beaches, as well as direct hunting of sea turtles in foraging areas, continues in many countries. They have been overexploited for centuries, mostly for their prized shell (often called tortoiseshell) (State of the World's Sea Turtles, 2022). International trade of tortoiseshell is thought to be the most important factor endangering the species worldwide. The second-most significant threat to hawksbill sea turtles is loss of nesting habitat caused by the expansion of human populations in coastal areas of the world, as well as the increased destruction or modification of coastal ecosystems to support tourism (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1998). Coastal pollution as a result of increased development degrades water quality, particularly coral reefs, which are primary foraging areas for hawksbills. Due to their preference for nearshore areas, hawksbills are particularly susceptible to nearshore fisheries gear such as drift nets, entanglement in gill nets, and capture on fish hooks of fishermen (National Marine Fisheries Service, 2013a; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1993).

F.6.2.3 Kemp’s Ridley Sea Turtle (*Lepidochelys kempii*)

F.6.2.3.1 Status and Management

The Kemp’s ridley sea turtle is listed as a single population and is classified as endangered under the ESA (35 *Federal Register* 18319). The most recent status review was released in 2015 by the USFWS and

NMFS (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2015). There is no critical habitat currently designated for this species. In 2010, the USFWS and NMFS received a petition to designate critical habitat on nesting beaches in Texas and along Gulf Coast states. The petition is still under consideration, and no proposed rule on the establishment of critical habitat has been released by either agency.

F.6.2.3.2 Habitat and Geographic Range

Kemp's ridley sea turtle nesting is essentially limited to the beaches of the western Gulf of Mexico, primarily in Tamaulipas, Mexico. Nesting also occurs in Veracruz, and a few historical records exist for Campeche, Mexico. Since 1978, the U.S. National Park Service, in partnership with USFWS, NMFS, Texas Parks and Wildlife Department, and the Instituto Nacional de Pesca (a Mexican federal agency), has led an effort to increase Kemp's ridley sea turtle nesting at Padre Island National Seashore, south Texas, to form a secondary nesting colony to safeguard against extinction (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2011). Occasional nesting has been reported from Florida, Alabama, Georgia, South Carolina, North Carolina, and Virginia (in 2012 and 2014) (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2015) with the furthest north nesting occurring in New York where 96 sea turtles were observed on Rockaway Peninsula in Queens, New York (Phorn, 2018).

Habitats frequently used by Kemp's ridley sea turtles in U.S. waters are warm-temperate to subtropical sounds, bays, estuaries, tidal passes, shipping channels, and beachfront waters, where their preferred food, the blue crab, is abundant (Lutcavage & Musick, 1985). The general migration pattern of females begins with travel through relatively shallow migratory corridors toward the nesting beach in the late winter in order to arrive at the nesting beach by early spring. Males and females can loop along the U.S. continental shelf large marine ecosystem in the spring, and back down the southeast U.S. continental shelf in the fall. From nesting beaches in the Gulf of Mexico, the migratory corridor traverses neritic areas of the Mexico and U.S. Gulf coasts with a mean water depth of 26 m approximately 20 kilometers (km) from the coast, occurring in late May through August with a peak in June (Shaver et al., 2016). Kemp's ridley sea turtles that headed north and east traveled as far as the waters off southwest Florida; however, waters off the upper Texas coast through Mississippi, especially off Louisiana, appear to be a "hotspot" as turtles returned to the area to forage over multiple years (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2015).

Evidence suggests that post-hatchling and small juvenile Kemp's ridley sea turtles, similar to loggerhead and green sea turtles of the same region, forage and develop in floating *Sargassum* habitats of the North Atlantic Ocean. Juveniles migrate to habitats along the U.S. Atlantic continental shelf from Florida to New England (Morreale & Standora, 1998; Peña, 2006) at around 2 years of age. Migrating juvenile Kemp's ridleys travel along coastal corridors generally shallower than 50 m in bottom depth (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2011). A Navy-funded study conducted in the mid-Atlantic region indicated that juvenile Kemp's ridley sea turtles utilize the lower to middle Chesapeake Bay in the spring and summer, similar to loggerheads that were also tagged for this study. Kemp's ridley sea turtles preferred to spend more time and forage in shallower waters closer to shore, such as small inlets, embayments, and flats close to the shore in the main stem of the Chesapeake Bay (Barco et al., 2017; Barco et al., 2018b; DiMatteo et al., 2021; DiMatteo et al., 2022). Suitable developmental habitats are seagrass beds and mud bottoms in waters of less than 10 m bottom depth and with sea surface temperatures between 72 and 90°F (22 and 32°C) (Coyne et al., 2000).

In the Gulf of Mexico, juveniles make seasonal east, west, and south migrations and move further offshore during the winter when water temperature drops (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2015). Important year-round developmental habitats in the northern Gulf of

Mexico include the western coast of Florida (particularly the Cedar Keys area), the eastern coast of Alabama, and the mouth of the Mississippi River (Lazell, 1980; Lutcavage & Musick, 1985; Weber, 1995). Coastal waters off western Louisiana and eastern Texas also provide adequate habitats for bottom feeding. A tagging study conducted between 1991 and 2013 within the northwestern Gulf of Mexico found that small benthic juveniles (ages 2 to 4 years) made up approximately 70 percent of the nearshore Kemp's ridley assemblage, indicating important foraging habitat within nearshore waters (Metz & Landry, 2016). Verkaik et al. (2016) found strong site fidelity within and between years to the Mississippi Sound during spring, summer, and fall for juvenile Kemp's ridley sea turtles. During the winter, sea turtles migrated to the nearshore waters of Louisiana. Foraging areas were identified along the Atlantic coasts of Florida and Georgia that had not been previously identified (Gredzens & Shaver, 2020). Satellite tracking of sea turtles identified four primary hotspots used for migration, which include western Gulf of Mexico, around the Yucatán Peninsula, a northern route following the Loop Current, and a southern hotspot towards Gorda Bank, Central America. Adult Kemp's ridley breeding females were found to stay inside the Gulf of Mexico, moving from primary nesting beaches in Rancho Nuevo and central Veracruz, Mexico to Louisiana, the southern Florida shelf and Yucatán Peninsula to forage and reside after their reproductive season (Cuevas et al., 2022). Lamont and Iverson (2018) observed seasonal movements triggered by colder winter temperatures where Kemp's ridleys were tracked leaving the bays in the Gulf of Mexico. Tracking data collected between 1989 and 2013 for Kemp's ridley and loggerhead turtles occurring in the Gulf of Mexico, found that co-occurrence of the two species was largely due to spatial overlap in foraging areas, with Kemp's ridley home ranges being up to 10 times larger than loggerheads (Hart et al., 2018).

Inter-nesting habitat occurs in a narrow band of nearshore western Gulf of Mexico waters in the United States and Mexico, within mean water depths of 14 to 19 m within a mean distance to shore of 6 to 11 km (Shaver et al., 2017). Up to half of all adult female Kemp's ridleys occupy this habitat for weeks to months during each nesting season.

Studies conducted by the National Park Service previously indicated that adult male Kemp's ridley sea turtles remain close to nesting beaches year-round. A tracking study conducted since 2006 at Padre Island National Seashore in Texas found that five of seven adult male Kemp's ridley sea turtles remained near the nesting beaches of the Texas coast for the duration of their tracking period (National Park Service, 2023).

F.6.2.3.3 Population Trends

The earliest estimate of population size was derived from analyzing archival film footage of a large arribada (mass nesting) event in 1947 and other life history information of the Kemp's ridley sea turtle. From these data sources and the analysis of the raw footage, Gonzalez (2011) suggest that the Kemp's ridley population during and prior to the 1947 nesting season was relatively robust, with the estimated number of nests exceeding 121,000. Gallaway et al. (2013) created a stock assessment model that incorporated several factors such as mortality estimates and growth parameters for tagged and recaptured sea turtles. The model estimated the total population to be 248,307 in 2012 for Kemp's ridleys age 2 and older. The lowest point in the decline of Kemp's ridleys occurred in 1985 (approximately 700 nests), representing a 99 percent decline in the number of nests compared to the 1947 estimate. Although the Kemp's ridley population has shown increases since 1985, the rate of recovery has declined in recent years. In 2010, Kemp's ridley nesting showed a steep decline (35 percent) followed by some recovery to 2009 levels, with other declines in 2013 and 2014 (Caillouet et al., 2016; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2015; Shaver et al., 2016). As of 2023, there are an estimated 5,500 females nesting in Mexico annually and approximately 55 females nesting in Texas annually (Williams, 2023). The number of Kemp's ridley sea turtle nests counted along Texas beaches has increased from 2015 (159 nests) to 2016 (186 nests) and 2017 (353 nests) (Shaver, 2018). Per a count in June of 2023, 222 Kemp's ridley nests were found on North Padre Island and the Padre Island National Seashore (Williams, 2023). Habitat protection has led to an increase in sea turtle nesting on the Texas coast since 1980 (Shaver et al., 2020a).

Subadult and adult females were presumed to have suffered a high mortality rate in 2009, which has manifested in a 40 percent decline in nesting activity in Mexico and Texas. The cause of this mortality has been found to be due to cold stunning, which is fairly common for sea turtles to undergo on Texas and Mexico coastlines. Cold stunning occurs as a result of prolonged exposure to cold waters, and sea turtles can experience debilitating conditions which could possibly lead to death. At lower latitudes, a mild shift in temperature during winter can cause cold stunning; at higher latitudes, it is most often seen that sea turtles do not begin migrating south in time, before the onset of late autumn storms. In both areas, however, cold stunning is a result of a sea turtle being caught unexpectedly in lower than expected temperatures (Griffin et al., 2019). The *Deepwater Horizon* oil spill of 2010 coincided with large numbers of Kemp's ridley sea turtles stranded on northern Gulf of Mexico, especially in Alabama, Mississippi, and Louisiana beaches, which further impacted numbers of nests to increase from 2009 levels. Of note, was a tracked female Kemp's ridley that had nested in 2011 but was tracked to foraging grounds where it remained for 4 years and as of 2016 had not returned to the nesting grounds (Gallaway et al., 2016).

In-water abundance, density, and distribution patterns of Kemp's ridley sea turtles were estimated in a recent Navy-funded study conducted along the east coast (DiMatteo et al., 2024). The Kemp's ridley model predicted a monthly abundance ranging from a high of 13,220 in October to a low of 8,341 in August. Animals were predicted to be in the mid-Atlantic from May until November, generally from the Chesapeake Bay north to Delaware Bay, and as far north as Long Island Sound in summer months. There were no sightings of Kemp's ridley turtles north of Cape Cod, where strandings of that species are rare. A very low density, though not zero, was predicted off the continental shelf. There were no sightings off the shelf, though survey effort was limited and occurred in June–September only (DiMatteo et al., 2024).

NOAA Southeast Fisheries Science Center (Garrison et al., 2023; Rappucci et al., 2023) also developed density spatial models to estimate sea turtle in-water abundance, density and distribution for the four most common sea turtle species in the Gulf of Mexico, which included the Kemp's ridley sea turtle. The model predictions were limited to the continental shelf waters and based on aerial surveys that were conducted in 2011-2012 and 2017-2018. Average monthly predictions were generated for the period 2015-2019 (Garrison et al., 2023). Kemp's ridley sea turtles were observed throughout the entire survey range in the Gulf of Mexico and throughout all seasons. Results indicated that Kemp's ridley sea turtle density was highest at intermediate water depths, declining rapidly in waters greater than 15 m deep. The model predicted monthly average abundance of Kemp's ridley sea turtles within the Gulf of Mexico as low as 48,398 in May and as high as 273,633 Kemp's ridley sea turtles in February (Garrison et al., 2023).

F.6.2.3.4 Predator and Prey Interactions

Kemp's ridley sea turtles feed primarily on crabs but are also known to prey on molluscs, shrimp, fish, jellyfish, and plant material (Frick et al., 1999; Marquez, 1994; Seney, 2016). Plant material, primarily macroalgae, is likely consumed incidentally with invertebrate prey items (Seney, 2016). Blue crabs and spider crabs are important prey species for the Kemp's ridley (Keinath et al., 1987; Lutcavage & Musick, 1985; Seney, 2016). They may also feed on shrimp fishery bycatch (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 1993), and Servis et al. (2015) noted instances of fish and horseshoe crab predation, indicating that Kemp's ridley sea turtles may opportunistically feed to supplement their diet. Kemp's ridley diets also varies by region within the Gulf of Mexico and western North Atlantic. Diets for Kemp's ridley in the northern Gulf of Mexico, North Carolina, and Virginia were dominated by invertebrates. Diets in the western Gulf of Mexico and eastern Gulf of Mexico were more evenly divided between invertebrates and fish or invertebrates and macroalgae/seagrass (Ramirez et al., 2020).

Major predators of Kemp's ridley sea turtle eggs and hatchlings on nesting beaches include raccoons, dogs, feral pigs, skunks, badgers, and fire ants. Predatory fishes such as jackfish and redfish may feed on hatchlings at sea. Sharks are the primary predator of juvenile and adult Kemp's ridley sea turtles (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2011). Aoki et al. (2023) documented tiger sharks, great white sharks, and sandbar sharks attacking Kemp's ridley sea turtles in the Gulf of Mexico.

F.6.2.3.5 Species-Specific Threats

Because the Kemp's ridley sea turtle is very range limited, the general threats facing sea turtles described previously may increase impacts on this species. For example, energy extraction and development in the Gulf of Mexico are a particular threat to Kemp's ridley sea turtles because most of the nesting activity occurs there (Shaver & Caillouet, 1998). Kemp's ridley sea turtles were found stranded on beaches with crude oil on them from the *Deepwater Horizon* oil spill, and most of the sea turtles found injured and dead following the spill were Kemp's ridley sea turtles (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2011; Wilkin et al., 2017). Just over 50 percent of surface-pelagic juvenile sea turtles captured following the spill were Kemp's ridleys (McDonald et al., 2017). It should be noted that the dramatic reversal of an increasing nesting trend in the Gulf of Mexico followed the *Deepwater Horizon*, and the removal of a cohort of Kemp's ridleys that would be sexually mature now may be responsible for declines shown in 2013 and 2014 (Caillouet et al., 2016; Gallaway et al., 2016; Mitchelmore et al., 2017; Putman et al., 2015a; Wallace et al., 2020). In their study analyzing other impacts the heavy oil can have on sea turtles, Mitchelmore et al. (2017) estimated that mortality of the sea turtles in the surrounding area would be 85 percent, with 30 percent of that being a result of ingestion of the oil (Mitchelmore et al., 2017). However, increased efforts are needed to document sea turtles during oil spills as a review of over 2,000 oil spill incidents that have occurred worldwide and dating back to the late 1960s reported resulting effects to sea turtles in less than 2 percent of those incidents (Wallace et al., 2020). Gallaway et al. (2016) looked at impacts of the spill on Kemp's ridley sea turtles and their prey (shrimp and crab), and found that a reduction in prey caused a reduction in Kemp's ridleys reproductive output, leading to an increase in remigration interval as well as reduction in the number of nests per female.

Shrimp trawling in the southeastern U.S. Atlantic and Gulf of Mexico was once a significant threat to Kemp's ridleys; however, the use of sea turtle excluder devices and the general decline of shrimp fishing in recent years have greatly reduced mortality levels (Caillouet et al., 2008; Nance et al., 2012). As described in Section F.6.1.5.2 (Commercial and Recreational Industries), recreational fishing has surpassed commercial fisheries in terms of bycatch of Kemp's ridley, particularly within the Gulf of Mexico (Putman et al., 2023). Vehicle activity on sea turtle nesting beaches can also disrupt the nesting process, crush nests, and create ruts and ridges in the sand that pose obstacles to sea turtles (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2011). Beach vehicular driving is permitted on most beaches in Texas, where adult sea turtles and hatchlings have been crushed by passing vehicles, as well as on some beaches in Mexico (Shaver et al., 2020a). The artificial lighting from the vehicles can also be impactful to hatchlings; artificial light comes from a much smaller source and is provided from a smaller point, in comparison to celestial light, which brightens everything around us from its one source. This difference can be detrimental to hatchlings, who become disoriented by artificial light and as a result, may not make it to sea once they hatch (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2008).

F.6.2.4 Loggerhead Sea Turtle (*Caretta caretta*)

F.6.2.4.1 Status and Management

In 2009, a status review conducted for the loggerhead (the first sea turtle species subjected to a complete stock analysis) identified nine distinct population segments within the global population (Conant et al., 2009). In a September 2011 rulemaking, the NMFS and USFWS listed five of these distinct population segments as endangered and kept four as threatened under the ESA, effective as of October 24, 2011 (76 *Federal Register* 58868). The North Pacific Ocean, South Pacific Ocean, North Indian Ocean, Northeast Atlantic Ocean, and Mediterranean Sea distinct population segments of the loggerhead sea turtle are classified as endangered under the ESA, and the Southeast Indo-Pacific Ocean, Southwest Indian Ocean, Northwest Atlantic Ocean, and South Atlantic Ocean distinct population segments are classified as threatened. The Northwest Atlantic Ocean Distinct Population Segment is the only one that occurs entirely in the Study Area; however, loggerheads from other distinct population segments may occur rarely in the Study Area. For example, mixing occurs, rarely, with South Atlantic loggerheads enabling a limited amount of gene flow between these two distinct population segments (National Marine Fisheries Service, 2010a; Tucker et al., 2014). Critical habitat has been designated in the Study Area (see Section 3.8, Reptiles, Figure 3.8-7 through Figure 3.8-10).

Specific areas designated as critical habitat include 38 occupied marine areas within the range of the Northwest Atlantic Ocean Distinct Population Segment of loggerhead sea turtles (79 *Federal Register* 39856). To characterize different use patterns and concentrations both seasonally and geographically, NMFS named five different habitat types that comprise the critical habitat designation, which include (1) nearshore reproductive habitat (portions of nearshore waters adjacent to nesting beaches used by females and hatchlings to egress to open-water environments), (2) winter habitats (warm waters south of Cape Hatteras where juveniles and adults tend to concentrate during winter months), (3) breeding habitats (areas with high concentrations of both male and female adults during the breeding season in proximity to Florida migratory corridor and nesting grounds), (4) constricted migratory habitat (migratory corridors restricted in width), and (5) *Sargassum* habitat (juvenile loggerhead developmental habitats where *Sargassum* supports adequate prey abundance and cover) (79 *Federal Register* 39856). Physical and biological features that support the five habitat types summarized above for loggerhead sea turtle conservation include oceanic conditions that would concentrate certain life stage loggerheads together at different locations and in different seasons. The USFWS designated approximately 685 mi. of nesting beaches (in North Carolina, South Carolina, Georgia, Florida, Alabama, and Mississippi) in a separate rulemaking (79 *Federal Register* 51264).

None of these critical habitat areas include Department of Defense areas of Marine Corps Base Camp Lejeune (Onslow Beach), Cape Canaveral Air Force Station, Patrick Air Force Base, and Eglin Air Force Base, which are exempt from critical habitat designation because their Integrated Natural Resources Management Plans incorporate measures that provide a benefit for the conservation of the loggerhead sea turtle.

In 2019, NMFS and USFWS initiated a 5-year review for the Northwest Atlantic Ocean distinct population segment (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). The following sections have been updated to reflect this recent 5-year review.

F.6.2.4.2 Habitat and Geographic Range

Loggerhead sea turtles occur in U.S. waters in habitats ranging from coastal estuaries to waters far beyond the continental shelf (Chapman & Seminoff, 2016; Dodd, 1988). Loggerheads typically nest on beaches close to reef formations and in close proximity to warm currents (Dodd, 1988), preferring beaches facing the ocean or along narrow bays (Reece et al., 2013) (79 *Federal Register* 39856). Nesting in the Study Area occurs from April through September, with a peak in June and July (Dodd, 1988; Weishampel et al., 2006; Williams-Walls et al., 1983). Florida holds the largest rookery in the Northwest Atlantic region, and the third largest globally, with more limited nesting along the Gulf Coast and north through Virginia (Ceriani et al., 2017). At emergence, hatchlings swim to offshore currents and remain in the open ocean, often associating with floating mats of *Sargassum* (Carr, 1986, 1987; Witherington & Hirama, 2006). Nesting activity within the Northwest Atlantic Ocean distinct population segment include the eastern Bahamas, southwestern Cuba, the eastern Caribbean Islands, and numerous locations from the Yucatán Peninsula to Virginia (Conant et al., 2009; National Marine Fisheries Service, 2010a; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2007b).

Within the United States, the highest concentration of loggerhead nesting occurs in Florida, discussed in more detail in Section F.6.2.4.3 (Population Trends), with additional nesting reported in Texas, Alabama, Georgia, North Carolina, South Carolina, Mississippi, Louisiana, and Virginia (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). Genetic studies indicate that, although females routinely return to natal beaches, males may breed with females from multiple populations and facilitate gene flow (Bowen et al., 2004).

Migration between oceanic and nearshore habitats occurs during the juvenile stage as sea turtles move seasonally from open-ocean current systems to nearshore foraging areas (Bolten, 2003; Mansfield, 2006). After reaching a length of 40 centimeters (cm) (Carr, 1987), early juvenile loggerheads make a transoceanic crossing, swimming back to nearshore feeding grounds near their beach of origin in the Western Atlantic Ocean (Bowen et al., 2004; Musick & Limpus, 1997). Juveniles are frequently observed in developmental habitats, including coastal inlets, sounds, bays, estuaries, and lagoons with depths less than 100 m (Hopkins-Murphy et al., 2003). Based on growth rate estimates, the duration of the open-ocean juvenile stage for Northwest Atlantic loggerhead sea turtles is estimated to be 8.2 years (Bjorndal et al., 2000). Oscillations in climate affect sea turtle migrations in that loggerhead sea turtles were found to be less abundant during positive North Atlantic Oscillation (migration is directed toward the warmer weather locations) (Dellinger et al., 2022).

Juvenile loggerhead sea turtles inhabit offshore waters in the North Atlantic Ocean and Mediterranean Sea (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). They are initially carried offshore by the Gulf Stream with some leaving the currents (associated with the North Atlantic Subtropical Gyre) to forage in the Sargasso Sea while others continue along the Gulf Stream currents to oceanic waters of the North Atlantic, Mediterranean Sea, coasts of Canary Islands, Azores, Madeira, France, United Kingdom, Ireland, and Canada (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). Juvenile loggerhead sea turtles typically return to the waters of the Northwest Atlantic Ocean after several years.

Within neritic habitats, juveniles commonly forage in nearshore coastal waters, coastal inlets, sounds, bays, estuaries, lagoons, and along the continental shelf during spring, summer, and fall months from Cape Cod, south to Florida, and into the Gulf of Mexico; during winter, they are found off the coast from North Carolina to Florida (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). Navy-funded aerial surveys and stranding data as well as tagging studies funded by the Navy and Bureau of Ocean Energy Management suggest that this species is the most abundant sea turtle species using Chesapeake Bay and waters off of Cape Hatteras (Andrady, 2011; Barco & Lockhart, 2015; Bureau of

Ocean Energy Management, 2021; Burt et al., 2014; National Oceanic and Atmospheric Administration, 2015; Swingle et al., 2016).

The Chesapeake Bay is an important foraging spot for juvenile and sub-adult loggerhead sea turtles, and they are the most common species of sea turtle in the bay (Barco et al., 2018a). Recent studies analyzing spatiotemporal patterns of loggerhead sea turtles in the Chesapeake Bay and nearby Atlantic were able to determine the foraging patterns in each location (Barco et al., 2017). Results showed that foraging within the bay was constrained due to physical barriers, causing the spatial patterns to be denser than what would be seen in the ocean. Higher foraging levels were actually seen around Cape Hatteras during winter and early spring, suggesting the area offers ideal habitat conditions to induce high foraging activity. It was also determined that in the winter months, 100 percent of foraging activity occurred south of 36° North, with 48 percent, 19 percent, and 71 percent in the spring, summer, and fall, respectively. Fifty-two percent of foraging activity was in the Chesapeake Bay and the Northwest Atlantic Ocean in spring, and 78 percent and 29 percent in the summer and fall respectively (Barco et al., 2017). Research by Barco et al. (2018a) showed similar results, with high density around the bay in cooler months and less density in the summer and fall (Barco et al., 2018a).

Within the Mid-Atlantic Bight, some adults and large juveniles forage on benthic prey in the neritic habitats from New York to Virginia in the summer, and within the shelf waters from Florida to North Carolina in the winter (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). The Mid-Atlantic Bight is considered an important regional foraging area for the Northwest Atlantic loggerhead regional management unit. A tagging study conducted on nesting females from 5 different nesting sites in the U.S. determined that the Subtropical Northwest Atlantic foraging area (Bahamas, southeast Florida, Florida Keys, and Cuba) within the Mid-Atlantic Bight is the most important foraging area in terms of proportion of individuals tagged and genetic diversity harbored in the area (Ceriani et al., 2023). A recent tagging study in the Chesapeake Bay (DiMatteo et al., 2022) found that loggerheads spend the majority of their time in small, distinct areas, rarely leaving the bounds of those areas, and exhibiting “searching” behavior, which is swimming and browsing the area, and is, resemblant of foraging behavior (DiMatteo et al., 2022).

Core Sound and Pamlico Sound, North Carolina, on the border between the Northeast and Southeast U.S. Continental Shelf Large Marine Ecosystems, represent important developmental habitat for juvenile loggerheads (Epperly et al., 1995a). Although these habitats are also used by greens and Kemp’s ridleys, loggerheads are the most abundant sea turtle species within the summer developmental habitats of North Carolina (Bureau of Ocean Energy Management, 2021; Epperly et al., 1995a; Epperly et al., 1995b; Epperly et al., 1995c). In a sampling study from 2004 to 2007, juveniles were the most abundant age group among loggerheads found in the Charleston, South Carolina, shipping channel between May and August (Arendt et al., 2012). Immature loggerhead sea turtles may occupy coastal feeding grounds for 20 years before their first reproductive migration (Bjorndal et al., 2001; Putman et al., 2015b).

Subadult and adult loggerhead sea turtles tend to inhabit deeper offshore feeding areas along the western Atlantic coast, from mid-Florida to New Jersey (Hopkins-Murphy et al., 2003; Roberts et al., 2005). As late juveniles and adults, loggerhead sea turtles most often occur on the continental shelf and along the shelf break of the U.S. Atlantic and Gulf coasts, as well as in coastal estuaries and bays (Putman et al., 2015b). Hawkes et al. (2006) found that adult females forage predominantly in shallow coastal waters along the U.S. Atlantic coast less than 100 m deep, likely exploiting bottom-dwelling prey.

As water temperatures drop from October to December, most loggerheads migrate from their summer developmental habitats and eventually return to warmer waters south of Cape Hatteras, where they spend the winter (Morreale & Standora, 1998). Post-nesting females and post-mating males reside in discrete foraging areas in the Gulf of Mexico, east coast of Florida, and Bahamas Banks and use migratory corridors in the eastern Gulf of Mexico, along the Florida Keys, or through the Florida Straits to the Bahamas (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). Iverson et al. (2020) documented the longest known post-nesting migration (over 4,300 km) for non-rehabilitated wild female loggerheads; this loggerhead was tagged in the Gulf of Mexico and was tracked migrating south to Nicaragua. From a southwestern Florida nesting location, Tucker et al. (2014) tracked nine female loggerheads over multiple nesting seasons, showing five distinct winter migration destinations—islands in the Caribbean, Florida Keys, West Florida Shelf, northern Gulf of Mexico, and Yucatán Peninsula. Bovery and Wyneken (2015) analyzed seasonal variation in sea turtle density and abundance off southeastern Florida and found that loggerheads were the most frequently sighted species, with increased sightings in spring. Loggerhead sea turtles were often found in coastal waters that were west of the Florida Current (approximately 20 km offshore).

Griffin et al. (2013) offered a conceptual model of foraging strategies, as shown by tagged loggerhead sea turtles from Georgia, South Carolina, and North Carolina nesting beaches. These strategies included seasonal and year-round strategies, with summer prevalence in waters north of Cape Hatteras along neritic habitats to Cape Canaveral, Florida, with winter foraging occurring further out on the mid to outer continental shelf. Large juvenile and adult loggerhead sea turtles are captured or observed along Florida's Atlantic coast year-round (Bovery & Wyneken, 2015; Pajuelo et al., 2016). As stated previously, loggerheads were the highest occurring sea turtle species in the Study Area, with higher occurrences in spring (Barco et al., 2018a; Barco et al., 2017; DiMatteo et al., 2022).

F.6.2.4.3 Population Trends

The 2008 Recovery Plan documented recovery objectives for the following five recovery units:

(1) Northern Recovery Unit: loggerheads originating from nesting beaches from Florida-Georgia border to southern Virginia (the northern extent of nesting range).

(2) Peninsular Florida Recovery Unit: loggerheads originating from nesting beaches from the Florida-Georgia border through Pinellas County on the west coast of Florida, excluding the islands west of Key West, Florida.

(3) Dry Tortugas Recovery Unit: loggerheads originating from nesting beaches throughout the islands located west of Key West, Florida, because these islands are geographically separated from other recovery units.

(4) Northern Gulf of Mexico Recovery Unit: loggerheads originating from nesting beaches from Franklin County on the northwest Gulf Coast of Florida through Texas (the western extent of U.S. nesting range).

(5) Greater Caribbean Recovery Unit: loggerheads originating from all other nesting assemblages within the Greater Caribbean (Mexico through French Guiana, the Bahamas, and the Lesser and Greater Antilles) (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2008).

These recovery units were based on genetic data available in 2008 and geographic and geopolitical boundaries that influence their exposure to threats and recovery efforts. In 2019, the Northwest Atlantic Loggerhead Recovery team reviewed the progress toward the criteria and determined the recovery units only met 3 of 20 listing factors toward recovery (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023).

Annual total nest counts for loggerhead sea turtles on Florida's 27 core index beaches have fluctuated widely since the beginning of monitoring in 1989, with a minimum of 28,876 nests in 2007 and a maximum of 70,945 nests in 2023. Loggerhead nest counts recorded from the 27 core index beaches has been increasing in recent years. However, a detailed analysis of Florida's long-term loggerhead nesting data from 1989 to 2023 shows three distinct phases: increasing (1989-1998), decreasing (1998-2007), and increasing (2007-2023) (Florida Fish and Wildlife Conservation Commission, 2024), with an overall stable trend over the monitoring period (1989-2023). These fluctuations may be attributed to small changes in number of females or change in reproductive parameters (Florida Fish and Wildlife Conservation Commission, 2024). These fluctuations may be attributed to small changes in number of females or change in reproductive parameters (Florida Fish and Wildlife Conservation Commission, 2023c). Ceriani et al. (2019) looked at 30 years of reproductive data from the largest nesting loggerhead population worldwide (the Florida breeding population) to examine the population fluctuations. Reproductive parameters such as clutch frequency and remigration intervals are uncertain thus could not determine if there is a change in the population of females. Even with decades of conservation actions to improve and/or protect sea turtles, there is no clear evidence that the population is recovering (Ceriani et al., 2019).

Available nest count data by state, recorded a total of 118,902 loggerhead nests in 2020. In addition to Florida, nests were recorded at beaches in North Carolina, South Carolina, Georgia, Alabama, Mississippi, Texas, Virginia, and Quintana Roo (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). Nest counts conducted at the Florida index beaches in 2022 recorded 62,807 loggerhead nests. As previously stated, these numbers do not represent Florida's total annual nest counts and nest counts are only conducted during a 109-day time window of May 15 through August 31 (Florida Fish and Wildlife Conservation Commission, 2023c).

Density spatial models developed by DiMatteo et al. (2024) for loggerhead sea turtles along the U.S. east coast predicted patterns of density and abundance that reasonably matched the sightings and satellite telemetry data. In contrast to the results of the other species of sea turtles modeled, mean monthly predicted abundance of loggerheads ranged from a high of 245,609 in February to a low of 135,066 in September. Abundance was generally higher in cooler months (December to May) off the east coast and lower in warmer months (June to November). Higher density was predicted off Florida year-round (DiMatteo et al., 2024). This was concluded to be likely due to lower productivity in warmer months. Thirty years of reproductive data analyzed in Ceriani et al. (2019) documents thousands of loggerheads off the Florida coast year-round. Cape Hatteras is an important overwintering area with moderate densities of loggerheads observed beginning in May and low monthly densities north of Long Island and into the Gulf of Maine (DiMatteo et al., 2024). Shelf waters (at depths greater than 3,000 m) surveyed from June to September did not record any loggerhead sightings, but hatchlings that may be present are difficult to detect from vessel and aerial surveys (DiMatteo et al., 2024; Putman et al., 2019).

NOAA Southeast Fisheries Science Center (Garrison et al., 2023) also developed density spatial models to estimate sea turtle in-water abundance, density and distribution for the four most common sea turtle species in the Gulf of Mexico, which included the loggerhead sea turtle. The model predictions were limited to the continental shelf waters and based on aerial surveys that were conducted in 2011-2012 and 2017-2018. Average monthly predictions were generated for the period 2015-2019 (Garrison et al., 2023). The model predicted monthly average abundance of loggerhead sea turtles within the Gulf of Mexico at a low of 86,867 loggerhead sea turtles in October and a high of 290,745 loggerhead sea turtles in February. Higher loggerhead densities were predicted in the central, northern Gulf of Mexico along the coast of Louisiana, especially during winter and summer months. Higher densities were also predicted in waters close to shore, with density declining rapidly in waters greater than 100 m depth (Garrison et al., 2023).

F.6.2.4.4 Predator and Prey Interactions

Loggerhead sea turtles are primarily carnivorous in both open-ocean and nearshore habitats, although they also consume some algae (Bjorndal, 1997). Diet varies by age class (Godley et al., 1998) and by specializing in specific prey groups dependent on location. For post-hatchlings that tend to be grouped in masses of *Sargassum* and other floating habitats, various diet analyses of gut contents show parts of *Sargassum*, zooplankton, jellyfish, larval shrimp and crabs, and gastropods (Burkholder et al., 2004; Carr & Meylan, 1980; Richardson & McGillvary, 1991). Both juveniles and adults forage in coastal habitats, where they feed primarily on the bottom, although they also capture prey throughout the water column (Bjorndal, 2003).

Donaton et al. (2019) investigated benthic communities and the diets of marine predators, and found that loggerhead sea turtle diets varied strongly due to oceanographic changes in the Northeast that were linked with warming temperatures and climate change. Adult loggerheads feed on a variety of bottom-dwelling animals, such as crabs, shrimp, sea urchins, sponges, and fish. Their powerful jaws also enable them to feed on hard-shelled prey, such as whelks and conch. During migration through the open sea, they eat jellyfish, molluscs, flying fish, and squid. Donaton et al. (2019) evaluated the stomach contents of stranded loggerhead sea turtles in New York waters between 1995 and 2014. A shift in diet was observed from larger crab species (rock crab and spider crab) to smaller crab species (hermit crabs). Though there are noticeable trends visible in their diet, the research confirmed theories that loggerhead diet is highly variable on a decadal scale, with variability stemming from changes in fishing pressure on benthic communities, as well as the global temperature increase over the last few decades. Even though they consume a variety of prey, it is noticed that they specialize in a specific mixture of prey. The effects of oceanographic and atmospheric change can be associated with fluctuations in population dynamics of pelagic and demersal fishes, which have important impacts on the benthic community, thus impacting the loggerhead's diet (Donaton et al., 2019).

Common predators of eggs and hatchlings on nesting beaches are ghost crabs, raccoons, feral pigs, foxes, coyotes, armadillos, and fire ants (Campbell, 2016; Dodd, 1988; Engeman et al., 2016; Engeman et al., 2019). Eriksson and Burton (2003) has shown that management interventions for feral pigs and raccoons can significantly increase nest success in Florida, one of the main nesting concentrations of loggerheads. Arroyo-Arce et al. (2017) documented an apparently rare instance of a jaguar (*Panthera onca*) in 2014 preying on a loggerhead sea turtle at Tortuguero National Park, Costa Rica, while the turtle was on the beach. In the water, hatchlings are susceptible to predation by birds and fish. Tiger sharks are the primary predator of juvenile and adult loggerhead sea turtles in the Northwest Atlantic and off the Gulf of Mexico (Aoki et al., 2023; Fergusson et al., 2000).

F.6.2.4.5 Species-Specific Threats

In addition to the general threats described previously, mortality associated with shrimp trawls has been a substantial threat to large juvenile and subadult loggerheads because these trawls operate in the nearshore habitats commonly used by this species. Although shrimping nets have been modified with sea turtle excluder devices to allow sea turtles to escape, the overall effectiveness of these devices has been difficult to assess (Bugoni et al., 2008). Shrimp trawl fisheries account for the highest number of loggerhead sea turtle fishery mortalities; however, loggerheads are also captured and killed in trawls, traps and pots, longlines, and dredges. NMFS estimated that almost 163,000 loggerhead sea turtles are captured in shrimp trawl fisheries each year in the Gulf of Mexico, with 3,948 of those sea turtles dying as a result of their capture. Each year, several hundred loggerhead sea turtles are also captured in herring, mackerel, squid, butterfish, and monkfish fisheries; pound net fisheries, summer flounder, and scup fisheries; Atlantic pelagic longline fisheries; and gillnet fisheries in Pamlico Sound. Combined, these fisheries capture about 2,000 loggerhead sea turtles each year. Although most are released alive, about

700 sea turtles are killed annually. The scallop recreational fisheries have also shown to create behavioral impacts by way of influencing temporary distribution of marine turtles and drive changes in their movement (Wildermann et al., 2018).

Coastal development and beach renourishment projects have also had a substantial impact on loggerhead habitat, particularly nesting habitats, in Florida (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2023). Activities associated with coastal development can alter the properties of the sand at nesting beaches, as development activities lead to more sewage runoff and therefore pollution on the beach. Vehicle use on sea turtle nesting beaches is a primary example of an issue for loggerheads. Vehicles are allowed on some beaches in Florida, Georgia, North Carolina, Virginia, and Texas. Vehicles can run over and kill hatchlings or nesting adult sea turtles on the beach, disrupt the nesting process, create ruts in the sand that impede sea turtle movement, and crush nests. The artificial lighting from the vehicles can also be impactful to hatchlings; artificial light comes from a much smaller source and is provided from a smaller point, in comparison to celestial light, which brightens everything around us from its one source. This difference can be detrimental to hatchlings, who become disoriented by artificial light and as a result, may not make it to sea once they hatch (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2008).

F.6.2.5 Leatherback Sea Turtle (*Dermochelys coriacea*)

F.6.2.5.1 Status and Management

The leatherback sea turtle is listed as a single population and is classified as endangered under the ESA (35 *Federal Register* 8491). In August 2020, NMFS and USFWS announced a petition to identify the Northwest Atlantic population as a distinct population segment in addition to six other distinct population segments. However, the Services determined that replacing the existing global listing with separate listings for individual distinct population segments is not warranted. Although the best available data indicates the populations meet the criteria for significance and discreteness, they found that it would not further the purposes of the ESA to recognize and list seven distinct population segments separately as endangered under the ESA. The current global listing of the species remains in effect (85 *Federal Register* 48332). Recent information on population structure (through genetic studies) and distribution (through telemetry, tagging, and genetic studies) have led to an increased understanding and refinement of the global stock structure. Leatherback sea turtles from nesting stocks originating throughout the Atlantic have the potential to be within the offshore portions of the Study Area, but only two of these—the Florida genetic stock and the Northern Caribbean genetic stock—nest on beaches in the jurisdiction of the United States.

Critical habitat has been designated in the Study Area for this species (see Section 3.8, Reptiles, Figure 3.8-6). In 1978, critical habitat was designated for the leatherback's terrestrial environment on St. Croix Island at Sandy Point because of its importance as a nesting habitat (43 *Federal Register* 43688). In 1979, critical habitat was designated for the waters next to Sandy Point, St. Croix, up to and including the waters from the 100-fathom curve shoreward to the mean high tide line (44 *Federal Register* 17710). The essential physical and biological feature of this critical habitats function as an important courtship and mating area adjacent to the nesting beach.

F.6.2.5.2 Habitat and Geographic Range

The leatherback sea turtle is distributed worldwide in tropical and temperate waters of the Atlantic, Pacific, and Indian Oceans (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b).

Important nesting areas in the Western Atlantic Ocean occur in Florida, St. Croix, Puerto Rico, Costa Rica, Panama, Colombia, Trinidad and Tobago, Guyana, Suriname, French Guiana, and southern Brazil (Brautigam & Eckert, 2006; Márquez, 1990; Spotila et al., 1996). They also occasionally nest in North and

South Carolina (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b). Other minor nesting beaches are scattered throughout the Caribbean, Brazil, and Venezuela (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b). Leatherback nesting season begins and ends a few months earlier than that of the other sea turtle species that nest in the Study Area, beginning in February in the more northern nesting habitats (e.g., Florida) and continues until around July in more southern nesting habitats (e.g., Puerto Rico).

Females remain in the general vicinity (within 100 km) of nesting beaches between nestings, with total residence in the nesting and inter-nesting habitat lasting up to four months (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b). Horrocks et al. (2016) tagged over 3,100 female leatherbacks in the Caribbean Sea and found that females traveled an average of 160 km between nesting events within the same season. Migrations between nesting seasons were typically to the north towards more temperate latitudes, which support high densities of jellyfish prey in the summer.

In the Atlantic Ocean, equatorial waters appear to be a barrier between breeding populations. In the northwestern Atlantic Ocean, post-nesting female migrations appear to be restricted to north of the equator, but the migration routes vary (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b). Leatherbacks made round-trip migrations from where they started through the North Atlantic Ocean heading northwest to fertile foraging areas off the Gulf of Maine, Canada, and Gulf of Mexico; others crossed the ocean to areas off Western Europe and Africa, while others spent time between northern and equatorial waters. These data support earlier studies that found adults and subadults captured in waters off Nova Scotia stayed in waters north of the Equator (James et al., 2005a; James et al., 2005b; James et al., 2006). A recent study by Rider et al. (2024) tagged leatherbacks off the coasts of Massachusetts in the summer and North Carolina in the spring in order to assess dive and movement behavior and to identify migratory corridors and foraging areas along the U.S. east coast. Researchers found that leatherbacks consistently use the southern New England coast as a primary foraging area, with the waters along the Nantucket Shoals being the most consistently used foraging habitat, as it is a highly productive region where jellyfish occur. In leatherbacks that were tagged and found foraging in the mid-Atlantic Bight from Cape Hatteras, North Carolina to the mouth of the Delaware Bay, leatherback distribution was closely associated with the thermocline. The Gulf Stream was found to serve as a potential intermediate foraging area that leatherback sea turtles migrate to in the winter, as they stray away from higher latitudes along the east coast before returning in spring and summer (Rider et al., 2024).

Northwest Atlantic leatherbacks use the western Equatorial Atlantic and Gulf of Mexico, as a foraging hotspot, even more so than off the Florida Panhandle (Sasso et al., 2021). Foraging was the predominant behavior observed during satellite tracking studies in the Gulf of Mexico. The main sites where this behavior was observed included the northeast corner from Louisiana to Florida, the coastal shelf of southwest Florida, and eastern side of Campeche Bay (Aleksa et al., 2018). Leatherback movements in the Gulf of Mexico are strongly influenced by variables such as abundance of jellyfish, convergence zones, salinity, temperature, and nutrients. Due to the extensive research that has been conducted on this, scientists are starting to feel as if they are now relatively familiar with the overall spatio-temporal patterns of movement of the sea turtles. However, it is still difficult to research post-hatchling and early juvenile sea turtles because these life stages are entirely oceanic (Sasso et al., 2021). Post-hatchling and early juvenile leatherback sea turtles are restricted to waters warmer than 79°F (26°C); consequently, much time is spent in the tropics (Eckert, 2002). They are not considered to associate with *Sargassum* or other flotsam, as is the case for all other sea turtle species (Horrocks, 1987; Johnson, 1989). Upwelling areas, such as equatorial convergence zones, serve as nursery grounds for post-hatchling and early juvenile leatherback sea turtles because these areas provide a high biomass of prey (Musick & Limpus, 1997).

Late juvenile and adult leatherback sea turtles are known to range from mid-ocean to the continental shelf and nearshore waters (Barco & Lockhart, 2015; Grant & Ferrell, 1993; Schroeder & Thompson, 1987; Shoop & Kenney, 1992). Although leatherbacks were observed annually in Chesapeake Bay, they were not common and unevenly distributed. Juvenile and adult foraging habitats include both coastal and offshore feeding areas in temperate waters and offshore feeding areas in tropical waters (Dodge et al., 2014). Dodge et al. (2014) tagged adults and subadult leatherback sea turtles off the coast of Massachusetts and found that the sea turtles showed a strong preference for the Northeast U.S. Continental Shelf waters during the summer, with the concentrated movements off Virginia and North Carolina. Additionally, leatherbacks were recorded occurring near the mouth of the Chesapeake Bay for multiple days during the summer, ranging from 5 to 15 days. Leatherback sea turtles may prefer a temperate neritic habitat during the summer, due to the availability of their gelatinous pretty sources (e.g., jellyfish) in the summer (Dodge et al., 2014). Leatherbacks have been shown to travel shorter distances at slower rates and increased diving rates in areas of high prey abundance, which is related to seasonal availability of prey (Wallace et al., 2015).

Eckert and Hart (2021) observed patterns on leatherback distribution throughout the Atlantic Ocean, finding that habitat ranged from nesting areas in the wider Caribbean Region to foraging areas that extend through the Caribbean and into the Gulf of Mexico. Major nesting sites were determined to be concentrated in the Guianas and Trinidad & Tobago. Other important nesting sites were found in Costa Rica, Panama, Columbia, the Virgin Islands, Puerto Rico, and Florida (Eckert & Hart, 2021). Leatherback sea turtles mate in waters adjacent to nesting beaches and along migratory corridors (Cummings et al., 2016; Figgener et al., 2016).

Studies done on the importance of the Gulf Frontal System and the foraging behaviors of leatherback sea turtles show that their high densities strongly coincided with seasonal phytoplankton blooms. Phytoplankton blooms provide ample nutrients to the water column, attracting marine predators like the sea turtle. Leatherbacks showed preferences for waters with strong sea surface temperature gradients and a deep mixed layer, which aids in the mixing of all the newly acquired available nutrients. Leatherbacks interact with this strong frontal system in the gulf during their post-nesting migration across the north Atlantic, where they feed on primarily jellyfish (Chambault et al., 2017).

F.6.2.5.3 Population Trends

Recent monitoring efforts throughout the Northwest Atlantic Region have noted a decline in annual counts of nests of nesting females. Reasons for the population decline include fisheries interactions (bycatch in fishing gear, illegal fishing activity), pollution (plastic, discarded gear, and chemical), and nesting habitat loss due to erosion and accretion of beaches (Northwest Atlantic Leatherback Working Group, 2018). One of the most globally important stocks of leatherback sea turtles, the Southern Caribbean Stock, nests in French Guiana, Guyana, Suriname, and Trinidad, but migrates and forages throughout the North Atlantic. The Western Caribbean stock of the Central American coast also migrates through the Study Area en route to North Atlantic foraging grounds. In the Study Area, nesting populations are found in southern Florida, Culebra, Puerto Rico, and the U.S. Virgin Islands. (National Marine Fisheries Service, 2011c; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013; Turtle Expert Working Group, 2007). Sporadic nesting also occurs in Georgia, South Carolina, (National Marine Fisheries Service and U.S. Fish and Wildlife Service, 1992; Rabon et al., 2003; Schwartz, 1989), and as far north as North Carolina as well as in the Gulf of Mexico on the Florida panhandle. A decline in nesting numbers has been reported from representatives of these countries prompting a trends analysis from the Northwest Atlantic Leatherback Working Group (Northwest Atlantic Leatherback Working

Group, 2018). Several sites in the analysis showed low abundance in early years followed by many years of increasing abundance, but recent declines are reverting back to low levels (e.g., St. Croix, Florida, Culebra [Puerto Rico], Cayenne [French Guiana]) (Northwest Atlantic Leatherback Working Group, 2018).

The Florida nesting stock comes ashore primarily along the east coast of Florida. In the 1980s, fewer than 100 nests per year were reported. Based on data extrapolated from the index nesting beach surveys, nesting activity has shown an annual growth rate of 1 percent between 1989 and 2005 (National Marine Fisheries Service, 2013b). Larger growth rates (10.2 percent increases per year) in nesting activity in this area have been shown from 68 Florida beaches since 1979 (Stewart et al., 2011; Stewart et al., 2014). Florida statewide nesting reports show nesting numbers fluctuating between 896 nests and 1,712 nests during a 5-year period between 2011 and 2015. Surveyors counted 486 leatherback nests on the 27 core index beaches in 2023. However, these counts do not include leatherback nesting at the beginning of the season (prior to May 15), nor do they include all beaches in Florida where leatherbacks nest (Florida Fish and Wildlife Conservation Commission, 2024). Leatherback nest counts have exponentially increased over the monitoring period; however, there has been a fluctuation in counts from 2014 to 2023. Leatherback nest counts reached a peak in 2014 followed by a steep decline from 2015–2017, and then an increase from 2018–2023. Only a few hundred leatherbacks nest annually in Florida and females may lay as many as 11 clutches during a nesting season, therefore, the fluctuations in nest counts may be the result of a change in the number of nesting females (Florida Fish and Wildlife Conservation Commission, 2024).

Density spatial models developed by DiMatteo et al. (2024) predicted leatherback sea turtle abundance, density, and distribution patterns throughout the entire U.S. east coast, including offshore areas. Leatherback sea turtles had the largest change between high and low abundance predictions compared to the other three sea turtle species (green, Kemp’s ridley, and loggerhead turtles) included in this study, with monthly predicted estimates spanning a full order of magnitude (DiMatteo et al., 2024). This pattern is supported by the sightings data, and may reflect the east coast’s importance as a nesting and migratory habitat, rather than a foraging habitat, as sea turtles from the wider Caribbean region migrate to the north Atlantic Ocean basin seasonally to forage. The model predicted higher abundance of leatherback sea turtles in warmer months than cooler months, with a mean abundance of 54,329 for September to a low of 4,655 leatherbacks for February. Sightings data supported predicted occurrence of leatherbacks off the coast of Georgia and Florida year-round in low numbers during the cool months and high numbers during the warm months. Shipboard surveys supported predictions of leatherback occurrence in the mid-Atlantic, from the Outer Banks north to Cape Cod and offshore in the Gulf Stream from June until November. Leatherbacks were found consistently in productive, offshore areas, due to their diet of pelagic, gelatinous prey. They remain foraging out at sea for several years, until they begin mating in nearshore waters (DiMatteo et al., 2024).

Leatherback sea turtles are also present year-round in the Gulf of Mexico, with almost all sightings during winter and spring occurring east of the mouth of the Mississippi River (Garrison et al., 2023). Abundance is greater in autumn months, as post-nesting sea turtles enter the Gulf of Mexico from their Caribbean nesting seasons in the summer. Like with other sea turtle species, leatherback movement patterns are strongly correlated with oceanic parameters like sea surface temperature, nutrients, distance from shore, and mixed layer depth, as these conditions affect the abundance of their prey, jellyfish. They remain scattered about the Gulf of Mexico until the winter when they return to the Caribbean to breed. The Gulf of Mexico is likely a preferred location due to its high nutrient availability, allowing them to sufficiently sustain their diets (Sasso et al., 2021).

NOAA Southeast Fisheries Science Center (Garrison et al., 2023) developed density spatial models to estimate sea turtle in-water abundance, density, and distribution for the four most common sea turtle species in the Gulf of Mexico, including the leatherback sea turtle. The model predictions were limited to the continental shelf waters and based on aerial surveys that were conducted in 2011–2012 and 2017–2018. Average monthly predictions were generated for the period 2015–2019. The model predicted monthly average animal abundance within the Gulf of Mexico at a low of 518 leatherbacks in November and a high of 6,922 leatherbacks in July, which is consistent with predictions that density would be higher during the months that have warmer water temperatures. Higher densities were also predicted in offshore waters deeper than 50 m. However, lower density was predicted in the central northern Gulf of Mexico, especially during the winter (Garrison et al., 2023).

F.6.2.5.4 Predator and Prey Interactions

Leatherbacks lack the crushing chewing plates characteristic of hard-shelled sea turtles that feed on hard-bodied prey. Instead, they have pointed tooth-like cusps and sharp-edged jaws that are adapted for a diet of soft-bodied open-ocean prey. Leatherback sea turtles feed throughout the water column, predominantly on jellyfish (Davenport, 1988; Eckert et al., 1989; Eisenberg & Frazier, 1983; Grant & Ferrell, 1993; James et al., 2005b; James et al., 2005c; Salmon et al., 2004) (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013; Wallace et al., 2015). In Atlantic Canada, leatherbacks feed on jellyfish of *Cyanea* spp. and *Aurelia* spp. (James & Herman, 2001; Votier et al., 2011); In North Carolina and Georgia, leatherback sea turtles feed on cannonball jellies (*Stomolophus meleagris*) (Frick et al., 1999; Grant & Ferrell, 1993). Patterns in feeding behavior off St. Croix, U.S. Virgin Islands, over a 24-hour period suggest an interaction between leatherback diving and vertical movements of the deep scattering layer (a horizontal zone of planktonic organisms), with more frequent and shallower dives at night compared with fewer and deeper day dives (Eckert et al., 1989). Studies by Nordstrom et al. (2020) and Wallace et al. (2015) confirmed that leatherbacks actually have a highly specified diet of gelatinous zooplankton, including jellyfish and salps, in their summer foraging grounds off Atlantic Canada.

Predators of leatherback nests are common to other sea turtle species (e.g., terrestrial mammals and invertebrates). Burns et al. (2016) found that nesting female leatherbacks expend a significant amount of time and energy, despite increased risk of direct predation while on land, to obscure nests. After laying nests and covering with sand, the female's return to the ocean is not linear, and is likely an attempt at decoy behavior as a further measure to protect the clutch. In the water, hatchlings are susceptible to predation by birds and fish. Sharks are the primary predator of juvenile and adult leatherback sea turtles (Aoki et al., 2023; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013, 2020b).

F.6.2.5.5 Species-Specific Threats

In addition to the general threats to sea turtles described previously, bycatch in commercial fisheries is a particular threat to leatherback sea turtles. Net fisheries and entanglement are dominant threats to leatherbacks at sea (Eckert & Hart, 2021). Incidental capture in longline and coastal gillnet fisheries has caused a substantial number of leatherback sea turtle deaths, likely because leatherback sea turtles dive to depths targeted by longline fishermen and are less maneuverable than other sea turtle species (Archibald & James, 2018). Shrimp trawls in the Gulf of Mexico have been estimated to capture about 3,000 leatherback sea turtles, with 80 of those sea turtles dying as a result (Finkbeiner et al., 2011; Wallace et al., 2010b). Along the Atlantic coast of the United States, NMFS estimated that about 800 leatherback sea turtles are captured in pelagic longline fisheries, bottom longline, and drift gillnet fisheries for sharks as well as lobster, deep-sea red crab, Jonah crab, dolphinfish and wahoo, and Pamlico Sound gillnet fisheries. Although most of these sea turtles are released alive, these fisheries kill about 300 leatherback sea turtles each year (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2013; Stewart et al., 2016).

Harvest of leatherback sea turtle eggs and adult sea turtles continues to be a threat in many parts of the world (Humber et al., 2014). A review of Pacific observer data from 34 swordfish-targeting shallow-set longline fleets found 331 leatherback sea turtle interactions observed between 1989 and 2015 and identified bycatch hotspots in the central North Pacific Ocean and eastern Australia (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b). These ocean-based threats have some countries identifying them as threatening 20 to 50 percent of the nation's adult leatherback sea turtle population, particularly in Trinidad and in the Guianas (Barragan et al., 2022).

All life stages of leatherback turtles are vulnerable to oil spills on land or at sea. Within the last 60 years, 2000, oil spill events have occurred in habitats that are used by sea turtles (Wallace et al., 2020). Plastics, a common type of marine debris, is a major threat to leatherback sea turtles. Plastic bags, often mistaken for jellyfish, have been found during necropsies of leatherbacks; and phthalates, a chemical derived from plastics, has been found in leatherback egg yolks (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b). Although unquantified, overall impact of marine debris on leatherback sea turtles is anticipated to be severe with an approximately increase from 5.2 to 19.3 million tons per year over the last 30 years of debris entering the marine environment (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b).

Lastly, climate change may impact leatherbacks in ways different from other sea turtle species because their distribution is so closely associated with jellyfish aggregations (which are affected by changing ocean temperatures and dynamics) (Nordstrom et al., 2020; Pike, 2014). Robinson et al. (2013) suggest that climate change impacts are contributing to the Pacific leatherback population declines through a shifting of nesting dates due to increased stressor exposure. The observed mean nesting date shifts in the Atlantic leatherback genetic stocks, in contrast to Pacific populations, may increase resiliency of Atlantic leatherbacks to climate-related impacts. Global ocean temperatures will continue to warm, and increases in seasonal and annual mean surface temperatures are expected to be larger in the tropics and Northern Hemisphere subtropics where leatherback sea turtles nest (National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2020b).

F.6.2.6 American Crocodile (*Crocodylus acutus*)

F.6.2.6.1 Status and Management

The American crocodile occurs within the jurisdictional boundaries of many different countries, with a primary distribution throughout coastal waters of the Caribbean Sea as well as the Pacific coast of Central and South America, ranging from Mexico to Ecuador (Thorbjarnarson et al., 2006). Population declines have been attributed to loss of habitat and, before they were listed as a protected species and throughout the listing process, extensive poaching (U.S. Fish and Wildlife Service, 2010d). The American crocodile was listed as endangered throughout its range under the ESA in 1979 (44 *Federal Register* 17710), where individuals found in Florida represent the northern extent of the species' range (U.S. Fish and Wildlife Service, 2010d). In 2007, the USFWS designated the American crocodile in Florida as a distinct population segment due to their genetic isolation. This designation also downlisted the Florida distinct population segment from endangered to threatened under the ESA, while the status of the population outside of Florida remains listed as endangered (72 *Federal Register* 13027) (U.S. Fish and Wildlife Service, 2010d). In 1976, Florida Bay and its associated brackish marshes, swamps, creeks, and canals was designated as critical habitat for the Florida population. In 1977, it was modified to include a more accurate map representation of the habitat (41 *Federal Register* 41914; 44 *Federal Register* 75074) (see Section 3.8, Reptiles, Figure 3.8-11). At the time of the original ESA listing, the state's population of crocodiles was concentrated throughout the inshore waters of Florida Bay. While no primary constituent elements of the critical habitat were indicated, this area represented key nesting habitat as all known breeding females inhabited and nested in this designated area (41 *Federal Register* 41914).

F.6.2.6.2 Habitat and Geographic Range

The American crocodile is typically found in fresh or brackish coastal habitats, including, but not limited to rivers, ponds, lagoons, and mangrove swamps (Mazzotti, 2014; Mazzotti et al., 2007; Wheatley et al., 2012). American crocodiles generally occur in water with salinities less than 20 parts per thousand; however, they possess salt lingual glands allowing them to excrete excess salt (Cherkiss et al., 2014; Wheatley et al., 2012) and occasionally inhabit more saline environments (e.g., Florida Bay) (Wheatley et al., 2012). Most crocodile sightings in more saline water are females attending nest sites, hatchlings at nest sites, or juveniles presumably avoiding adults (Mazzotti et al., 2007). Females construct nests on elevated, well-drained sites near the water such as ditch banks and beaches. In the United States, artificial nesting sites within berms along canal banks provide nearly ideal nesting conditions because they are elevated, well drained, and near relatively deep, low-to-intermediate salinity water (Mazzotti et al., 2007). These artificial nesting habitats appear to be compensating for natural habitat elsewhere in Florida and account for much of the increase in nesting documented since 1975 (U.S. Fish and Wildlife Service, 2022a).

The American crocodile is known to inhabit inshore marine waters and is not predisposed to travel across the open ocean (Cherkiss et al., 2014). Instead, they prefer calm warm waters with minimal wave action, and most frequently occur in sheltered, mangrove-lined estuaries (Mazzotti, 1983). No available evidence suggests that crocodiles cross the Florida Straits; therefore, this species is not expected to occur in offshore areas in the Study Area. The American crocodile, however, can travel long distances in nearshore environments. For example, Cherkiss et al. (2014) tracked an individual American crocodile over a 14-year period. The crocodile was originally marked in Homestead, Florida, as a young-of-the-year in 1999. It was later recaptured multiple times, sometimes as far as 388 km away along the southwest coast of Florida. After several relocations and numerous sightings, this individual returned to the same canal system in which it was first captured. Studies have only suggested that the reason for the return is likely due to searching for familiar territory, however, not all crocodiles exhibit this behavior, and distance has a strong influence on whether they will return to capture sites (Brunell et al., 2023). Overall, sightings of American crocodiles occur with increasing frequency in many of the lower keys (U.S. Fish and Wildlife Service, 2022a).

Within the United States, the current known range of the American crocodile in Florida consists of coastal areas of central and Southern Broward, Miami-Dade, Monroe, Collier, and Lee Counties. Crocodile are regularly observed along the shorelines of Florida Bay, Biscayne Bay, and the upper Florida Keys, and within the Cooling Canal System and adjacent canals and wetlands (U.S. Fish and Wildlife Service, 2022a). Most nesting occurs on the mainland shore of Florida Bay within the Northeast Florida Bay and Flamingo/Cape Sable areas. Other nesting occurs primarily at Key Largo and the Cooling Canal System with frequent nesting at Biscayne Bay north to as far as Virginia Key; throughout the Florida Keys south of North Key Largo, and the southwest coast of Florida from north of Highlands Beach to Sanibel Island (U.S. Fish and Wildlife Service, 2022a).

F.6.2.6.3 Population Trends

In 1976, the American crocodile population in Florida was estimated to be between 200 and 300 individuals (40 *Federal Register* 58308), with only 10 to 20 breeding females estimated in 1975 (40 *Federal Register* 58308). An estimated 20 nests were laid in Florida in 1975. The Florida population of the American crocodile has increased, and its distribution has expanded, since it was listed as endangered. As a result of conservation measures, including habitat protection, the number of nests increased to 85 in 2004 (Mazzotti et al., 2007). The most recent population estimate, provided by the USFWS, estimates the current Florida population of crocodiles to range between 1,300 and 3,150 adults, with an estimated 149 to 189 nests recorded per year in Florida between 2013 and 2021 (U.S. Fish and Wildlife Service, 2022a). Based on systematic monitoring of four of the five nesting areas of the American crocodile in Florida, three are exhibiting stable or increasing trends in nesting and natural recruitment. The species is gradually recovering in the United States, but survey data from Central and South America are relatively poor.

Increased sightings of crocodiles on the airfield at the Naval Air Station Key West has initiated interest in having surveys for crocodiles performed on station. The Navy is currently monitoring occurrences of American crocodiles at Naval Air Station Key West. In 2014, 21 American crocodiles were identified at the air station. Nesting may also occur on the coastal portions of the air station (Mazzotti, 2014). During 2016 spotlight surveys (occurring in January, April, June, and August), a total of seven crocodiles were observed at Naval Air Station Key West. Nesting activity was confirmed at Naval Air Station Key West in 2022 (Wheatley-Techmer, 2023).

F.6.2.6.4 Predator and Prey Interactions

The American crocodile typically forages from shortly before sunset to shortly after sunrise (U.S. Fish and Wildlife Service, 2010d). During these times, crocodiles feed on any prey items that can be caught and overpowered (Mazzotti et al., 2007). Adults feed on fish, crabs, birds, turtles, snakes, and small mammals, while young feed on aquatic invertebrates and small fish.

Fire ants are predators of crocodile eggs. Crocodile hatchlings may be preyed on by large fish, birds, other large reptiles and amphibians, or even other crocodiles. Larger juvenile and adult crocodiles have no known natural predators (Mazzotti et al., 2007).

F.6.2.6.5 Species-Specific Threats

Habitat loss is a primary threat to the American crocodile (U.S. Fish and Wildlife Service, 2010d). Development in coastal areas of Florida diminishes American crocodile habitat and restricts the species' breeding range. Although they are expanding their overall distribution north, many of the areas on the east and west coast of Florida (north of their core range) are highly urbanized with very little habitat. American crocodiles are an adaptable species in terms of habitat use (altered and artificial habitats) and have adapted to increasing development and urbanization in Florida (U.S. Fish and Wildlife Service, 2022a).

Sea level rise due to global warming is expected to result in the inundation and loss of existing American crocodile nesting habitat with sea level expected to rise 17 to 41 in. above the 1992 mean sea level by 2060 and 32 to 103 in. above 1992 mean sea level by 2100 (U.S. Fish and Wildlife Service, 2022a). In addition, high salinities in the wetlands of Florida Bay likely accelerated the reduction of the population as both growth and survival of American crocodiles decrease with increasing salinities (U.S. Fish and Wildlife Service, 2022a).

The introduction in Florida of Nile crocodiles (*Crocodylus niloticus*), confirmed recently through genetic analyses, presents threats to the American crocodile (Rochford et al., 2016). As a competitor for prey and habitat, the Nile crocodile can also likely predate smaller American crocodiles. In addition, many crocodilian species are already known to hybridize in captivity and where their native ranges overlap in the wild (e.g., Cuban crocodile [*Crocodylus rhombifer*]), which can degrade the genetic integrity of the American crocodile (Weaver et al., 2008). Because of similarity of appearance, Nile crocodile persecution by humans would likely include accidental poaching of American crocodiles (Rochford et al., 2016). Introduction of exotic species such as green iguana (*Iguana iguana*), Burmese pythons, and the Argentine black and white tegu (*Salvator marianae*) have the potential to be significant predators of American crocodiles and/or their eggs (U.S. Fish and Wildlife Service, 2022a).

F.6.2.7 American Alligator (*Alligator mississippiensis*)

F.6.2.7.1 Status and Management

American alligator populations began to decline in the late 1800s, when unregulated hunting for the hides became prevalent, with population numbers close to extinction in some areas (Savannah River Ecology Laboratory & Herpetology Program, 2012). A hunting ban in the 1950s and other recovery efforts allowed the species to rebound (52 *Federal Register* 21059). American alligators were listed as an endangered species in 1967 under a law that preceded the ESA of 1973 (National Park Service, 2012). No critical habitat has been designated for this species. Federal legislation in the 1970s and 1980s, including the ESA and amendments to the Lacey Act in 1981, ensured the alligators' protection, and eventually their comeback. In 1987, the alligator was declared "no longer biologically threatened or endangered" (52 *Federal Register* 21059). However, to ensure protections to the American crocodile and other endangered crocodilians, the American alligator is listed under the ESA classification of "threatened due to similarity of appearance" to the American crocodile (52 *Federal Register* 21059). Accordingly, federal agencies are no longer required to consult with USFWS pursuant to section 7 of the ESA. Hunting and trade of the American alligator are now permitted and regulated by the USFWS (National Park Service, 2012; Savannah River Ecology Laboratory & Herpetology Program, 2012).

F.6.2.7.2 Habitat and Geographic Range

The American alligator occurs on the Atlantic Coast of North America from Florida through coastal North Carolina, and along the Gulf Coast into Texas (Savannah River Ecology Laboratory, 2023). The American alligator's primary habitats are freshwater swamps and marshes but may also include lakes, canals, ponds, reservoirs, and rivers. Females and juveniles occasionally use seasonal wetlands, such as Carolina Bays (Savannah River Ecology Laboratory, 2023). As alligators lack lingual salt glands, the species has a limited capacity to tolerate highly saline environments (Mazzotti & Dunson, 1989). In coastal areas, alligators move between freshwater and estuarine waters. Size and sex influence the habitat that alligators reside in; adult males generally prefer deep, open water within coastal water bodies during the spring breeding season. Adult females are also found in open water during breeding season, but they prefer to move to lake and marsh edges during nesting and hatching seasons.

Males have a minimal role in parenting; however, females will remain with juveniles and hatchlings for up to a year or more for protection during this vulnerable life stage (Savannah River Ecology Laboratory & Herpetology Program, 2012). After juveniles have hatched, they remain with the adult female for up to a year or more for protection during this vulnerable life stage (National Park Service, 2012; Savannah River Ecology Laboratory & Herpetology Program, 2012). Smaller alligators prefer wetlands with dense vegetation for protection and prey advantage (Savannah River Ecology Laboratory & Herpetology Program, 2012).

F.6.2.7.3 Population Trends

Alligators are active year-round with the highest activity occurring during the warmer months. Breeding season begins in May with nest building by females occurring in June. Nests are constructed within marsh reeds or other vegetation, with rotting vegetation keeping eggs warm during incubation. Females lay between 20 and 60 eggs in late June to mid-July with incubation lasting approximately 65 days before hatching begins (Savannah River Ecology Laboratory, 2023). Following state and federal management of this species, alligator populations have rebounded to an estimated total in the millions of individuals (Savannah River Ecology Laboratory & Herpetology Program, 2012). The Navy is currently monitoring occurrences of American alligators at Naval Air Station Key West. During 2016 spotlight surveys (occurring in January, April, June, and August), one alligator was observed at Naval Air Station

Key West. No nesting activity was observed on the Naval Air Station Key West properties (Wheatley-Techmer, 2023). Nesting may occur on the coastal portions of the air station (Mazzotti et al., 2016).

F.6.2.7.4 Predator and Prey Interactions

American alligators are opportunistic carnivores. Adults eat a variety of animals, including large fish, turtles, snakes, birds, and small mammals. Hatchlings and smaller alligators eat insects, crayfish, snails and other invertebrates, small fish, and amphibians (Savannah River Ecology Laboratory & Herpetology Program, 2012).

Alligator eggs are often preyed upon by raccoons, opossums, skunks, feral pigs, and other terrestrial nest predators. Similarly, young alligators are preyed upon by raccoons, crabs, large snakes, turtles, birds, and even fish (Savannah River Ecology Laboratory & Herpetology Program, 2012).

F.6.2.7.5 Species-Specific Threats

Main threats to the American alligator is destruction and degradation and loss of wetland habitat, often associated with human development. Alligator nests may be depredated by racoons and bears whereas juveniles are vulnerable to predation by wading birds, otters, and larger alligators (Florida Fish and Wildlife Conservation Commission, 2023a).

F.6.3 SPECIES NOT LISTED UNDER THE ENDANGERED SPECIES ACT

F.6.3.1 Diamondback Terrapin (*Malaclemys terrapin*)

F.6.3.1.1 Habitat and Geographic Range

The typical habitat of the diamondback terrapin includes coastal swamps, estuaries, lagoons, tidal creeks, mangroves, and salt marshes with salinities ranging from 0 to 35 ppt. Diamondback terrapins have salt glands around their eyes, allowing them to secrete excess salt from their blood, and survive in salty environments (University of Georgia, 2023). Although diamondback terrapins are found in brackish water, periodic access to freshwater is required for long-term health. Diamondback terrapins play an important role in coastal saltwater marsh ecosystems by aiding in seed dispersal, controlling insect and snail populations, and contributing to other ecological services (e.g., removing suspended sediments and contaminants in water) through perpetuating eelgrass spread (Pfau & Roosenburg, 2010).

Although diamondback terrapins are an aquatic turtle and spend the majority of their life in water, they do leave the water to bask and lay eggs (University of Georgia, 2023). During the cold winter months, diamondbacks hibernate in the mud at the bottom of tidal creeks. Nesting females wander considerable distances on land before nesting. Nests are usually laid in sand dunes or scrub vegetation near the water. Eggs are typically laid in late May through August and generally take 50 to 80 days to hatch (University of Georgia, 2023).

The distribution of diamondback terrapins is best described as discontinuous along the approximately 5,000 km of coastline between Cape Cod, Massachusetts, and Corpus Christi, Texas (Pfau & Roosenburg, 2010). Throughout this distribution, there are seven defined subspecies of the diamondback terrapin based primarily on differences in carapace morphology and skin coloring (Hart & Lee, 2006). The subspecies are listed below:

- Carolina diamondback terrapin (*Malaclemys terrapin centrata*)
- Texas diamondback terrapin (*Malaclemys terrapin littoralis*)
- Ornate diamondback terrapin (*Malaclemys terrapin macrospilota*)
- Mississippi diamondback terrapin (*Malaclemys terrapin pileata*)
- Mangrove diamondback terrapin (*Malaclemys terrapin rhizophorarum*)

- Eastern Florida diamondback terrapin (*Malaclemys terrapin tequesta*)
- Northern diamondback terrapin (*Malaclemys terrapin terrapin*)

Despite this extensive distribution, its zone of occurrence is very linear and in places fragmented, resulting in a relatively small total area of occupancy (Hart & Lee, 2006).

F.6.3.1.2 Population Trends

Terrapins have a long history of exploitation by humans, who harvested them for food for decades (University of Georgia, 2017). The current population size of diamondback terrapins in the United States is unknown but estimated to be over 100,000 individuals. Most diamondback terrapin populations range from stable to declining (University of Georgia, 2017).

F.6.3.1.3 Predator Prey Interactions

Diamondback terrapins feed largely on invertebrates, crustaceans, mollusks, and a variety of plant materials and algae. Adult terrapins are predators of the Salt Marsh Periwinkle (*Littoraria irrorata*), a snail that grazes on epiphytes that grow on salt marsh cord grass (*Spartina alterniflora*). As a periwinkle predator, diamondback terrapins are frequently identified as a potential saltmarsh keystone species because periwinkles damage and kill cord grass, leaving a barren mudflat and increasing marsh erosion rates (International Union for Conservation of Nature, 2019). Nests, hatchlings, and sometimes adults are eaten by raccoons, skunks, coyotes, feral hogs, and non-native Norway rats (International Union for Conservation of Nature, 2019).

F.6.3.1.4 Species-Specific Threats

The species has declined significantly from historic levels, in part due to 19th and 20th century harvesting for fashion and food utilization. Harvesting of turtles and eggs is no longer a primary threat to this species. In states with a commercial blue crab (*Callinectes sapidus*) fishery, which includes almost the terrapin's entire range, incidental drowning in crab pots is considered to be the major threat. Additionally, adult female terrapins are frequently struck and killed by motor vehicles while attempting to cross roads in search of nesting sites (International Union for Conservation of Nature, 2019). Additional threats include habitat loss from climate change, predation from wild hogs, raccoons, and rats, and harvest from the pet trade (Florida Fish and Wildlife Conservation Commission, 2023b).

F.7 BIRDS AND BATS

F.7.1 GENERAL BACKGROUND

With exceptions noted below, the general background for birds and bats in the Study Area as described in the 2018 Final EIS/OEIS (Section 3.9.2.1, General Background) has not appreciably changed. As such, the information presented in the 2018 Final EIS/OEIS remains valid. The following information was updated from the 2018 Final EIS/OEIS:

- use of offshore areas by bats for migration
- effects of climate change on seabirds
- more current research on seabirds
- for endangered species, new species have been listed and new Critical Habitat has been proposed, population estimates have been updated, and new information on distribution and threats has been added

There are at least 160 species of birds that regularly occur in the Study Area (Sibley, 2014). These are primarily waterbirds of marine, estuarine, coastal, and inshore waters including seabirds, wading birds, shorebirds, and waterfowl. Many additional species are seasonal migrants that transit through the Study Area between breeding and wintering areas. Seabirds that forage primarily on the open ocean are of particular interest because they have the broadest distribution within the Study Area and thus the greatest exposure to Action Proponent activities. Seabirds are a diverse group that are adapted to living in marine environments and use coastal (nearshore) waters, offshore waters (continental shelf), or open-ocean areas (Enticott & Tipling, 1997; Harrison, 1983).

Bats include resident and migratory species (National Park Service, 2017). Although all bats are terrestrial, many species occur above coastal (nearshore) waters, offshore waters (continental shelf), or open-ocean areas while migrating or foraging and will use islands, ships, and other offshore structures and vessels as stopover sites for resting or roosting (Constantine, 2003; Cryan & Brown, 2007; Pelletier et al., 2013; Thompson et al., 2015; U.S. Department of Energy, 2016).

F.7.1.1 Group Size

A variety of bird group sizes may be encountered throughout the Study Area, ranging from individual birds to thousands of birds in single-species and mixed-species flocks. Group size varies with species, location, time of day, time of year, and weather conditions. During spring and fall periods, diurnal and nocturnal migrants occur as pairs, small or large groups migrating over open ocean and at stopover areas and wintering grounds (Elphick, 2007).

No communal roosts or other large concentrations of bats are known within the Study Area. Bats could occur in the Study Area as individuals or in a small group foraging or migrating and this presence would vary with season, location, time of day, weather, as well as among species (Constantine, 2003; Dowling, 2018; Pelletier et al., 2013; Solick & Newman, 2021; U.S. Department of Energy, 2016).

F.7.1.2 Habitat Use

In the Study Area, birds forage in and migrate through a variety of habitats including the open ocean and nearshore waters including estuaries, coastal wetlands, kelp beds, lagoons, and in the intertidal zone. When and where birds occur varies by species and is influenced by environmental factors, prey location, and time of year. Other species, like neotropical and other migrants, utilize the Study Area to transit between their wintering and breeding grounds (Elphick, 2007; Shackelford et al., 2005). Location and timing of this foraging and migration is highly dependent on species.

Bats are wide-ranging, occurring on every continent except for Antarctica. While all bats are terrestrial, numerous studies have shown that many species forage or migrate over marine environments, sometimes at considerable distances from shore. True et al. (2023) tagged and tracked Eastern red bats on the mid-Atlantic coast during fall mating and migration season. They found that although bats transited long distances over open water (across Chesapeake and Delaware Bays), long distance movements tended to be landward rather than along the coast and flights occurred in early evening hours in favorable conditions. Hatch et al. (2013) reported that bats observed were located between 16.9 and 41.9 km from shore (with an average distance of 30 km). Several North American bats have been found on Bermuda, located approximately 670 mi. (1,078 km) from the coast of the U.S. (Constantine, 2003; Pelletier et al., 2013). Thompson et al. (2015) reported a large flock of little brown bats (*Myotis lucifugus*) roosting on a ship and buoys approximately 68 mi. (110 km) off the coast of Maine during optimal summertime conditions, with warm air and no wind. While resident bats occur in marine environments, migratory bats – particularly long-distance migratory bats – are the most likely species to be observed in the Study Area (Bureau of Ocean Energy Management, 2013; Pelletier et al., 2013; U.S. Department of Energy, 2016). One study found that the Eastern red bat (*Lasiurus borealis*) (73 percent of all occurrences) and hoary bat (*Lasiurus cinereus*) (22 percent of all occurrences) were the most likely species to be detected at buoy monitoring sites (U.S. Department of Energy, 2016), perhaps because they prefer open areas (Tetra Tech Inc, 2016d). Occurrence in a given area over the open ocean, however, is infrequent and seasonal, occurring most frequently during summer, particularly when the air is warm, the humidity is high, the wind speed is low, and when near forested land (Ahlén et al., 2009; Bureau of Ocean Energy Management, 2013; Johnson et al., 2011; U.S. Department of Energy, 2016).

Mexican free-tailed bats (*Tadarida brasiliensis mexicana*) have been detected flying over 3,000 m above the ground, presumably in pursuit of food (McCracken et al., 2021). Several studies have shown that bats typically fly close to the water's surface (e.g., lower than 10 m above sea level) when flying over water (Pelletier et al., 2013). However, many of these studies have had a limited ability to detect bats migrating at higher altitudes. Aerial surveys for bats, using high-definition video cameras mounted on a small aircraft at 610 m off the mid-Atlantic coast, revealed that, "of the six bats observed during aerial surveys for which flight height was estimable, all six were at altitudes over 100 m above sea level and five of the six were over 200 m above sea level," (Hatch et al., 2013).

Solick and Newman (2021) provided a synthesis of existing literature and data on the presence of bats offshore. They note the limits of existing data, as evidenced by the prevalence of observations occurring in daytime when bats are not generally active, as well as the spatial limits of acoustic monitoring. In the existing literature, long-distance migratory species, such as Eastern red bats, silver-haired bats, and hoary bats, are most commonly observed offshore but other species that do not migrate over long distances (e.g., tricolored [*Perimyotis subflavus*] and big brown bat [*Eptesicus fuscus*]) have also been detected (Bort et al., 2023; Smith et al., 2022a). Species are most frequently detected during autumn migration periods (August through October) and the lack of observations at other times of the year suggests that migratory species use coastal areas primarily during this time (Solick & Newman, 2021). Bats were detected primarily as individuals but some groups of dozens to hundreds have been observed. Dowling (2018) observed six species of bats on Martha's Vineyard and documented offshore movements by the little brown bat and Eastern red bat including migrating and foraging over Vineyard Sound. Dowling also tracked coastal movements of long-distance migrant species, Eastern red bat and hoary bat, and observed island to island movement and movement along the coast over the ocean, influenced by time of day, season, and weather. The U.S. Department of Energy (2016) monitored bats on islands, offshore structures and coastal sites in the New England Gulf of Maine, mid-Atlantic coast, and Great Lakes to better understand when and where bats occur offshore (greater than 3 NM from the coast). They observed that the highest bat activity levels occurred at coastal sites near forest edges and varied seasonally across all areas and across all species, with detections peaking in early fall, and with temperature and wind speed (activity occurring disproportionately during warmer temperatures and lower wind speeds). Finally, a single tricolored bat, not known to be a long-distance migrant, was observed in August 2018 more than 103 km off the coast of North Carolina when it landed on a boat that was conducting marine mammal observations (Bort et al., 2023).

An offshore bat acoustic survey performed by the Bureau of Ocean Energy Management in support of an EIS prepared for Offshore Wind Development in Coastal Virginia (Bureau of Ocean Energy Management Office of Renewable Energy Programs, 2022) found that *Myotis* and other cave bat species activity in the mid-Atlantic decreases 6 mi. (20 km) from shore. The survey results indicated low levels of bat activity across seasons, with activity peaking in the fall migration period under specific conditions such as low wind, good visibility, and high temperatures. Bats were documented day and night roosting on the vessels in the offshore project area.

The U.S. Department of Energy (2016) conducted acoustic surveys of bat activity on islands, offshore structures, and coastal sites in the New England Gulf of Maine, mid-Atlantic coast, and Great Lakes regions from 2012 to 2014. Bat activity patterns were highly seasonal in all regions and across species, with detection rates peaking in late summer and early fall and a high percentage of bat passes occurring within a limited number of nights. The *Myotis* genus was the most frequently detected species group, representing 43 percent of all identified bats. At most sites, bat activity increased rapidly during the first hour past sunset and then declined steadily for the rest of the night.

F.7.1.3 Dive Behavior

Many of the seabird species found in the Study Area forage in the water column. Some species dive, skim, or grasp prey at or just below the water's surface (Cook et al., 2011; Jiménez et al., 2012). Other seabirds, including diving ducks, cormorants, and alcids (murres, auks, auklets, and puffins) feed on the bottom or in the water column at depths greater than 100 feet (Cook et al., 2011; Ehrlich et al., 1988). Some seabirds forage by diving more shallowly into the water column (e.g., terns, gannets). Others are considered surface divers plunging directly from the surface underwater after prey (puffins, loons, ducks). Birds able to pursuit dive use their wings and feet for propulsion through the water, exhibited by shearwaters and some petrels. Using this strategy, pursuit divers usually float on the water and dive under to pursue fish and other prey (Burger et al., 2004). Most diving species tend to forage near the surface of the water column or on the bottom in shallow water (e.g., clams, mussels, and other invertebrates) (Cook et al., 2011). Dive durations are correlated with depth and range from a few seconds in shallow divers to several minutes in alcids (Ponganis, 2015). While no bats dive into the water column to feed, the Mexican bulldog bat (*Noctilio leporinus*) forages for fish near the water surface.

F.7.1.4 Hearing and Vocalization

Several studies of seabird hearing have been published since the 2018 Final EIS/OEIS. These newer studies of long-tailed duck, common eider, and surf scoter (McGrew et al., 2022), and Atlantic puffin and common murre (Mooney et al., 2019) (support previous conclusions that birds generally have greatest hearing sensitivity between 1 and 4 kilohertz (kHz).

Although hearing range and sensitivity has been measured for many land birds, fewer studies have focused on seabird hearing and most published literature on bird hearing focuses on terrestrial birds and their ability to hear in air. A review of 32 terrestrial and marine species indicates that birds generally have greatest hearing sensitivity between 1 and 4 kHz (Beason, 2004; Dooling, 2002). Very few can hear below 20 hertz (Hz), most have an upper frequency hearing limit of 10 kHz, and none exhibit hearing at frequencies higher than 15 kHz (Dooling, 2002; Dooling & Popper, 2000). Hearing capabilities have been studied for only a few seabirds (Beason, 2004; Beuter et al., 1986; Crowell et al., 2015; Johansen et al., 2016; Thiessen, 1958; Wever et al., 1969); these studies show that seabird hearing ranges and sensitivity in air are consistent with what is known about bird hearing in general.

The hearing range of insect-eating bats in North America is 10–100 kHz. The most sensitive frequency band is 20–50 kHz, where bats can detect sounds at approximately 20 decibels re 20 micropascals (Bohn et al., 2006; Koay et al., 1997). Bats are generally unable to hear frequencies below 500 Hz. While hearing is the primary sense used by echolocating bats to forage and avoid obstacles, they use a combination of auditory and visual landmark recognition (Denzinger & Schnitzler, 2013; Gonzalez-Terrazas et al., 2016; Jensen et al., 2005; Schnitzler et al., 2003), magneto-reception (Holland et al., 2008; Holland et al., 2006; Wang et al., 2007), and spatial memory for long-distance navigation (Barchi et al., 2013; Ulanovsky & Moss, 2008, 2011; William & Williams, 1970; Williams et al., 1966). The variety of vocalizations produced by bats can be separated into two general categories: ultrasonic echolocation sounds and communication sounds. Echolocation is used while foraging and for spatial orientation (Jensen et al., 2005; Schnitzler et al., 2003). Sound detection levels are somewhat dependent on ambient noise, and bats increase the loudness of their calls, called the Lombard effect, when they encounter noise (Hage et al., 2013; Hotchkin & Parks, 2013; Luo & Wiegrebe, 2016). Simmons and

Simmons (2024) found that echolocating bats have decreased susceptibility to noise-induced temporary hearing loss as compared to other mammals. Liu et al. (2021) found that cochlear hair cells of echolocating bats provide protection against intense noise and prevent or ameliorate hearing loss.

F.7.1.5 General Threats

Approximately half of the 346 species of seabirds that depend on ocean habitats worldwide are declining (Crowell et al., 2015). Threats to bird populations in the Study Area include human-caused stressors including interactions with commercial and recreational fishing gear, predation and competition by introduced species, disturbance and degradation of nesting areas by humans and domesticated animals, noise pollution from construction and other human activities, nocturnal collisions with power lines and artificial lights, collisions with aircraft, pollution, such as that from oil spills and plastic debris, and the effects of climate change including sea level rise, more frequent and intense storm events, and marine heat waves (Anderson et al., 2007; Burkett et al., 2003; California Department of Fish and Game, 2010; Carter & Kuletz, 1995; Carter et al., 2005; Clavero et al., 2009; International Union for Conservation of Nature and Natural Resources, 2010b; Jones et al., 2023; North American Bird Conservation Initiative & U.S. Committee, 2010; Onley & Scofield, 2007; Piatt & Naslund, 1995; U.S. Fish and Wildlife Service, 2005; Waugh et al., 2012; Weimerskirch, 2004). Disease, volcanic eruptions, storms, and harmful algal blooms are also threats to birds (Anderson et al., 2007; Jessup et al., 2009; North American Bird Conservation Initiative U.S. Committee, 2009; U.S. Fish and Wildlife Service, 2005).

Beach-nesting birds are vulnerable to disturbance from people, pets, and off-road vehicles (North American Bird Conservation Initiative U.S. Committee, 2009). Feral species (primarily cats [*Felis catus*] and rats [*Rattus spp*.], occasionally pigs [*Sus scrofa*], and cattle [*Bos taurus*]) may destroy nests. Seabirds are especially vulnerable to feral species on islands where nests and populations have been devastated through predation or habitat destruction. Invasive plants can also eliminate nesting habitat on beaches (Clavero et al., 2009; North American Bird Conservation Initiative U.S. Committee, 2009).

Lighting on boats and on offshore oil and gas platforms has also contributed to bird fatalities in open ocean environments. Birds are attracted to these lights, usually in inclement weather conditions (Merkel & Johansen, 2011). Studies have examined the effects of various types of lights on migrating songbirds (Poot et al., 2008). Land-based lighting has been linked to episodes of “fallout” (grounding) involving seabirds, especially petrels, and ship-based lighting could have similar effects (Rodríguez et al., 2017).

Large-scale wind energy development in offshore areas has the potential to affect bird populations through displacement from foraging habitats and mortality to species that fly within the rotor swept zones of large wind turbines (approximately 20 m and 200 m from the surface) (Williams et al., 2015).

Natural causes of seabird and shorebird population declines include disease, storms, and harmful algal blooms (Jessup et al., 2009; North American Bird Conservation Initiative U.S. Committee, 2009; Onley & Scofield, 2007). In addition, seabird distribution, abundance, breeding, and other behaviors are influenced by cyclical environmental events, such as the El Niño Southern Oscillation and Pacific Decadal Oscillation in the Pacific Ocean (Congdon et al., 2007; Vandenbosch, 2000).

Within the last two decades, a fungal infection called white-nose syndrome emerged as a significant threat, causing declines in cave-hibernating bat species. Other threats at the hibernacula include modifications to caves, mines, and surrounding areas that change airflow and alter the microclimate within the hibernacula. Human disturbance and vandalism pose significant threats during hibernation through direct mortality and by inducing arousal and consequent depletion of bat fat reserves. Natural catastrophes can also have a significant effect during winter because of the concentration of individuals in a relatively few sites. Habitat loss and degradation and temperature and precipitation changes related to climate change affects bats on breeding grounds, as well as along migratory pathways and wintering areas. Pesticides and other contaminants, especially those directly impact the food supply of bats (U.S. Fish and Wildlife Service, 2007). Collision with moving turbine blades at wind energy facilities are to also a threat at local and regional levels (U.S. Fish and Wildlife Service, 2007, 2022c). Increased temperature to hibernacula due to any changes in the microclimate (e.g., human disturbance, climate change) can further facilitate the spread of white-nose syndrome creating potentially cumulative impacts to this species (McCoshum et al., 2023).

F.7.2 ENDANGERED SPECIES ACT-LISTED SPECIES

The following sections provide detailed species descriptions, including status and management, habitat and geographic range, population trends, predator and prey interactions, and species-specific threats for birds and bats listed under the ESA. Changes in the ESA listings and critical habitat designations since the 2018 Final EIS/OEIS include:

- listing of the black-capped petrel as endangered in 2024;
- listing of northern long-eared bat as endangered in 2023;
- proposed listing of tricolored bat as endangered in 2021; and
- proposed critical habitat for red knot in 2023.

F.7.2.1 Piping Plover (*Charadrius melodus melodus*)

The piping plover is divided into two subspecies of plovers. The piping plovers that breed on the Atlantic coast of the United States and Canada belong to the Atlantic subspecies *C. m. melodus* (U.S. Fish and Wildlife Service, 2020b) and occur within the Study Area.

F.7.2.1.1 Status and Management

The USFWS listed the Atlantic coast piping plover population as threatened under the ESA in 1985 (50 *Federal Register* 50726) and has instituted a recovery plan for this shorebird species (U.S. Fish and Wildlife Service, 1996). In 2001 and 2002, critical habitat was designated for the Great Lakes breeding population and Northern Great Plains breeding population (66 *Federal Register* 22938; 67 *Federal Register* 57638, respectively), and for all three breeding populations while on their wintering grounds (66 *Federal Register* 36038, July 10, 2001). Critical habitat for wintering plovers has been designated in coastal areas near or within the Study Area as shown in Figure 3.9-1 through Figure 3.9-4.

The USFWS designated 137 areas along the coasts of North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Texas as critical habitat for wintering populations. This critical habitat includes 1,798.3 mi. (2,891.7 km) of mapped shoreline and 165,211 acres (66,881 hectares [ha]) of mapped area along the gulf and Atlantic coasts and along interior bays, inlets, and lagoons (66 *Federal Register* 36038, July 10, 2001). In 2008 and 2009, the USFWS updated designated critical habitat for wintering piping plover populations in North Carolina and Texas, adding 2,043 acres (827 ha) in North Carolina and 139,029 acres (56,263 ha) along the Gulf Coast of Texas (73 *Federal Register* 62816, October 21, 2008; and 74 *Federal Register* 23476, May 19, 2009, respectively). Any critical habitat located above the mean high tide line is outside the Study Area, as described in Section 3.0.2 (Ecological Characterization of the Study Area).

The 2004 National Defense Authorization Act allows military installations to be excluded from critical habitat designation for endangered species under the ESA provided that the installation's Integrated Natural Resource Management Plan (INRMP) affords (1) a benefit to the species; (2) certainty that the management plan will be implemented; and (3) certainty that the conservation effort will be effective.

On Navy installations with an approved INRMP where piping plovers breed or overwinter, the Navy is exempt from critical habitat designations.

F.7.2.1.2 Habitat and Geographic Range

In the Study Area, the Atlantic breeding population of piping plovers nest and breed on coastal beaches from southern Maine to North Carolina and are primarily an inhabitant of sandy shorelines in the northeast and southeast U.S. Continental Shelf Large Marine Ecosystems (Haig & Elliott-Smith, 2004; O'Brien et al., 2006). Piping plovers nest above the mean high tide line (outside the Study Area) on coastal beaches, sand flats at the ends of sandpits and barrier islands, gently sloping foredunes (dunes parallel to the shoreline), blowout areas behind primary dunes, and washover areas cut into or between dunes (U.S. Fish and Wildlife Service, 1996). Individuals migrate through and winter in coastal areas of the United States from North Carolina to Texas and portions of Yucatan in Mexico and the Caribbean (U.S. Fish and Wildlife Service, 2009c). Different breeding populations tend to occur in different coastal wintering areas, although there is some overlap (Gratto-Trevor et al., 2012). In winter, the species is only found in coastal areas in habitats that include mudflats and dredge spoil areas and, most commonly, sandflats (Gratto-Trevor et al., 2012; O'Brien et al., 2006). Plovers appear to prefer sandflats adjacent to inlets or passes, sandy mudflats along spits (beaches formed by currents), and overwash areas as foraging habitats. Piping plover migration routes overlap with breeding and wintering habitats.

Recovery results from birds banded during the breeding season indicate that most Atlantic coast breeders winter along the southern Atlantic coast from North Carolina to Florida, although some birds have been reported to winter in Texas (Gratto-Trevor et al., 2012). Evidence suggests that most of the Great Lakes population winters south along the Atlantic coast. Both spring and fall migration routes are believed to follow the Atlantic coast (Gratto-Trevor et al., 2012). Evidence suggests that most of the threatened Northern Plains population winters on the Gulf Coast (Gratto-Trevor et al., 2012). Islands in the Caribbean, the Bahamas and West Indies, serve as important wintering habitat (U.S. Geological Survey, 2007).

F.7.2.1.3 Population Trends

A 1991 international census documented 5,482 piping plovers and a decade later in 2001 the population estimate was 5,945 piping plovers (Haig & Elliott-Smith, 2004). The current population has been estimated to be between 7,600 to 8,400 individuals (BirdLife International, 2023a). Coastal Atlantic United States populations have trended upward since listing, though some areas' breeding populations are remaining at depressed levels and showing little or no increase in size. Since its 1985 listing, the Atlantic coast population estimate has increased from 790 pairs to an estimated 1,900 pairs in 2016, and the United States portion of the population has almost tripled, from approximately 550 pairs to an estimated 1,700 pairs (U.S. Fish and Wildlife Service, 2020b). Between 1989 and 2008, the New England population increased while the other three populations experienced decreases. The New York – New Jersey population decreased by 35 percent between 2007 and 2014, the southern population decreased by 24 percent from 2016 to 2018, and the eastern Canada population decreased 35 percent from 2007 to 2017 (U.S. Fish and Wildlife Service, 2020b). Also, the Maine population declined 64 percent, from 66 pairs in the year 2002 to 24 pairs in 2008, mostly due to loss of habitat from spring storms and dune stabilization projects. More recently, numbers have declined, with 3,973 piping plovers observed during the winter census of the 2011 International Piping Plover Census, with Texas having by far the largest number of any state (2,145), and more than 1,000 piping plovers discovered wintering in the Bahamas (Elliott-Smith et al., 2015). The 2011 breeding census resulted in an estimated breeding population of at least 5,723 birds, 75 percent of which were in the United States, with a breeding population of 1,476 pairs in the Atlantic coastal states (Elliott-Smith et al., 2015). Though the abundance of the Atlantic coast plovers has reduced near-term extinction threats, geographic variation in population growth and sensitivity to survival and productivity are cause for continuing conservation concern (U.S. Fish and Wildlife Service, 2009a).

F.7.2.1.4 Predator and Prey Interactions

Feeding habitats of breeding piping plovers include intertidal portions of ocean beaches, washover areas, mudflats, sandflats, wrack lines (line of deposited seaweed on the beach), shorelines of coastal ponds, lagoons, and salt marshes (Gratto-Trevor et al., 2012; U.S. Fish and Wildlife Service, 1996). They hunt visually using a start-and-stop running method, gleaning and probing the substrate for a variety of small invertebrates (marine worms, crustaceans, molluscs, insects, and the eggs and larvae of many marine invertebrates) (Maslo et al., 2012; U.S. Fish and Wildlife Service, 1996). Foraging occurs throughout the day and at night.

Piping plovers are preyed upon by a large array of terrestrial and aerial species and have been identified as a substantial source of mortality for piping plover eggs and chicks. These predators, such as crows, gulls, raptors, raccoons (*Procyon lotor*), ghost crabs (*Ocypode quadrata*), domestic cats (*Felis domesticus*), domestic dogs (*Canis lupus familiaris*), coyotes (*Canis latrans*), the Virginia opossum (*Didelphis virginiana*), the American mink (*Neovison vison*), and the red fox (*Vulpes vulpes*) (Stantial et al., 2021).

F.7.2.1.5 Species-Specific Threats

The localized declines of the Atlantic coast piping plover population is attributed to habitat loss and degradation and increased predator populations in coastal environments (U.S. Fish and Wildlife Service, 1996). Excessive disturbance may cause the parents to flee the nest, exposing eggs or chicks to the hot sun or predators. High disturbance levels around nest sites can also result in the abandonment of nests and, ultimately, decreased breeding success (Cohen & Gratto-Trevor, 2011). Causing parents or juveniles to flush while foraging may stress juveniles enough to negatively influence critical growth and development. The birds will flush in the presence of dogs, and often react to dogs at farther distances than they do with people (Mengak et al., 2019). In areas where dogs are off leash, the dogs will often chase the birds, eliciting a high stress response to the piping plover, which can negatively impact that individual’s survival (Rutter, 2016). Few areas used by wintering piping plovers are free of human disturbance, and nearly 50 percent have leashed and unleashed dog presence (U.S. Fish and Wildlife Service, 2009c).

Along the Atlantic coast, commercial, residential, and recreational development have decreased the amount of coastal habitat available for piping plovers. Trends show continued loss and degradation of habitat in migration and wintering areas due to sand placement projects, inlet stabilization, sand mining, erosion prevention structures (groins, seawalls, and revetments, exotic and invasive vegetation, and wrack removal) (U.S. Fish and Wildlife Service, 2009c). Unusual events, such as hurricanes, can impact hundreds of young-of-the-year and adults. Storms can also, over time, positively impact local piping plover populations by leveling dunes and creating suitable nesting habitat (U.S. Fish and Wildlife Service, 1996). Beach development and stabilization activities, dredging, recreational activities, and pollution are factors that impact the plover population on wintering grounds (U.S. Fish and Wildlife Service, 1996). There are also unknown sources of mortality experienced during migration or on the wintering grounds (Calvert et al., 2006; Root et al., 1992). Recent data suggest that lighting on vessels and on offshore oil and gas platforms may cause mortality and could help explain some of these unknown mortality events (Merkel & Johansen, 2011). New potential threats include wind turbine development projects that introduce the possibility of collision, disturbance, and displacement of plovers (Burger et al., 2011). Another threat is climate change resulting in sea level rise that would directly impact Atlantic coast piping plovers breeding and wintering habitat (U.S. Fish and Wildlife Service, 2009c).

F.7.2.2 Red Knot (*Calidris canatus rufa*)

Red knots are found on the Atlantic coast of the United States and Canada. They belong to the subspecies *C. canutus rufa* (Cornell Lab of Ornithology, 2013). This subspecies of red knot, referred to as the rufa red knot, is listed as threatened under the ESA.

F.7.2.2.1 Status and Management

Four petitions to emergency list the red knot have been submitted since 2004, and in December of 2014, the USFWS listed the red knot as threatened under the ESA (79 *Federal Register* 73706). Critical habitat was proposed in 2021 (86 *Federal Register* 37410), and the comment period was reopened in 2023 to consider additional areas (88 *Federal Register* 22530). Proposed Critical Habitat includes coastal areas near or within the Study Area as shown in Figure 3.9-5 through Figure 3.9-8.

The physical and biological features for the rufa red knot proposed critical habitat include: (1) beaches and tidal flats used for foraging; (2) upper beach areas used for roosting, preening, resting, or sheltering; (3) ephemeral and/or dynamic coastal features used for foraging or roosting; (4) ocean vegetation deposits or surf-cast wrack used for foraging and roosting; (5) intertidal peat banks used for foraging and roosting; (6) features landward of the beach that support foraging or roosting; and (7) artificial habitat mimicking natural conditions or maintaining the physical or biological features (1) through (6) (88 *Federal Register* 22530).

The key elements of the species recovery plan include: (1) conserving populations of red knots among the four Recovery Units; (2) improve the resiliency of the Southern and Western Gulf of Mexico/Central America/Pacific Recovery Units; (3) protect crucial nonbreeding habitats for adults and juveniles; and (4) monitoring breeding habitats while tracking reproductive success. Protection of the Southern Recovery Unit is important as this unit migrates the longest distance making them more vulnerable to threats. The Western Gulf of Mexico/Central America/Pacific Recovery Unit is the smallest unit and use noncoastal habitats during their migration (U.S. Fish and Wildlife Service, 2023b). The Western Hemisphere Shorebird Reserve Network has established an international network of wetlands in an effort to protect important sites used by shorebirds, including the red knot (Tsipoura & Burger, 1999). Additionally, efforts to develop protection for Delaware Bay, an important migration staging area for red knots, are underway by the Western Hemisphere Shorebird Reserve Network (Cornell Lab of Ornithology, 2013).

F.7.2.2.2 Habitat and Geographic Range

The species breeds on the central Canadian arctic and migrates to four wintering regions: (1) the southeast United States and through the Caribbean; (2) the western Gulf of Mexico from Mississippi through Central America; (3) northern Brazil and extending west along the northern coast of South America; and (4) Tierra del Fuego at the southern tip of South America (mainly in Chile) and extending north along the Patagonian coast of Argentina (U.S. Fish and Wildlife Service, 2020d). Red knots show very high fidelity to each of the four wintering regions. Red knots will briefly use important stopover areas such as the Delaware Bay to forage before returning to their breeding grounds each year. An interior red knot population winters in Texas and Louisiana and migrates through the west and Midwest to central Canada.

Red knots migrate some of the longest distances known for birds, with many individuals annually flying more than 19,000 mi. (30,000 km) (U.S. Fish and Wildlife Service, 2020d), during which they may cross over each of the open-ocean areas in the Study Area. However, outside of migration they are typically found in nearshore habitats along coastlines. Fall migration peaks in August with birds flying south along the Atlantic coast to major wintering grounds on the coasts of Argentina and southern Chile (Cornell Lab of Ornithology, 2013). During migration stopovers, the red knot uses marine habitats and generally prefers coastal, sandy habitats near tidal bays, inlets, and estuaries for foraging (Cornell Lab of Ornithology, 2013). Red knots migrate in large flocks and stop over at the same coastal sites along the Atlantic coast during spring migration to feed on eggs of horseshoe crabs (*Limulus polyphemus*). In particular, Delaware Bay is one of the largest known spring (mid-May to early June) stopover sites for this species (71 *Federal Register* 53756, September 12, 2006) (Clark et al., 1993). Up to 80 percent of the entire estimated red knot population has been observed at once in the Delaware Bay during spring migration, leading to the area being designated as the first hemispheric site in the Western Hemisphere Shorebird Reserve Network (Clark et al., 1993; Niles et al., 2008; Tsipoura & Burger, 1999).

During fall and spring migration and winter months, red knots occur in nearshore coastal habitats, along the Atlantic and gulf coasts from southern New England to Florida and into the Gulf of Mexico (Cornell Lab of Ornithology, 2013). The Virginia Atlantic barrier islands are a second major stopover location, with red knot peak counts between 5,500 and 9,100 birds since 1995 (Niles et al., 2008). They primarily occur in intertidal surf zone habitats, particularly near coastal inlets, estuaries, and bays.

F.7.2.2.3 Population Trends

The red knot population was previously estimated at 100,000 to 150,000 individuals in the 1980s (Niles et al., 2008). However, annual aerial and ground surveys of Delaware Bay show fluctuation but generally a downward trend. Population surveys during the stopover period in the spring of 1998 at Delaware Bay estimated 50,000 red knots. In 2004, the same survey was repeated, and the estimated population was substantially lower at 18,000 (Niles et al., 2008). Surveys of red knots at both migration stopover sites and wintering grounds continually show substantial population declines in recent decades (71 *Federal Register* 53756, September 12, 2006). For example, surveys during the mid-1980s of wintering red knot populations in South America (Argentina and Chile) provided an estimate of 67,500 individuals (Niles et al., 2008); but according to the USFWS, since 2005, numbers have been under 20,000 birds, and dipped below 10,000 in 2011. Recent wintering counts are similarly low with just over 13,000 individuals observed in 2019 and less than 12,000 in 2020 (U.S. Fish and Wildlife Service, 2020d). Studies from 1994 to 2002 also show decreased annual adult survival rates related to these population declines (Niles et al., 2008).

F.7.2.2.4 Predator and Prey Interactions

Red knots forage by surface pecking and probing for intertidal invertebrates and various species of mussels and other molluscs (Cornell Lab of Ornithology, 2013). During spring migration, a major food source for red knots are horseshoe crab eggs; millions of which can be found in the Delaware Bay during the second half of May (Botton et al., 1994). Red knot migration coincides with the horseshoe crabs laying their eggs, allowing birds to restore their fat reserves to continue their northward migration to their breeding grounds in the Arctic (Cornell Lab of Ornithology, 2013; Tsipoura & Burger, 1999).

Outside of the breeding grounds, red knot predators include the peregrine falcon (*Falco peregrinus*), merlin (*Falco columbarius*), northern harrier (*Circus cyaneus*), short-eared owl (*Asio flammeus*), great black-backed gull (*Larus marinus*), and accipiters (*Accipiter* spp.) (Niles et al., 2008). Predators on breeding grounds include arctic fox (*Alopex lagopus*), long-tailed jaeger (*Stercorarius longicaudus*), and parasitic jaeger (*Stercorarius parasiticus*) (Piersma et al., 1993).

F.7.2.2.5 Species-Specific Threats

The red knot is threatened under the ESA mainly by habitat loss and degradation of foraging resources such as reduction of horseshoe crab populations (U.S. Fish and Wildlife Service, 2010c). Horseshoe crabs are harvested for their blood for biomedical research and their eggs for bait in the conch and eel fishing industries; consequently, the reduction in the amount of horseshoe crab eggs available for red knots, especially in Delaware Bay, is believed to be the cause of lower weight gain in red knots during migratory stopovers and contributing to lower adult survival (Niles et al., 2008). Beach erosion, shoreline protection and stabilization projects, human disturbance, limited food resources, oil spills, red tides, hunting, and severe weather all threaten the stability of the population (Niles et al., 2008; U.S. Fish and Wildlife Service, 2010c). Because large percentages of the entire population gather at single sites during migration (i.e., Delaware Bay) and winter, the species is especially vulnerable to loss of key resources at these sites (Clark et al., 1993; Cornell Lab of Ornithology, 2013; Niles et al., 2008).

F.7.2.3 Roseate Tern (*Sterna dougallii*)

Five subspecies of the roseate tern have been described, though some taxonomic designations are uncertain: *S. d. dougallii* in the North Atlantic, Europe, and the Caribbean; *S. d. korustes* in India, Sri Lanka, and Burma; *S. d. gracilis* in Australia and Indonesia; and *S. d.arideensis* on the Seychelles Islands (Cornell Lab of Ornithology, 2014). All subspecies are similar in appearance to *S. d. dougallii*, with slight differences in wing length and bill color. The North Atlantic and Caribbean population of *S. d. dougallii* is the subspecies that occurs within the Study Area (U.S. Fish and Wildlife Service, 2010a).

F.7.2.3.1 Status and Management

In 1987, the USFWS listed the roseate tern as endangered under the ESA along the Atlantic coast of the United States (Maine to North Carolina); in Canadian provinces of Newfoundland, Nova Scotia, and Quebec, as well as in Bermuda (52 *Federal Register* 42064). The species is listed as threatened under the ESA in the Western Hemisphere, including Florida, Puerto Rico, and the Virgin Islands (52 *Federal Register* 42064). No critical habitat has been designated for this species in the United States. In 2006, Canada designated critical habitat for the species (U.S. Fish and Wildlife Service, 2010a). Recovery and management plans have been implemented to protect breeding colonies, foraging areas, and wintering grounds(U.S. Fish and Wildlife Service, 1993, 1998). The plans intend to increase breeding population size, distribution, and productivity by maintaining, expanding, and enhancing nesting habitat (U.S. Fish and Wildlife Service, 1998). Recovery and management methods include posting nesting areas with signs and fencing, discouraging and controlling competing gull species, managing vegetation to enhance nesting habitat, and attempting to attract individuals to historically occupied sites (U.S. Fish and Wildlife Service, 1998).

F.7.2.3.2 Habitat and Geographic Range

Roseate terns arrive at their breeding grounds in late April and early May (early to mid-May in the Caribbean population) and spend approximately 2 weeks feeding before they occupy nesting grounds (U.S. Fish and Wildlife Service, 1998). Northeastern roseate terns migrate in late August and early September, traveling in groups through the eastern Caribbean and along the north coast of South America to wintering grounds along the northern and eastern South American coast (Cornell Lab of Ornithology, 2014; Kirkham & Nettleship, 1987; National Audubon Society, 2017; U.S. Fish and Wildlife Service, 1998, 2010a). The migratory pathway of Caribbean birds is not known, but the route is almost certain to be 2,000 to 4,000 km (1,243 to 2,485 mi.) shorter than the route taken by the northeastern population (U.S. Fish and Wildlife Service, 2010a).

Roseate terns are colonial breeders. The Caribbean population of roseate tern breeds from the Florida Keys through the West Indies to islands off Central America and northern South America (U.S. Fish and Wildlife Service, 1993). In the Study Area, the Gulf of Mexico and Caribbean Sea Large Marine Ecosystems contain the population in the Florida Keys and Dry Tortugas, and Puerto Rico. The North Atlantic populations are known to nest on a limited number of small islands off New York and Massachusetts, while the Caribbean population similarly nests in Puerto Rico, the Dry Tortugas, and the

Florida Keys, as well as other non-U.S. affiliated Caribbean islands (Cornell Lab of Ornithology, 2014). They nest on islands near or under cover, such as vegetation, rocks, driftwood, and even human-made objects. They have also been documented nesting on sand dunes found at the end of barrier beaches (U.S. Fish and Wildlife Service, 1998). North American roseate terns use moderately to heavily vegetated sites for nesting (Burger & Gochfeld, 1988). Unlike the northeastern population, Caribbean roseate tern nests are exposed, occurring vegetation or rocks, on open sandy beaches, narrow rock ledges close to the water line, or among coral rubble (U.S. Fish and Wildlife Service, 1993). The exposure of these nests leaves them vulnerable to flooding from large storms, which has been documented to occur on Pelican Shoal in the Florida Keys (U.S. Fish and Wildlife Service, 2022d).

Within the Study Area, North American roseate terns occur within open-ocean areas (Gulf Stream and North Atlantic Gyre) more often during migration and staging for migration than during winter or the breeding season. Between May and September, small numbers of common and roseate terns are widely distributed at sea, southeast of Cape Cod and throughout the Gulf of Maine, east to the southeast edge of Georges Bank. Flocks of terns, including roseate terns, have been observed resting on the sea. Such occurrences at sea are typically associated with the occurrence of predatory fish (e.g., tuna) that drive prey species to the surface (U.S. Fish and Wildlife Service, 2010a). Roseate terns have been observed to forage up to 50 km for nest sites off the northeast coast of the United States (Bureau of Ocean Energy Management, 2019).

Most breeding North American roseate terns occur in this large marine ecosystem from late April/early May to late August/early September (Table 3.9-1). Approximately 90 percent of the northeast population breeds at two large colonies on Great Gull Island, New York; and Bird and Ram Island, Massachusetts (U.S. Fish and Wildlife Service, 2020c). Sand flats and beaches of southeastern Massachusetts, particularly along outer Cape Cod and nearshore islands provide important roosting and loafing habitats during fall staging. The Nantucket Shoal between the Massachusetts mainland and the islands of Martha’s Vineyard and Nantucket is a particularly important foraging area for the entire northeastern population (U.S. Fish and Wildlife Service, 2010a). Wintering North American roseate terns occur along the southeast Atlantic and gulf coasts (U.S. Fish and Wildlife Service, 2010e). The Caribbean population of roseate tern breeds from the Florida Keys through the West Indies to islands off Central America and northern South America (U.S. Fish and Wildlife Service, 1993). Within the Study Area, the Gulf of Mexico and Caribbean Sea Large Marine Ecosystems contain the population in the Florida Keys and Dry Tortugas, and Puerto Rico.

F.7.2.3.3 Population Trends

The estimated global population of roseate terns is approximately 200,000 to 220,000 (BirdLife International, 2023b). They are a widespread species that breed on every continent except Antarctica, with populations in the Indian Ocean, Caribbean, Australasian, European, African, and North American regions (Gochfeld, 1983). Approximately 4,374 pairs are estimated in the northeast U.S. population, with an additional 63 pairs in Canada (U.S. Fish and Wildlife Service, 2020c). Within the Caribbean population, approximately 1,000 pairs occur in Puerto Rico, with an estimated 500 to 2,300 pairs in the U.S. Virgin Islands (Cornell Lab of Ornithology, 2014). The roseate tern experienced drastic declines in the late nineteenth century due to commercial hunting of feathers for the millinery (hat-making) industry (U.S. Fish and Wildlife Service, 1998), as well as humans seeking eggs for food (Kirkham & Nettleship, 1987). Populations again showed decline in the 1940s and 1970s as the geographic range and the number of breeding colonies decreased (U.S. Fish and Wildlife Service, 1998).

Groups of roseate terns can be small due to their limited population size and limited nesting habitat in North America. In the northeast, breeding colonies of roseate terns range from 2 to more than 1,000 pairs, depending on breeding colony location (U.S. Fish and Wildlife Service, 1998). After chicks fledge from their breeding colonies, terns tend to congregate in large numbers at post-breeding staging areas to build up energy reserves for their seasonal fall migration to South America (U.S. Fish and Wildlife Service, 2010a). Northeastern roseate terns are always mixed with gulls and other species of terns, while populations in the Caribbean and the Seychelles Islands are known to form single-species colonies (Cornell Lab of Ornithology, 2014). Duffy (1986) found that roseate terns foraging in smaller flocks experienced higher survival rates, while in larger groups they were often out-competed by common terns.

F.7.2.3.4 Predator and Prey Interactions

The roseate tern is a coastal species that forages for small schooling fishes over shallow waters around bays, channels, sandbars, shoals, and reefs (Cornell Lab of Ornithology, 2014; Nisbet & Spendelow, 1999). They are also known to forage out over deeper waters than other tern species (Olsen & Larsson, 1995). Local commutes of up to 16 mi. (25 km) from nesting grounds to dependable foraging sites have been documented (Nisbet & Spendelow, 1999). During chick provisioning flights, breeding roseate terns have been observed foraging over 30 mi (50 km) offshore (Loring et al., 2019). Roseate terns generally concentrate in areas where prey is available close to the surface, driven there either by water movements or larger predatory fish.

Roseate terns are specialized aerial plunge-divers that often completely submerge themselves when seizing fish (U.S. Fish and Wildlife Service, 2010e). Roseate terns tend to plunge from heights above the water's surface ranging from 3 to 20 ft., although plunges from greater than 39 ft. have been observed (Cornell Lab of Ornithology, 2014). Roseate terns do not plunge deep into the water column, usually less than 3 ft. Given the shallow depth of dives, submergence times of roughly 1-2 seconds can be anticipated. Roseate terns will often fly into the wind and hover (a behavior known as "kiting") with rapid wingbeats and then, with accelerated flapping, aerial plunge into the water (Kaufman, 1990; U.S. Fish and Wildlife Service, 1998). Prey species are herring, mackerel, anchovies, and sand eels (Cornell Lab of Ornithology, 2014).

Roseate tern eggs and young are preyed upon by hermit and land crabs, ants, snakes, other birds (e.g., hawks, owls, gulls, and some shorebirds), and mammals such as rats and feral cats (U.S. Fish and Wildlife Service, 1993).

F.7.2.3.5 Species-Specific Threats

Roseate tern population declines have been attributed to commercial hunting and egg collection, habitat loss and disturbance, organochlorine contamination, predation, and competition from gulls (U.S. Fish and Wildlife Service, 1998). These threats, combined with the small number of breeding sites used by the species, warranted the listing of the species (Nisbet & Spendelow, 1999). Roseate terns are sensitive to disturbance on their nesting grounds, and many suitable nesting sites have been lost or abandoned due to the expansion of recreational, residential, and commercial use (Gochfeld, 1983). Beach erosion and the expansion of gull populations have also displaced roseate terns from suitable nesting habitat (Cornell Lab of Ornithology, 2014). Roseate terns are vulnerable to predation and flooding because they nest on the ground, often in low-lying areas (Gochfeld, 1983). Storms and prolonged periods of cold, wet weather also impact nest success (U.S. Fish and Wildlife Service, 1993). Climate change and sea level rise may exacerbate erosion of nesting grounds and could result in more severe or more frequent storms, which could disturb these habitats and result in reduced survival of adults, eggs, chicks, and fledglings (U.S. Fish and Wildlife Service, 2010a). Starvation is likely a greater cause of death during the winter in areas such as the southern Caribbean where nutrients are relatively poor (Gochfeld, 1983). Although little is known about roseate tern ecology during migration and wintering periods, one major cause of death is believed to be humans hunting this species on its wintering grounds (outside the United States) (Cornell Lab of Ornithology, 2014). Emerging potential threats include wind turbine development projects that introduce the possibility of collision, disturbance, and displacement of this species during the breeding and migratory seasons (Burger et al., 2011).

F.7.2.4 Bermuda Petrel (*Pterodroma cahow*)

F.7.2.4.1 Status and Management

The USFWS listed the Bermuda petrel as endangered under the ESA in 1970 (35 *Federal Register* 8498). There is no designated critical habitat for this seabird species. This extremely rare seabird nests only in Bermuda in the Atlantic Ocean (White, 2004). The Bermuda petrel was thought to be extinct for about three decades until its existence was confirmed in the mid-1900s. In 1951, 18 pairs of the Bermuda petrel (commonly referred to as “cahow”) were rediscovered breeding on a group of four rocky islets in Castle Harbor, Bermuda. An intensive recovery and management program followed, which included removing predators, such as rats (Murphy & Mowbray, 1951), and adapting nest burrow entrances with baffles and artificial burrows to prevent nest site competition with the white-tailed tropicbird (*Phaethon lepturus*) (Murphy & Mowbray, 1951). Efforts to establish a new breeding colony in the higher areas of Nonsuch Island Nature Reserve have been slow but promising (Dobson & Madeiros, 2009). The total population is estimated at approximately 250-275 individuals with 71 breeding pairs in 2005, 96 breeding pairs in 2009 (Dobson & Madeiros, 2009), and 101 breeding pairs in 2012 (U.S. Fish and Wildlife Service, 2013). Based on the USFWS 5-year review of the status of Bermuda petrel, the total population of the cahow is estimated as 250-275 individuals (U.S. Fish and Wildlife Service, 2019).

F.7.2.4.2 Habitat and Geographic Range

The Bermuda petrel is a pelagic species and spends most of its life at sea, except during the breeding season from January to June where it comes ashore to breed. Breeding occurs outside the Study Area, exclusively in Bermuda on five small islets off Nonsuch Island in the North Atlantic Gyre (National Audubon Society, 2005). Available islet nesting habitat is limited to 2.4 acres (0.97 ha), which is occupied by a varying number of breeding pairs each year (BirdLife International, 2008b). During the breeding season, the Bermuda petrel arrives and leaves the island only at night to avoid predation (Wurster & Wingate, 1968). During the breeding season, the Bermuda petrel nests in colonies, but is otherwise solitary (Onley & Scofield, 2007). Due to its solitary behavior the Bermuda petrel is unlikely to approach ships (Enticott & Tipling, 1997; Onley & Scofield, 2007). More specific nest density or colony size information was not found.

Raine et al. (2021) deployed GPS tags on 6 chick-rearing adults in April 2019. The maximum distance birds flew from Bermuda during foraging trips ranged from 61 to 2,513 km. Results indicated that foraging habitat exists beyond the core home range of the population, as far north as the highly productive Gulf Stream frontal system, and within the territorial waters of both the U.S. and Canada.

In the nonbreeding season (June–December) (Brooke, 2004), the species migrates from the breeding grounds in Bermuda to foraging routes over much of the Atlantic Ocean, including waters of the North Atlantic Gyre and the Gulf Stream (includes off-shelf portions of the Virginia Capes and Navy Cherry Point Range Complexes) (Lee & Mackin, 2008; National Audubon Society, 2005; Onley & Scofield, 2007). However, dispersal and at-sea distribution are generally poorly known (Brooke, 2004; Onley & Scofield, 2007). One additional migration route was recorded into the northeast Atlantic, off the coast of southwestern Ireland (Dobson & Madeiros, 2009). First reported off North Carolina’s Outer Banks in April 1983 (Lee, 1987), today the species regularly occurs off the North Carolina coast (National Audubon Society, 2005; White, 2004). Recent data recorded during the nonbreeding season documented western routes to the Gulf Stream and northern movements to the Bay of Fundy, into the

Gulf of St. Lawrence, and over the Grand Banks. An additional route was recorded off the coast of southwestern Ireland (Madeiros, 2009).

F.7.2.4.3 Population Trends

The Bermuda petrel is an extremely rare seabird that is slowly but steadily increasing: 18 pairs were recorded in the year 1951; 70 pairs raising 40 young were recorded in 2003; 71 pairs raising 35 young were recorded in 2005 (International Union for Conservation of Nature and Natural Resources, 2010a). The reproductive output between 2000 to 2001 and 2007 to 2008 ranged from 29 to 40 fledglings per year (Madeiros et al., 2012). Conservation efforts continue and the species is recovering in number, with the population estimated at 250-275, with 101 breeding pairs as of 2012 (U.S. Fish and Wildlife Service, 2013). In 2021, 143 breeding pairs were recorded, producing a total of 71 fledged chicks (Bermuda Government Department of Environment and Natural Resources, 2021).

F.7.2.4.4 Predator and Prey Interactions

Bermuda petrels feed mostly on squid, but their diet also consists of shrimp and small fish (National Audubon Society, 2005). Specific information on the feeding behavior of Bermuda petrels is lacking, but petrels of the genus *Pterodroma* often land on the ocean surface where they scavenge or grab prey; they also feed on the wing (while flying), where they are able to catch flying fish (Onley & Scofield, 2007).

Maximum dive depths for several species of *Pterodroma* petrels in New Zealand were determined from depth gauges that were attached to individual birds and recovered after varying lengths of time during which the birds were foraging at sea (Taylor, 2008). Mean maximum dive depths ranged from 1.1 to 4.7 m, with a maximum depth recorded of 23 m. Maximum dive depths were similarly determined for the Providence petrel (*Pterodroma solandri*), an Australian species, and found to average 2.9 m (Bester et al., 2011). It is reasonable to conclude that in addition to feeding at the surface, petrels of the genus *Pterodroma,* (probably including the Bermuda petrel) frequently engage in surface plunging or pursuit diving to reach prey several meters below the surface. No data are available on submergence times, but to reach these depths presumably requires a petrel to be underwater for roughly 5 to 10 seconds.

Predator information for Bermuda petrels is also not well understood but based on live webcams that have been monitoring nests of the petrels since 2011, some potential predatory behavior can be observed on the eggs. Land crabs have been observed eating inviable eggs and individuals of competing petrel species have also been observed lurking in or around active Bermuda petrel nest sites (Kannan et al., 2021).

F.7.2.4.5 Species-Specific Threats

Current threats to this species include habitat loss; competition for nest sites with the white-tailed tropicbird (Dobson & Madeiros, 2009); egg failure from contaminants (Brooke, 2004; Wurster & Wingate, 1968); light pollution from a nearby Bermuda airport; sea level rise; and increasing frequency and magnitude of tropical storms and hurricanes, which destroy nests through erosion, wave damage, and flooding (BirdLife International, 2008a, 2008b; Dobson & Madeiros, 2009; Madeiros et al., 2012; U.S. Fish and Wildlife Service, 2013). The USFWS 5-year review of the status of Bermuda petrel lists the following threats: hurricane erosion of the breeding islands, sea level rise, physical constraints of the present nesting areas, natural and introduced predators, nest site competition from white-tailed tropic bird (*Phaethon lepturus*), and nest predation by rats and avian predators (U.S. Fish and Wildlife Service, 2019).

F.7.2.5 Black-Capped Petrel (*Pterodroma hasitata*)

F.7.2.5.1 Status and Management

The black-capped petrel was listed as endangered in January of 2024 (88 *Federal Register* 89611). Critical habitat for the black-capped petrel is now identified to be prudent but is not determinable currently. The USFWS anticipates publishing proposed critical habitat for the black-capped petrel in 2024 (U.S. Fish and Wildlife Service, 2023c). The black-capped petrel is a pelagic seabird that is endemic to the western North Atlantic whose breeding distribution is not well known. It was considered extinct in the mid-1900s but was rediscovered in 1963 when evidence was obtained which indicated that a fair-sized population was still breeding on the Massif de la Selle of southeastern Haiti (Wingate, 1964). Previous nesting area in Valle Nuevo in the Cordillera Central of the Dominican Republic was documented to support nesting only as recently as 2018 (Jodice et al., 2021). Confirmed breeding of the four known colonies is restricted to Hispaniola, in both Haiti and the Dominican Republic (U.S. Fish and Wildlife Service, 2018). It may also nest on other Caribbean islands, including Dominica, Guadeloupe, Cuba, or Jamaica (American Bird Conservancy, 2024). The actual population size of the black-capped petrel is unknown, but published estimates range from approximately 2,000 to 4,000 birds, among which are perhaps 500-1,000 breeding pairs across the species' range given data and observations over the past 10-15 years (Simons et al., 2013). Wingate's 1964 estimate compared to Simons' et al. 2013 estimate suggests a decline in abundance of approximately 50–75 percent over the last 50 years (U.S. Fish and Wildlife Service, 2018).

F.7.2.5.2 Habitat and Geographic Range

The black-capped petrel spends most of its life at sea, ranging over the Caribbean and into the Gulf of Mexico and Atlantic Ocean, from eastern U.S. waters to northeastern Brazil (American Bird Conservancy, 2024). Black-capped petrels are highly pelagic and undertake long-distance foraging trips. Foraging by Black-capped petrels occurs mainly in flocks with observed feeding flocks comprised also of other avian species (Simons et al., 2013). A collection of at-sea observations indicates that waters in or adjacent to the Florida Current and the Gulf Stream between northern Florida and southern Virginia provide a distinct and relatively confined foraging range of black-capped petrels, with concentrations observed there throughout the year (BirdLife International, 2024). The offshore region from southern Florida to Cape Hatteras, North Carolina is the only marine area where regular and sizable concentrations of the species occur (Simons et al., 2013). Most of the world's population of Black-capped petrels forages off the coast of the southeastern United States, making this area important for the survival of the species (Simons et al., 2013). At sea, the primary habitat of the black-capped petrel is generally deep waters (e.g., 200 to 2,000 m depths) (U.S. Fish and Wildlife Service, 2018). In such areas, Haney (1987) reported that black-capped petrels were most common near seamounts, submarine ridges, and other benthic landscapes that promoted vigorous current mixing and deep upwelling, which typically bring food items to the surface. Areas of the strongest and most persistent upwelling are where greatest numbers of petrels are found, especially in the deeper offshore zone near southern South Carolina and northern Georgia, and the Cape Hatteras, North Carolina region (Jodice et al., 2015). More specifically, black-capped petrels are most abundant in the immediate frontal region between current eddies (Simons et al., 2013), and relatively scarce in shallower continental shelf areas (U.S. Fish and Wildlife Service, 2018).

Black-capped petrels occur on land during the breeding season, which is estimated to be from January to July (Simons et al., 2013). Breeding occurs outside the Study Area, exclusively in southwestern Hispaniola (U.S. Fish and Wildlife Service, 2018). These nesting sites are mostly found less than 30 km from the coast (Goetz et al., 2012), although there are some accounts of nests further inland. However, little data exist on specific characteristics of historical nesting sites (U.S. Fish and Wildlife Service, 2018).

After fledglings exit from the nest burrow, the nest sites remain vacant until approximately mid-October, when the next nesting cycle begins. Active nest burrows may remain in use by the same pair year after year, although burrows are sometimes abandoned after death of one member of the pair (U.S. Fish and Wildlife Service, 2018).

The geographic distribution of breeding populations needs further study. Using radar surveys, Environmental Protection in the Caribbean (2015) detected 64-320 "petrel-like targets" across eight sites in the Dominican Republic (including three potential new nesting areas), and detections ranging from 6-1,570 individuals across an additional eight sites in Haiti. Similar radar surveys in Dominica detected a total of 968 "petrel-like targets" across 17 locations while also visually confirming at least eight individuals (Environmental Protection in the Caribbean, 2015). Most of the detections were at four locations: Morne Diablotin, Morne Anglais, Morne Micotrin, and Morne Trois Piton. Recent advances in detection methodology, specifically digital acoustic monitoring, helped McKown (2014) find evidence of approximately 60 active nest sites in the nesting areas of southwestern Dominican Republic (U.S. Fish and Wildlife Service, 2018).

F.7.2.5.3 Population Trends

The black-capped petrel has a very small, fragmented and declining breeding range and population. It has already been extirpated from some sites, and declines are likely to continue as a result of habitat loss and degradation, hunting and invasive predators (BirdLife International, 2024). Population estimates and trends for this species has uncertainties because the inherent difficulty of conducting accurate counts, particularly at breeding sites, where all current counts have been conducted (U.S. Fish and Wildlife Service, 2018). Though uncertain, estimates suggest that the numbers of breeding pairs at sites in the Dominican Republic may be between 10 to 100 (Simons et al., 2013), while those in Haiti may range from 500 to 1,500 (Goetz et al., 2012). Both Simons et al. (2013) and Goetz et al. (2012) suggested that nesting areas in Haiti may contain up to 95 percent of currently known nest sites for this species (U.S. Fish and Wildlife Service, 2018).

F.7.2.5.4 Predator and Prey Interactions

The black-capped petrel is primarily nocturnal and crepuscular, feeding on squid, fish, crustaceans, and *Sargassum* (BirdLife International, 2024). In its primary foraging range, it is most influenced by the position of the Gulf Stream, a dynamic current system, and not sea surface temperature or depth (Simons et al., 2013). Their high, arching flight pattern likely facilitates visual detection of potential feeding sites (Simons et al., 2013). Black-capped petrels may also be attracted to waste discarded from ships and fishing vessels. However, it is believed that such attraction may be greatest in times of low or unpredictable natural food abundance (Simons et al., 2013).

Primary predators to the black-capped petrel are land mammals found near nesting sites. Black rats (*Rattus rattus*), Norway rats (*Rattus norvegicus*), domestic dogs (*Canis familiaris*), feral pigs (*Sus scrofa*), domestic cats (*Felis domesticus*) and the Indian mongoose (*Herpestes javanicus*) were historically introduced following European colonization and increased predation pressure (U.S. Fish and Wildlife Service, 2018).

F.7.2.5.5 Species-Specific Threats

The black-capped petrel has threats on land and at sea that impact their population and breeding. Current threats to this species include nesting habitat degradation; harvesting of both adults and nestlings from nest burrows for human consumption; forest fires during their nesting season; predation by introduced mammals; collision with communication towers; artificial lighting; collision of wind farms on and near Caribbean islands; offshore oil and gas activities; exposure to mercury and plastic pollution; entanglement in clear monofilament fishing lines or getting caught in hooks; and climate change

(Furness, 2003; Goetz et al., 2012; Simons et al., 2013; U.S. Fish and Wildlife Service, 2018; Wheeler et al., 2021). The USFWS updated Species Status Assessment Report (U.S. Fish and Wildlife Service, 2018) explains in detail each threat to the species and their conservation efforts.

F.7.2.6 Indiana Bat (*Myotis sodalis*)

F.7.2.6.1 Status and Management

The Indiana bat was originally listed as in danger of extinction under the Endangered Species Preservation Act of 1966 and is currently listed as endangered under the ESA. In 2009, its recovery priority was changed from 8 (meaning that the species has a moderate degree of threat and high recovery potential) to 5 (meaning that the species has a high degree of threat and a low recovery potential) due to the emergence and poor understanding of white-nose syndrome (U.S. Fish and Wildlife Service, 2009b). Critical habitat was designated for the species in 1976 (41 *Federal Register* 41914–41916, September 24, 1976). Eleven caves and two mines in six states (Illinois, Indiana, Kentucky, Missouri, Tennessee, and West Virginia) were listed as critical habitat. Significant information gaps remain regarding the species' ecology that hinder sound decision-making on how best to manage and protect the species (U.S. Fish and Wildlife Service, 2007).

F.7.2.6.2 Habitat and Geographic Range

Indiana bats hibernate, typically beginning in mid-October (in northern areas) or by the end of November (in southern areas) and ending by early May (for females) or mid-May (for males), with female peak emergence in mid-April and male peak emergence early May. It is thought that spring migration, which may occur either immediately upon emergence or a few days after emergence, may cause higher mortality due to low fat reserves and food supplies. Large numbers of Indiana bats complete their migration in mid-May, and fall migration begins during the first two weeks of August (U.S. Fish and Wildlife Service, 2007). Mating occurs just prior to hibernation (U.S. Fish and Wildlife Service, 2024b).

Hibernacula are patchily distributed northeast-southwest from Vermont to Tennessee, and east-west from Tennessee to Arkansas. Between 1995 and 2005, 281 hibernacula were active for at least one year. Of these, only one county (in Connecticut) containing one Priority 4 (i.e., lowest priority) hibernacula was located along the eastern coast of the U.S., and only one county (in New Jersey) containing two Priority 3 (i.e., second-lowest priority) hibernacula are adjacent to a county located along the eastern coast of the U.S. (U.S. Fish and Wildlife Service, 2007).

Confirmed Indiana bat sightings in coastal and coastal adjacent counties since 2007, have been documented in Connecticut, New York, New Jersey, and Florida. However, white nose syndrome was introduced into bat caves in 2006 and bat mortality due to this fungus were evident as of 2007 and the fungus as spread throughout the U.S (U.S. Fish and Wildlife Service, 2024f). While coastally adjacent counties in Florida had one sighting of the Indiana bat in 2007 (Loeb & Winters, 2013), an extensive literature search suggests there has been no other recent sightings of Indiana bats in any county in Florida. Therefore, it is assumed that the species is no longer present in the state of Florida. For the other states, there is evidence of more recent presence of the Indiana bats but in New York and Connecticut, specific county details are unavailable. Thus, the 2007 data are utilized for determining coastal and coastal adjacent county presence and absence in these two states. At the Connecticut sites, the bats were seen in both winter hibernacula and summer maternity roosting. For the one New York County, this species was only seen in its winter hibernacula (Loeb & Winters, 2013). For New Jersey, there was a recent survey completed of all endangered bat species across the state and Indiana bats were found in six coastal and coastal adjacent counties. In all six counties, the species exhibited maternity roosting. In two of the six counties, hibernacula were also identified (U.S. Fish and Wildlife Service, 2023a).

Maternity colonies are generally more clustered, located along the borders of Iowa, Missouri, and Illinois as well as throughout Indiana and southern Michigan (U.S. Fish and Wildlife Service, 2007). Many colonies are present across the Eastern states, but they are mostly inland. At present, the state with the most coastal and coastally adjacent colonies of Indiana bats is New Jersey (U.S. Fish and Wildlife Service, 2023a).

F.7.2.6.3 Population Trends

Estimates of prehistoric Indiana bat populations, based on paleontological evidence, range from 1.7 million to 13 million. One analysis of bone deposits at Bat Cave, Kentucky, in Mammoth Cave National Park, revealed an estimated 300,000 Indiana bats had died during a single flood event; it is uncertain whether this catastrophic population loss occurred during prehistoric times or during a large flood in 1937 that devastated much of the Ohio River valley (U.S. Fish and Wildlife Service, 2007).

When the Indiana bat was originally listed, its range-wide population was estimated at approximately 880,000. In 1983, when the first recovery plan was completed and approved, the range-wide population was estimated at about 550,000. Despite the acquisition and protection of over 35 caves and mines by government agencies or private conservation organizations, the range-wide Indiana bat population was estimated at 353,000 bats in 1997 (U.S. Fish and Wildlife Service, 2009b). These earlier estimates are considered low, however, due to discoveries of new hibernacula. For example, one hibernaculum was discovered in Missouri in 2012 that contained a minimum of 123,000 bats when partially surveyed in January 2013 and over 167,000 bats when more completely surveyed in January 2015. Based on earlier accounts of very large numbers of unidentified bats using this hibernaculum for decades, USFWS decided to add the same number of bats as was found in 2015 (i.e., 167,000) to each previous biennium total for Missouri through 1981. Based on the best available data for the species, USFWS currently estimates that approximately 635,000 bats occurred range-wide in 2007 and that the population fell to approximately 524,000 in 2015 (U.S. Fish and Wildlife Service, 2015). The population has grown slightly based on the population estimate conducted in 2019 that identified 537,297 individuals across 223 hibernacula in 16 states (U.S. Fish and Wildlife Service, 2024b).

F.7.2.6.4 Predator and Prey Interactions

Predators of Indiana bats include snakes, owls, and racoons. However, their nocturnal behavior, flying abilities, and their ability to roots in tall structures makes predation for these species more difficult (Animal Diversity Web, 2024).

Indiana bats feed on flying insects, with only a very small amount of spiders (presumably ballooning individuals) included in the diet. Four orders of insects contribute most to the diet: Coleoptera, Diptera, Lepidoptera, and Trichoptera. Terrestrial-based prey (moths and beetles) were more common in southern studies, whereas aquatic-based insects (flies and caddisflies) dominated in the north. It is presumed that this difference indicates southern bats foraged more in upland habitats, and northern bats hunted more in wetlands or above streams and ponds. Indiana bats are also known to consume other flying insects such as Hymenopterans (winged ants) and Asiatic oak weevils (*Cyrtepistomus castaneus*) when opportunistically available (U.S. Fish and Wildlife Service, 2007).

F.7.2.6.5 Species-Specific Threats

Threats to the Indiana bat vary during its annual cycle. Within the last 20 years, white-nose syndrome emerged as a significant threat as it causes precipitous declines in populations of cave-hibernating bat species (see the 2018 Final EIS/OEIS Section 3.9.2.1.5.3, Disease and Parasites). Other threats at the hibernacula include modifications to caves, mines, and surrounding areas that change airflow and alter the microclimate within the hibernacula. Human disturbance and vandalism pose significant threats during hibernation through direct mortality and by inducing arousal and consequent depletion of fat reserves. Natural catastrophes can also have a significant effect during winter because of the concentration of individuals in a relatively few sites. During summer months, possible threats relate to the loss and degradation of forested habitat. Migration pathways and swarming sites may also be affected by habitat loss and degradation (U.S. Fish and Wildlife Service, 2007). Pesticides and other contaminants, especially those directly impact the food supply of these bats, are also threats for this species (U.S. Fish and Wildlife Service, 2007, 2024b).

F.7.2.7 Northern Long-Eared Bat (*Myotis septentrionalis*)

F.7.2.7.1 Status and Management

The northern long-eared bat was listed as threatened under the ESA on May 4, 2015, and on November 29, 2022, the USFWS published a final rule (87 *Federal Register* 73488) reclassifying the northern long-earned bat as endangered. It occurs in 37 states, the District of Columbia, and 13 Canadian provinces (U.S. Fish and Wildlife Service, 2016a). This rule came into effect March 31, 2023 (U.S. Fish and Wildlife Service, 2024a). It occurs in the District of Columbia, 13 Canadian provinces, and 37 states (U.S. Fish and Wildlife Service, 2016a), including coastal states from Maine to South Carolina (U.S. Fish and Wildlife Service, 2024d). The primary threat for this species is the fungal infection white-nose syndrome (U.S. Fish and Wildlife Service, 2024d). In January 2016, the USFWS established a white-nose syndrome zone under Rule 4(d) of the ESA. Incidental take of the northern long-eared bat is only allowed outside of the white-nose syndrome zone. The boundary of this zone is updated monthly as new data are collected and is available online at the USFWS's Midwest Region website. In addition, other issues like loss of habitat, climate change, and anthropogenic disturbance are also threats for this species (U.S. Fish and Wildlife Service, 2016a, 2024d). Data suggest that the population has declined by 49 percent range-wide and will continue to decline over the next decade (U.S. Fish and Wildlife Service, 2022f).The USFWS has not designated any specific habitat as critical habitat for the species. The USFWS determined that designating wintering habitat as critical habitat for the species would likely increase the threat of vandalism, disturbance, or the spread of white-nose syndrome. Furthermore, the USFWS has determined there are no areas within the summer habitat that meet the definition of critical habitat (U.S. Fish and Wildlife Service, 2016b).

F.7.2.7.2 Habitat and Geographic Range

Hibernation generally occurs from October through April, depending on the local climate. Suitable habitat for hibernation includes caves and cave-like structures (e.g., abandoned or active mines, railroad tunnels). The spring migration period typically runs from mid-March to mid-May. Suitable summer habitat for the northern long-eared bat consists of a wide variety of forested and wooded habitats as well as linear features such as fence rows, riparian forests, and other wooded corridors with variable amounts of canopy closure. Ultimately, these bats are flexible when it comes to roosting habitat. Any type of crack, crevice, bark, dead tree, or even the hibernaculum cave (for non-reproductive females and males) can function as roosting habitat (U.S. Fish and Wildlife Service, 2024d). Mature forests are an important habitat type for foraging northern long-eared bats (U.S. Fish and Wildlife Service, 2016a).

Unlike the true long-distance migratory bats (*Lasiurus* spp. and *Lasionycteris* spp.), the northern long-eared bat does not undertake long-distance migrations between summer and winter ranges but will make shorter distance movements between summer roosts and winter hibernacula (Yates, 2015). Within the United States, its range extends along the eastern coast from Canada to northeastern North Carolina, with additional small patches along the coast of southern North Carolina and southern South Carolina (U.S. Fish and Wildlife Service, 2017a, 2017b). Within the Study Area, northern long-eared bats are most likely to occur off the coast of the Northeastern United States and Canada (U.S. Department of Energy, 2016).

In a literature review, Pelletier et al. (2013) report that northern long-eared bats were found along the coastline or offshore on islands at:

- Kejimkujik National Park, Brier Island, and Bon Portage Island in Nova Scotia, Canada. Nova Scotia is a peninsula that is separated from the mainland to the south by 30 to 50 mi. of water. Brier Island and Bon Portage Island are separated from Nova Scotia by approximately 8 mi. and about 2 mi., respectively. Observed during summer months.
- Bay of Fundy National Park, New Brunswick, Canada, in summer to early fall.
- Martha's Vineyard, Massachusetts, approximately 4 mi. from mainland, during mist-netting surveys from April through October.
- Mount Desert Island, Maine (2 mi. off the coast), between May and September.

In addition, U.S. Department of Energy (2016) reports that ongoing mist-netting surveys at coastal sites in the northeast have also indicated relatively high numbers of northern long-eared bats post the introduction of white-nose syndrome compared to other, non-coastal areas in the northeast.

Northern long-eared bats have been detected during surveys at a variety of Navy installations along the eastern coast. These installations include:

- Naval Computer and Telecommunications Area Master Station Atlantic Detachment Cutler, located on the coast in Cutler, Maine, near the border with Canada. Data suggest there were likely some long-distance migratory tree-roosting bats spending the summer residency period at the installation and that other long-distance migratory bats moved through the Installation during the fall (Tetra Tech Inc, 2014). However, no northern long-eared bats were detected at the Installation in surveys by Yates (2015).
- Naval Weapons Station Earle in Colts Neck, New Jersey, where northern long-eared bats were present and roosting at the installation (Tetra Tech Inc, 2016c). The survey report authors note that the "presence of a sustained population of northern long-eared bats on Naval Weapons Station Earle is a testament to the amount of preferred habitat, contiguous forest, that the installation is able to provide compared to the surrounding areas."
- Naval Weapons Station Yorktown and Naval Supply Center Cheatham Annex in Williamsburg, Virginia (Tetra Tech Inc, 2017b). One bat was detected during the 2016 surveys, and a juvenile was detected during 2014 surveys. The authors report that the presence of the juvenile "suggests that there may be successful northern long-eared bat maternity colonies in the area."
- Two installations along the coast in Virginia Beach, Virginia:
    - Joint Expeditionary Base Fort Story (Tetra Tech Inc, 2016a).
    - Naval Air Station Oceana Dam Neck Annex (Tetra Tech Inc, 2016b).

In addition to the above, although no northern long-eared bats were detected at Naval Air Station Oceana in Virginia Beach, Virginia, they were detected near the installation in 2014 and 2015, and there is suitable habitat available on the installation (Tetra Tech Inc, 2016d).

A literature review for this species suggests that since 2016, the northern long-eared bat has been observed in coastal or coastally adjacent counties in Maine (U.S. Fish and Wildlife Service, 2016c), Massachusetts (Mass.gov, 2024), Connecticut (CT.gov, 2023), New Jersey (U.S. Fish and Wildlife Service, 2023a), Virginia (Virginia Department of Wildlife Resources, 2023), North Carolina (U.S. Fish and Wildlife

Service, 2021b), and South Carolina (South Carolina Department of Natural Resources, 2024). This bat species has been observed both in hibernacula and maternity roosting across its coastal range, with peak present in coastal areas within the northeast.

There is a small chance that the species could be present along the coast in Florida and Georgia. In Florida, however, the latest data suggest that the last sighting of the northern long-eared bat was at last 50 years ago (Florida Bat Conservancy, 2024). For Georgia, all current sightings of this species have been inland (U.S. Fish and Wildlife Service, 2024c).

F.7.2.7.3 Population Trends

The U.S. Fish and Wildlife Service (2016d) estimated the range-wide northern long-eared bat population at over 6.5 million adults. The Midwest supports 43 percent of the total population, followed by the southern range (38 percent), the Eastern range (17 percent), and the Western range (2 percent). Arkansas and Minnesota are the two states with the largest populations, with approximately 863,850 (13 percent) and 829,890 (13 percent) adults, respectively. In areas affected by white-nose syndrome, however, the population is likely overestimated as (1) there is a clear downward trend in these areas, (2) most data are at least a year old, and (3) three years of occupancy data were used.

Based on the winter hibernacula data, there has been a 49 percent decline in the northern long-earned bat population. There has also been a decline in the number of extant winter colonies by 81 percent and the colonies are becoming smaller. Data modeling suggests that by 2030, there will be a 95 percent decline in the population and a spatial decline of 75 percent. While the summer data support a similar trend (U.S. Fish and Wildlife Service, 2022f), the data are less certain than the winter data because of this species flexibility with roosting habitat. The most likely cause of this population trend is the white-nose syndrome, climatic variability, and habitat loss (U.S. Fish and Wildlife Service, 2022f).

F.7.2.7.4 Predator and Prey Interactions

At present, there is no known predator for this bat species. However, the predators are likely the same as the for other bats in its range (State University of New York College of Environmental Science and Forestry, 2024). The northern long-eared bat has a diverse diet including moths, flies, leafhoppers, caddisflies, and beetles, and its diet differs geographically and seasonally. It forages using both hawking (catching prey in flight) and gleaning (picking motionless insects from vegetation and water surfaces) behaviors (U.S. Fish and Wildlife Service, 2016a, 2017a). Lepidopterans (moths) and coleopterans (beetles) are the most common insects found in northern long-eared bat diets, although arachnids are also a common prey item. Most foraging occurs above the understory, 1 to 3 m above the ground, but under the canopy on forested hillsides and ridges, rather than along riparian areas (U.S. Fish and Wildlife Service, 2016b).

F.7.2.7.5 Species-Specific Threats

The northern long-eared bat is one of the species of bats most impacted by white-nose syndrome (see 2018 Final EIS/OEIS Section 3.9.2.1.5.3, Disease and Parasites), which has caused declines of 90 to 100 percent where the disease has been found and is the primary factor supporting the endangered species status determination. Declines in the numbers of northern long-eared bats are expected to continue as white-nose syndrome extends across the species’ range (U.S. Fish and Wildlife Service, 2016a). In addition to this fungal infection, other sources of population impacts include wind turbines, climate change, and habitat loss and disturbance (U.S. Fish and Wildlife Service, 2024d).

F.7.2.8 Tricolored Bat (*Perimyotis subflavus*)

F.7.2.8.1 Status and Management

The tricolored bat was proposed as an endangered species under the ESA on September 13, 2022. The species occurs throughout portions of North and Central America, including a small part of southeastern Canada, 39 states, Mexico, Guatemala, Honduras, Belize, and Nicaragua (U.S. Fish and Wildlife Service, 2022b). The Fish and Wildlife Service has determined that the designation of critical habitat would not be beneficial to the species. White-nose syndrome is what puts the tricolored bat's population most at risk. Over 100 State and Federal agencies, Tribes, organizations, and institutions are working to fight against white-nose syndrome and conserve affected bats. Partners from all 39 states in the tricolored bat range, Canada, and Mexico are committed to manage disease surveillance, population monitoring, and management actions in response to white-nose syndrome (87 *Federal Register* 56381). USFWS created a white-nose syndrome zone to help identify where this disease is located, and which bats are affected. The boundary of this zone is updated monthly as new data are collected by USFWS's Midwest Region website (U.S. Fish and Wildlife Service, 2020a). It is estimated that over 90 percent of tricolored bat colonies have been affected by white-nose syndrome and the disease is currently present across 59 percent of the species' range (U.S. Fish and Wildlife Service, 2022c). In addition, other issues like loss of habitat, climate change, and anthropogenic disturbance, including vandalism and disturbance of winter habitat are also threats for this species (Center for Biological Diversity, 2024). One complication is that the data suggest the sex ratio is skewed with more male than female bats in a hibernacula, potentially due to higher annual survival rates by males but the data on this information are limited (for review see: McCoshum et al. (2023)). If the sex ratio data and higher survival rates of males are accurate, then whatever is causing these issues will compound the other threats impacting the population. The USFWS have not designated any specific habitat as critical habitat for the species. The USFWS determined that designating wintering habitat as critical habitat for the species would likely increase the threat of vandalism, disturbance, or the spread of white-nose syndrome. This species exhibits high site fidelity both for their overwintering habitat as well as for their summer habitat (U.S. Fish and Wildlife Service, 2024e). The USFWS have not designated any specific habitat as critical habitat for the species. The USFWS determined that designating wintering habitat as critical habitat for the species would likely increase the threat of vandalism, disturbance, or the spread of white-nose syndrome. This species exhibits high site fidelity both for their overwintering habitat as well as for their summer habitat (U.S. Fish and Wildlife Service, 2024a).

F.7.2.8.2 Habitat and Geographic Range

Tricolored bats are the first among North American bats to hibernate and the last to emerge in late spring, making their hibernation period extended (U.S. Fish and Wildlife Service, 2022e). A suitable habitat for hibernation in the winter includes caves and cave-like structures (e.g., abandoned or active mines). In the southern U.S., since caves are scarce, tricolored bats are often found in road-associated culverts, tree cavities and abandoned water wells. Suitable spring and summer habitat for the tricolored bat consists of a wide variety of forested and wooded habitats. They are found in trees, primarily in leaf clusters of live or recently dead deciduous hardwood trees. In the southern and northern portions of their range, tricolored bats will also roost in Spanish moss (*Tillandsia usneoides*) and lichen (*Usnea trichodea*) (U.S. Fish and Wildlife Service, 2023e).

For many years, this species has been considered a short-distance migratory or residential species, with limited movements from summering grounds to winter hibernation sites. Although, recent research has demonstrated that tricolored bats may be partially migratory, with some individuals in the central and northern portion of their habitat range. It is estimated that these bats migrate hundreds of kilometers north to south between summer breeding and foraging and winter hibernation (Smith et al., 2022a).

Within the Study Area, tricolored bats are most likely to occur off the coast of the eastern United States and Canada (U.S. Fish and Wildlife Service, 2023d).

F.7.2.8.3 Population Trends

The total adult population of the tricolored bat is unknown but exceeds 10,000 individuals. Summer and winter colony sizes are relatively small. Winter colonies average 61 individuals and can range up to 396 individuals in size (NatureServe, 2023). Current demographic conditions based on past species population reduction indicate the total number of tricolored bats known to still be thriving in winter colonies has declined by 29 percent. Declining trends in tricolored bat populations are also supported by summer data. The tricolored bats' range-wide occupancy declined by 28 percent in the period 2010-2019, and mobile acoustic detections decreased by 53 percent. The species also has a low probability of recovering from low population size given their low reproduction rates. Projected declines in tricolored bat populations show that under current conditions, by 2030, their range-wide abundance will decline by 89 percent, the number of known winter colonies will decline by 91 percent, and their spatial extent will decline by 65 percent (U.S. Department of the Interior & Fish and Wildlife Service, 2022).

F.7.2.8.4 Predator and Prey Interactions

Predators of tricolored bats include house cats, racoons, other bats, frogs, voles, and even a dark fishing spider (*Dolomedes tenebrosus*) (for review see: McCoshum et al. (2023)). Tricolored bats are opportunistic feeders and consume small insects including caddisflies, moths, beetles, wasps, flying ants and flies. This species of bat emerges early in the evening and forages near treetop level or above, but occasionally forages closer to ground later in the evening. The tricolored bat exhibits slow, erratic, fluttery flight, while foraging (U.S. Fish and Wildlife Service, 2023e). Small moths, beetles, and other insects among the trees along the waterways and forest edges are the most common prey for tricolored bats. They use echolocation to find prey and navigate through their habitat (U.S. Fish and Wildlife Service, 2022g).

F.7.2.8.5 Species-Specific Threats

The leading threat to the tricolored bat is white-nose syndrome. Within the last 20 years, white-nose syndrome emerged as a significant threat as it causes precipitous declines in populations of cave-hibernating bat species (see the 2018 Final EIS/OEIS Section 3.9.2.1.5.3, Disease and Parasites). The tricolored bat's prolonged hibernation period increases their exposure, and vulnerability, to white-nose syndrome, which is driving dramatic population declines across their range (U.S. Fish and Wildlife Service, 2022e). Other threats during their dormancy period include modifications to caves, mines, and surrounding areas that change airflow and alter the microclimate within their hibernation location. Human disturbance creates significant threats during hibernation by causing increased arousals. This leads to additional energy use in the winter when food and water resources are scarce. Disturbance is even more impactful to a bat in hibernation with white-nose syndrome because frequent arousals from torpor increase the probability of mortality in bats with little fat stores (U.S. Department of the Interior & Fish and Wildlife Service, 2022). Wind energy facilities are starting to become a consequential stressor at local and regional levels for tricolored bat mortality, especially in combination with impacts from white-nose syndrome. Most bat mortality located at these wind energy facilities is caused by direct collisions with the moving turbine blades. As of September 2022, wind energy development overlaps with 53 percent of the tricolored bat range in the U.S. and is still expanding (U.S. Fish and Wildlife Service, 2022g). Climate change variables are also starting to affect tricolored bat populations. Changes in temperature and precipitation create disturbance to bats in their roosting, foraging, commuting, and wintering habitats (U.S. Fish and Wildlife Service, 2022c).

F.7.3 SPECIES NOT LISTED UNDER THE ENDANGERED SPECIES ACT

F.7.3.1 Bird Species

There are 11 major taxonomic groups of birds represented in the Study Area. These birds may be found in the air, at the water surface, or in the water column. The vertical distribution provided for each taxonomic group are generalized and may not apply to all species within that group. The sections following describe each group of birds.

F.7.3.1.1 Geese, Swans, Dabbling and Diving Ducks (Order Anseriformes)

There are 50 species of swans, geese, dabbling, diving, and sea ducks in the family Anatidae in North America. No birds from this group are considered Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a). Birds from this group range from dabbling ducks found in coastal bays, estuaries, and lagoons to more open water ducks found in deeper water environments. Twenty-three of these species are diving ducks that inhabit nearshore or offshore waters of the Study Area (Sibley, 2014). Eiders, scoters, long-tailed duck (*Clangula hyemalis*), and harlequin duck (*Histrionicus histrionicus*) are sea ducks that winter in nearshore ocean waters. All these species can be found in deep water where they dive to forage (Sibley, 2014), some also forage on the bottom in shallow water. Most diving duck species dive to depths of up to 33 ft. (10 m) but long-tailed ducks have been reported to dive to depths 218 ft. (66 m) with a dive time of around 35 seconds (Sibley, 2014). Some inshore shark species, as well as alligators and crocodiles, prey on ducks on the surface of the water (Ehrlich et al., 1988).

Sea ducks and some diving ducks (e.g., scaups) breed inland but winter in large numbers in the Atlantic coastal waters of the Study Area and dive to the bottom, feeding primarily on benthic invertebrates. The harlequin duck is small and agile and prefers very turbulent water such as freshwater streams during the breeding season. Their winter habitat includes coastal intertidal areas, but they roost at night on open water farther offshore (greater than 0.6 mi. [1 km]) (Robertson & Goudie, 1999). The long-tailed duck winters in small groups in shallow ocean habitat.

Representative species that can be found in coastal bays, estuaries, and lagoons include geese (e.g., Canada goose [*Branta tellate*], brant [*Branta bernicla*]); swans (e.g., trumpeter swan [*Cygnus buccinators*], tundra swan [*Cygnus columbianus*]); dabbling ducks (e.g., mallard [*Anas platyrhynchos*], gadwall [*Anas strepera*], mottled duck [*Anas fulvigula*], American black duck [*Anas rubripes*], American wigeon [*Anas tellate*], northern shoveler [*Anas clypeata*], blue-winged teal [*Anas discors*], and green-winged teal [*Anas crecca*]); diving ducks (e.g., redhead [*Aythyatellate*], bufflehead [*Bucephala albeola*], common goldeneye [*Bucephala clangula*], and red-breasted merganser [*Mergus serrator*]); eiders (e.g., common eider [*Somateria mollissima*], king eider [*Somateria spectabilis*]); and scoters (e.g., surf scoter [*Melanitta perspicillata*], black scoter [*Melanitta tellate*]) (American Ornithologists' Union, 1998).

F.7.3.1.2 Loons (Order Gaviiformes)

There are five species of loons in the family Gaviidae in North America (American Ornithologists' Union, 1998), three of which occur in the Study Area. There are no birds in this group on the list of Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a). Loons are medium to large fish-eating birds that capture prey by diving underwater (Sibley, 2014). Loons can dive down to 250 ft. (76 m) with an average dive time of 40 seconds (Sibley, 2014). Loons move ashore only to breed, and all loon species nest on banks of inland ponds or lakes, requiring specific habitat features such as undeveloped shoreline and nest sites that have steep drop offs so they can approach their nest from underwater (Cornell Lab of Ornithology, 2009). For example, common loons spend their time in both freshwater and saltwater environments but prefer to nest on islands where the shoreline is not developed. Most loons need about 100 ft. (30.5 m) of room to take off, so size is another habitat feature that is important for nesting areas. During migration, loons fly high above land or water in loose groups or singly. They winter in coastal, nearshore, or open water marine habitats (Sibley, 2014). For example, the Pacific loon (*G. pacifica*) prefers deep water and is found on the open ocean and in bays. The red-throated loon, a representative species within the Study Area, has a circumpolar distribution, breeds in high latitudes on remote ponds, and winters along the Atlantic and Pacific coasts (American Ornithologists' Union, 1998).

F.7.3.1.3 Grebes (Order Podicipediformes)

There are seven species of grebes in the family Podicipedidae in North America (American Ornithologists' Union, 1998). There are no birds in this group on the list of Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a). Grebes can be found in a variety of aquatic habitats ranging from seasonally flooded scrubland and roadside ditches to deep lakes and coastal bays. Most grebe species winter in open waters while preferring marshy, vegetated habitats during the summer months (Sibley, 2014). Grebes forage by diving for small aquatic animals such as insects, fish, and crustaceans in the water column. For example, horned grebes can dive for up to 3 minutes and travel 500 ft. underwater, where they are sometimes preyed upon by sharks and orcas (Ehrlich et al., 1988). Grebes tend to escape predators by diving or sinking, leaving only the head exposed, rather than taking flight. All grebe species build floating nests in marshes and winter on the ocean and nearshore coastal areas (Sibley, 2014).

F.7.3.1.4 Albatrosses, Fulmars, Petrels, Shearwaters, and Storm-Petrels (Order Procellariiformes)

Procellariiformes is a large order of open ocean seabirds that are divided into four families: Diomedeidae (albatrosses), Procellariidae (petrels and shearwaters), Hydrobatidae (storm-petrels), and Pelecanoididae (diving petrels) (Enticott & Tipling, 1997; Onley & Scofield, 2007). This order includes species that are generally long-lived, breed once a year, and lay only one egg; thus, they have a low reproductive output. One of these species is listed as endangered under the ESA (Section F.7.2.1, Bermuda petrel [*Pterodroma cahow*]) (U.S. Fish and Wildlife Service, 2010b) and six are Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a).

Many seabirds spend most of their lives at sea and come to land only to breed, nest, and occasionally roost (Schreiber & Chovan, 1986). Colonial breeding is believed to have evolved in response to the limited availability of relatively predator-free nesting habitats and distance to foraging sites from breeding grounds (Siegel-Causey & Kharitonov, 1990). Benefits of colonial breeding include increased detection of predators and decreased chance of predation of young while parent birds are foraging away from the nest (Gill, 1995).

Seabirds can be found in high numbers resting on the water surface in flocks where prey is concentrated (Enticott & Tipling, 1997). Some species are found around fishing boats, where they often feed on bycatch and may become injured from longline gear (Enticott & Tipling, 1997; Onley & Scofield, 2007). Also, because of their pelagic nature, this group is preyed on by some pelagic shark species (Ehrlich et al., 1988). Oceanic fronts (gradients in current speed, temperature, salinity, density, and enhanced circulation) attract seabirds due to increased foraging opportunities. For example, the at-sea distribution of some seabirds is associated with oceanic fronts, which support increased numbers of prey and provide favorable foraging conditions (Bost et al., 2009).

There are 20 species of Procellariiformes in North America, with 13 species representing two families - the storm-petrels and petrels and shearwaters (American Ornithologists' Union, 1998) - occurring within the Study Area. Most of the petrel species in the Study Area are not considered part of the diving petrels and forage along the surface of the ocean. Petrels are colonial nesters and tend to nest on remote islands uninhabited by people.

Storm-petrels pick prey off the surface while foraging. Most breed in natural holes/cryptic burrows and visit their colonies only at night (Enticott & Tipling, 1997; Onley & Scofield, 2007). Fulmarine petrels, such as the northern fulmar (*Fulmarus glacialis*) and the black-capped petrel (*Pterodroma hasitata*), feed by landing on the sea and grabbing prey near the surface. Most fulmarine petrels nest in burrows or on cliff ledges and visit nests by day (Enticott & Tipling, 1997; Onley & Scofield, 2007). Gadfly petrels are generally species of the *Pterodroma* genus and are long-winged, fast flying, and highly pelagic. They feed on the wing and land on the sea (Onley & Scofield, 2007). Some gadfly petrels nest in burrows or crevices and visit colonies at night (Enticott & Tipling, 1997; Onley & Scofield, 2007).

Shearwaters are small- to medium-sized and dive to varying depths for prey (Onley & Scofield, 2007). For example, Cory's shearwater (*Calonectris diomedea*) rarely dives to 16 ft. (5 m) below the surface, while sooty (*Puffinus griseus*) and short-tailed shearwaters (*Puffinus tenuirostris*) can reach depths of 230 ft. (70 m), swimming underwater with half-open wings (Enticott & Tipling, 1997; Onley & Scofield, 2007). Greater shearwaters in the South Atlantic Ocean have been reported to dive down to 62 ft. (19 m) and as long as 40 s in a single dive. However, the majority of their dives were less than 6.6 ft. (2 m) (Ronconi et al., 2010).

F.7.3.1.5 Boobies, Gannets, Cormorants, and Frigatebirds (Order Suliformes)

The Suliformes order is a diverse group of large seabirds including anhingas, gannets, boobies, cormorants, and frigatebirds. This order is composed of 16 species in 4 families—12 species representing 2 families that occur within the Study Area. The magnificent frigatebird (*Fregata magnificens*) is considered a Bird of Conservation Concern (U.S. Fish and Wildlife Service, 2021a). Species of concern within the Study Area include the brown booby (*Sula dactylatra),* masked booby (*Sula dactylatra*), and magnificent frigatebird (American Ornithologists' Union, 1998).

Suliformes are less pelagic than the Procellariiformes, although some of these species such as frigatebirds are pelagic. Most species are colonial, feed on fish, and use a variety of breeding habitats including trees and bushes (but not burrows). Breeding strategies vary among species, with some being long-lived and having low breeding success, while others have higher annual breeding success, but higher annual adult death (Enticott & Tipling, 1997; Onley & Scofield, 2007).

Cormorants are voracious predators on inshore fishes and have been implicated as a major threat to the recovery efforts of Atlantic salmon in the Gulf of Maine where they feed on juvenile salmon (smolts) leaving the estuaries (Fay et al., 2006; National Marine Fisheries Service & U.S. Fish and Wildlife Service, 2005). Their offshore foraging range is limited by their need for undisturbed, dry nocturnal roosting sites (Shields et al., 2002).

Boobies and gannets are large seabirds that plunge from the air to capture their prey. Filling similar niches, boobies inhabit warmer areas and gannets colder regions. Boobies and gannets often nest on islands in colonies, with gannets nesting on cliffs (BirdLife International, 2012) and boobies generally on the ground if predators allow (Pratt et al., 1987). They forage offshore in large flocks at night, often feeding on squid.

Like tropicbirds and pelicans, members of this group all have webbed feet and eight toes, and all have a throat sac, called a gular sac (Brown & Harshman, 2008). This sac is highly developed and visible in pelicans and frigatebirds but is also readily apparent in boobies and cormorants. Pelicans use the sac to trap fish, frigatebirds use it as a mating display and to feed on fish, squid, and similar marine life (Dearborn et al., 2001), and cormorants and boobies utilize the sac for heat regulation. These birds nest in colonies, but individual birds are monogamous (Brown & Harshman, 2008).

F.7.3.1.6 Tropicbirds (Order Phaethontiformes)

Tropicbirds are medium-sized seabirds, predominately white with black patterning on the back, wings, and face. They have thick, pointed bills that are red or orange in color that are slightly decurved. Their most notable feature is the extremely long and narrow central tail feathers, which can be 11 to 22 in. long. Their wingspans average around 3 ft. Superficially, tropicbirds resemble terns. Tropicbirds are highly pelagic foragers in tropical and subtropical oceans, coming to land mainly to breed (Sibley, 2014). Tropicbirds are plunge-divers that feed on fish and could occur as rare visitors offshore in the Study Area in the Gulf of Mexico, Caribbean Sea, and southeast U.S. Continental Shelf Large Marine Ecosystems, and in the Gulf Stream and North Atlantic Gyre Open-Ocean Areas (Sibley, 2014). No birds from this group are considered Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a).

F.7.3.1.7 Pelicans, Herons, Egrets, Ibis, and Spoonbills (Order Pelecaniformes)

Pelecaniformes is a large group composed of long-legged, large billed species that includes pelicans, herons, egrets, ibis, and spoonbills. However, with the exception of two species of pelicans (described below), they are inhabitants marshes and are likely to occur only in nearshore areas of the Study Area. Herons, egrets, ibises, and spoonbills feed on variety of live aquatic prey hunted while wading in shallow nearshore or inshore waters. Two species, reddish egret (*Egretta rufescens*) and great blue heron (*Ardea herodias*), are Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a).

The brown pelican (*Pelecanus occidentalis*) primarily occurs in shallow (less than 150 ft. [46 m]) warm coastal marine and estuarine environments, as well as offshore where they forage primarily on fish by headfirst plunge-diving. Most plunge-diving is limited to 3.5 to 6.5 ft. (1 to 2 m) within the water column. Foraging occurs within 12 mi. (20 km) of nesting islands during the breeding season, and up to 47 mi. (75 km) offshore during the nonbreeding season (Shields et al., 2002). American white pelicans (*Pelecanus erythrorhynchos*) are found in shallow coastal bays, inlets, and estuaries that support forage fish (Knopf & Evans, 2004). Flocks forage cooperatively, swimming and encircling fish as a coordinated group or driving them into shallows, where they are caught with synchronized bill dipping (Enticott & Tipling, 1997; Onley & Scofield, 2007).

F.7.3.1.8 Flamingos (Order Phoenicopteriformes)

Flamingos are gregarious (social) wading birds in the genus *Phoenicopterus*, and the only genus in the family Phoenicopteridae. The American flamingo (*P. ruber)* species is found in the Study Area. The distribution range of the flamingo is extremely large and includes many Caribbean and South American countries. However, their occurrence in the United States is limited to the southern tip of Florida (Everglades National Park) (Sibley, 2014; Stevens & Pickett, 1994).

These wading birds forage in intertidal areas by rhythmically swinging their bills from side to side and filtering small organisms out of the mud (Sibley, 2014). Though most of their life cycle is spent along coastal areas, migration over offshore areas does occur (Elphick, 2007).

F.7.3.1.9 Osprey, Bald Eagles, Kites, and Falcons (Orders Accipitriformes and Falconiformes)

Accipitriformes is a large group consisting of 60 species in three families (American Ornithologists' Union, 1998). This order generally has broad wings well-suited for soaring. Falconiformes include nine North American species that, with the exception of the caracara (*Caracara cheriway*), are fast flying predators with pointed wings and a streamlined body shape (Sibley, 2014). Members of both orders hunt by day and feed on a variety of prey, including fish, small mammals, reptiles, and carrion. Species that are likely to occur within the Study Area include the osprey (Pandion haliaetus), bald eagle (*Haliaeetus leucocephalus*), peregrine falcon (*Falco peregrinus*), and swallow-tailed kite (*Elanoides forficatus*). The swallow-tailed kite is a Bird of Conservation Concern (U.S. Fish and Wildlife Service, 2021a).

Ospreys live near slow-moving waters of coastal, nearshore, and freshwater environments in many parts of the Study Area. They are plunge feeders but also have the ability to capture prey with their feet while keeping their head above water. Fish make up a large portion of their diet, and therefore, their vision is well adapted to detecting underwater objects from 33–131 ft. (10–40 m) above water (Poole et al., 2002). Ospreys migrate from northern latitudes to southern latitudes twice a year and cross bodies of open ocean to reach their destinations (Lott, 2006).

Bald eagles nest, forage, and winter along the Atlantic coast especially in the Chesapeake Bay region. Bald eagles also occur throughout Florida, although no bald eagle sightings have been recorded at Port Canaveral in 27 years (Federal Emergency Management Agency, 2012; Florida Fish and Wildlife Conservation Commission, 2017). Bald eagles have steadily increased since the ban on the pesticide DDT from 60 pairs in the 1970s to 646 in 2001. The Chesapeake Bay is very important to bald eagles because it is a convergence point for all three geographically distinct populations (northeast, southeast, and Chesapeake Bay) and has played an important part in their recovery (Watts et al., 2007). Bald eagles are opportunistic feeders that generally prefer fish over other food types (Buehler, 2000). Adults are known to scavenge prey items, pirate food from other species, and capture prey such as ducks from the water's surface.

Swallow-tailed kites breed in the southeastern United States but winter in South America, making long-distance migrations each year between wintering and breeding grounds. Studies in Florida show swallow-tailed kites feed on various animals in the following proportions: frogs (53 percent), birds (30 percent), and reptiles (11 percent) and the remaining prey were insects (Meyer et al., 2004).

Most peregrine falcons occur throughout the nearshore and coastal portions of the Study Area, particularly near barrier islands and mudflats during the winter months. Some peregrine falcons migrate along the coast, cross bodies of water such as the Gulf of Mexico and occur offshore of the Atlantic coast to reach their wintering/breeding territories on a yearly basis (Lott, 2006). They can reach altitudes up to 12,000 ft. (Cornell Lab of Ornithology, 2011). Peregrine falcons feed mostly on other birds, including shorebirds, ducks, grebes, gulls, and petrels. They occasionally feed on fish while in coastal habitats (Cornell Lab of Ornithology, 2011).

F.7.3.1.10 Coots, Cranes, Rails (Order Gruiformes)

The order Gruiformes consists of a wide variety of loosely related birds that are highly varied including the coots, cranes and rails, which could occur along the shoreline in the Study Area. Two species, king rail (*Rallus elegans*) and yellow rail (*Coturnicops noveboracensis*), are Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a).

The members of this order most commonly found in the Study Area are coots and rails. Coots are medium-sized water birds similar to ducks in body and habitat. Coots are omnivorous, primarily eating plant material but also insects, fish, eggs, snails, and tadpoles by dabbling at the water surface, diving or grazing on land. In nonbreeding season, they are commonly found in groups in shallow waters. Rails are a large diverse family of small- to medium-sized, ground-living birds. There are five species in North America, all of which may occur in the Study Area along the southeastern and eastern coastlines of the U.S. These are secretive species commonly found in marshes.

F.7.3.1.10.1 Order: Caprimulgiformes

The order Caprimulgiformes contains birds such as nightjars, frogmouths, and oilbirds that are distributed all over the world except Antarctica. Two species that migrate through the Study Area, Chuck-will's-widow (*Antrostomus carolinensis*) and Eastern whip-poor-will (*A. vociferus*), are Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a). Both species breed in forests in southeastern U.S. near the Study Area and winter in central and South America. Both feed on insects caught on the wing at night, primarily beetles, and both nest on the ground.

F.7.3.1.11 Shorebirds, Phalaropes, Gulls, Noddies, Terns, Skimmers, Skuas, Jaegers, and Alcids (Order Charadriiformes)

Shorebirds are small, generally long-legged coastal birds, many of which forage below the high tide in the surf zone by picking and probing for small aquatic prey (Sibley, 2014). Shorebirds undergo some of the longest distance migrations known for birds, for example, the red knot annually migrates more than 9,300 mi. (U.S. Fish and Wildlife Service, 2005). Though most of their life cycle is spent in coastal areas, shorebird migration over open ocean does occur (Elphick, 2007).

The Charadriiformes include shorebirds, phalaropes, gulls, noddies, terns, skimmers, skuas, jaegers, and alcids (Cornell Lab of Ornithology, 2009). There are 81 species from this diverse group that occur within the Study Area ranging from small shorebirds to large pelagic seabirds. Two endangered species under the ESA belong to this group, the roseate tern and piping plover (U.S. Fish and Wildlife Service, 2010a). Twenty-five species from this group are Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a). Some species in this order are highly pelagic (e.g., jaegers, skuas, alcids), whereas others are more coastal or nearshore species (e.g., shorebirds, gulls).

Although taxonomically grouped among some shorebirds, two species of phalaropes in the family Scolopacidae that occur within the Study Area are functionally seabirds, spending the nonbreeding months out on the open ocean. For example, the red-necked phalarope (*Phalaropus lobatus*) spends up to 9 months at sea, gathering in small flocks at upwellings and convergence zones, foraging on zooplankton and other small aquatic animals that rise to the surface (Rubega et al., 2000). The red phalarope ranges farthest from shore, spending 11 months at sea feeding on small invertebrates (Cornell Lab of Ornithology, 2002).

Skuas and jaegers are oceanic birds that come to land only to nest. On the nesting grounds they prey on lemmings, small birds, and other animals; in other seasons they pirate much of their food from other seabirds by chasing them and forcing them to relinquish captured prey (Sibley, 2014). Representative species from this group include: semipalmated plover (*Charadrius semipalmatus*), great skua (*Stercorarius skua*), long-tailed jaeger (*Stercorarius longicaudus*), sooty tern (*Onychoprion fuscatus*), brown noddy (*Anous stolidus*), dovekie (*Alle alle*), common murre (*Uria aalge*), razorbill (*Alca torda*), long-billed murrelet (*Brachyramphus perdix*), Atlantic puffin (*Fratercula arctica*), and red phalarope (*Phalaropus fulicarius*).

Noddies are tropical tern-like seabirds found foraging over warm, open ocean waters where they feed by swooping or dipping along the surface. Brown noddies breed in colonies on islands, islets, and rocky outcrops in warm seas. They only lay one egg a year and build their nests in trees, shrubs, cliffs, and manmade structures (Sibley, 2014).

Terns are generally more marine or pelagic than gulls, though some tern species do occur more commonly within coastal areas (e.g., least terns). Terns roost and nest in large groups on shorelines, and feed on small fish by plunge-diving headfirst from the air into the water, often beginning from a hovering position. They feed closer to shore when raising young during the nesting season, but venture farther offshore for longer periods after young have fledged (Sibley, 2014). In the North Atlantic, Gulf Stream eddies attract foraging seabirds such as the sooty tern and bridled tern (*Onychoprion anaethetus*) (Bost et al., 2009).

Alcids or auks (family *Alcidae*), are small oceanic species that inhabit cold Northern Hemisphere seas, rarely wandering south into the tropics (Pratt et al., 1987). They come to land only to breed (Enticott & Tipling, 1997) and nest colonially in crevices or burrows. Alcids do not undergo long-distance foraging trips but form feeding aggregations in areas where food is concentrated, though they do not form tight flocks (Enticott & Tipling, 1997). All alcids use their wings to dive underwater where they feed on fishes and invertebrates. Auks are pursuit divers and are entirely wing-propelled rather than foot-propelled, as are loons and grebes, for example. Atlantic puffins can dive between 135 to 224 ft. (41 to 68 m) for periods of up to 1 minute (Burger & Simpson, 1986).

The Charadriiformes influence the distribution and abundance of invertebrates, and indirectly algae, in rocky intertidal communities of New England (Ellis et al., 2007). Gulls are one particular group that can be found over land, along the coast, in nearshore, and offshore environments. The great black-backed gull (*Larus marinus*) and the herring gull (*Larus argentatus*) are dominant predators along the rocky shores throughout the North Atlantic. They forage while walking, swimming, or flying, sometimes dipping into the water and sometimes plunge-diving (National Audubon Society, 2015). They often feed on crabs, sea urchins, and mussels in the rocky intertidal habitat; once a prey item is caught, the gull will fly up and drop it on rocks below to break it open.

F.7.3.1.12 Neotropical Migrant Songbirds, Thrushes, Cuckoos, Swifts, Owls, and Allies (Orders Passeriformes, Cuculiformes, Apodiformes and Strigiformes)

There are 185 bird species in the orders Passeriformes, Cuculiformes, Apodiformes, and Strigiformes that are considered nocturnal migrants and neotropical migrants with a potential to occur in the Study Area. Eight of these species are Birds of Conservation Concern (U.S. Fish and Wildlife Service, 2021a). Most of these species are nocturnal migrants and take advantage of favorable weather conditions to migrate (Kerlinger, 2009). Oceans are typically an obstacle for this group of birds because most songbirds cannot swim, or even rest on the water's surface. Migrants tend to avoid large water crossings and follow land to the extent possible. Migration has a substantial risk to birds, ranging from mass mortality events due to inclement weather events (Newton, 2007) and other mortality events associated with lighting of vessels (Merkel & Johansen, 2011) and oil and gas platforms (Poot et al., 2008). In the Gulf of Mexico, long-distance migrants are commonly found stopping over and resting on oil and gas platforms as well as on small boats and vessels. Most neotropical migrants, especially warblers and thrushes from the family *Parulidae* and family *Turdidae*, cross water at some point twice a year to reach their wintering and breeding grounds. For example, the Bicknell's thrush *(Cartharus bicknelli*) breeds in mountainous forests of New England and migrates across open oceans in the fall to reach their wintering grounds in the Caribbean.

Aerial insect feeders such as swifts and predatory birds such as owls may feed opportunistically during migration across the ocean (Elphick, 2007), but the vast majority of bird species in this diverse group do not feed within the Study Area.

F.7.3.2 Bats

At least 24 species of bats are known or expected to occur in the Study Area (Table 3.9-3), either during migration or foraging. Additional bat species are known to occur in areas near, or adjacent to, the Study Area.

In temperate North America, most species that roost in trees, such as hoary bats, migrate south for winter when insects become scarce. In the fall, hundreds of hoary bats from across the United States gather along the coasts and in northern Mexico. Mexican free-tailed bats that roost in Carlsbad Caverns during the summer also migrate to Mexico over winter (National Park Service, 2017).

The Navy has performed bat surveys (both mist-netting and passive acoustic surveys) at several installations along the eastern coast of the United States. Results of these surveys are described below. Since echolocation calls for Eastern red bats and Seminole bats are indistinguishable from each other, survey results combine these two species. In addition, it typically is not possible to identify specific species from passive acoustic survey recordings of *Myotis* species, and occasionally it is not possible to make a determination more specific than "high-frequency call."

- Cutler, Maine (Tetra Tech Inc, 2014):
    - All seven bat species expected to occur in Maine that are not federally listed are known to occur at Naval Support Activity Cutler: little brown bat, Eastern small-footed bat, tricolored bat, silver-haired bat, big brown bat, Eastern red bat, and hoary bat.
    - Little brown bats were the most frequently detected species and occurred across the installation at all acoustic sites during the 2013 survey. Eastern red bat was the second most common species recorded at the Installation and occurred across all acoustic sites. Silver-haired bats and the Eastern red bat are known to be active from late April through mid-October, big brown bats from late March through early October, and hoary bats from early May through early October.
    - The installation provides the local bat community with habitat from the late spring to late fall. The data also suggest that bats are utilizing habitat and traveling closer to the coast within forested and edge habitats.
    - The occurrence of migratory bat species during the summer season indicates that long-distance migratory tree-roosting bats spent the summer residency period at the installation. Data also suggest that long-distance migrants move through the installation during the fall.
- Colts Neck, New Jersey (Tetra Tech Inc, 2016c):
    - Baseline bat survey at NWS Earle acoustically documented activity of eight different bat species, including big brown bat, Eastern red bat, hoary bat, silver-haired bat, little brown bat, Eastern small-footed bat, northern long-eared bat, and tricolored bat. Mist-net surveys further confirmed the presence of big brown bats, Eastern red bat, and northern long-eared bat.
- Norfolk and Portsmouth, Virginia (Tetra Tech Inc, 2017a):
    - At Naval Station Norfolk and Naval Supply Center Craney Island Fuel Terminal at Norfolk and Portsmouth, Virginia, mist-netting surveys captured Eastern red bats (*Lasiurus borealis*). Approximately 75 percent of acoustic detections were identified as Eastern red bats/Seminole bats; the remainder were mostly designated as "high-frequency" bats. Manual review of all tricolored bat passes were determined to not contain enough detail to accurately identify to species.
- Virginia Beach, Virginia (Tetra Tech Inc, 2016d):
    - Surveys at Naval Air Station Oceana in Virginia Beach, Virginia detected nine bat species: Rafinesque's big-eared bat, big brown bat, Eastern red/Seminole bat, hoary bat, silver-haired bat, southeastern bat, little brown bat, evening bat, and tricolored bat. Big brown bats were the most commonly recorded, accounting for 50 percent of the total calls, followed by silver-haired bats (24 percent), Eastern red bats/Seminole bats (11 percent), hoary bats (4 percent), and *Myotis* sp. bats (4 percent). Species with 2 percent or less of the total calls were little brown bats, southeastern bats, Rafinesque's big-eared bats, evening bats, tricolored bats, and high frequency bats.

Surveys at Joint Expeditionary Base Fort Story on the coast acoustically documented activity of at least ten different species of bats including Rafinesque's big-eared bat, big brown bat, Eastern red/Seminole bat, hoary bat, silver-haired bat, southeastern myotis, little brown bat, northern long-eared bat, evening bat, and tricolored bat. Eastern red bats, however, are very common and Seminole bats only occur occasionally in Virginia. The overall activity rate at Joint Expeditionary Base Fort Story was the highest detected at the four Navy bases surveyed in Virginia (Tetra Tech Inc, 2016a).

References

Abend, A. G. and T. D. Smith. (1999). *Review of Distribution of the Long-finned Pilot Whale (Globicephala melas) in the North Atlantic and Mediterranean* (NOAA Technical Memorandum National Marine Fisheries Service-NE-117). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Acevedo-Gutiérrez, A., D. A. Croll, and B. R. Tershy. (2002). High feeding costs limit dive time in the largest whales. *The Journal of Experimental Biology 205* 1747–1753.

Adzigbli, L. and D. Yuewen. (2018). Assessing the impact of oil spills on marine organisms. *Journal of Oceanography and Marine Research 6* (1).

Aguilar-Perera, A. (2006). Disappearance of a Nassau grouper spawning aggregation off the southern Mexican Caribbean coast. *Marine Ecology Progress Series 327* 289–296. DOI:10.3354/meps327289

Aguilar, A. (2009). Fin whale, *Balaenoptera physalus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 433–437). Cambridge, MA: Academic Press.

Aguilera, M., M. Medina-Suárez, J. Pinós, A. Liria-Loza, and L. Benejam. (2018). Marine debris as a barrier: Assessing the impacts on sea turtle hatchlings on their way to the ocean. *Marine Pollution Bulletin 137* 481–487. DOI:10.1016/j.marpolbul.2018.10.054

Ahlén, I., H. J. Baagøe, and L. Bach. (2009). Behavior of Scandinavian bats during migration and foraging at sea. *Journal of Mammology 90* (6): 1318–1323.

Ahrens, R. N. M. and W. E. Pine. (2014). Informing recovery goals based on historical population size and extant habitat: A case study of the Gulf sturgeon. *Marine and Coastal Fisheries 6* (1): 274–286. DOI:10.1080/19425120.2014.976679

Aissi, M., A. Celona, G. Comparetto, R. Mangano, M. Wurtz, and A. Moulins. (2008). Large-scale seasonal distribution of fin whales (*Balaenoptera physalus*) in the central Mediterranean Sea. *Journal of the Marine Biological Association of the United Kingdom 88* 1253–1261. DOI:doi:10.1017/S0025315408000891

Ajemian, M. J., J. J. Wetz, B. Shipley-Lozano, J. D. Shively, and G. W. Stunz. (2015). An Analysis of Artificial Reef Fish Community Structure along the Northwestern Gulf of Mexico Shelf: Potential Impacts of "Rigs-to-Reefs" Programs. *PLoS ONE 10* (5): e0126354. DOI:10.1371/journal.pone.0126354

Aleksa, K., C. Sasso, R. Nero, and D. Evans. (2018). Movements of leatherback turtles (*Dermochelys coriacea*) in the Gulf of Mexico. *Marine Biology 165* (158): 1–13. DOI:10.1007/s00227-018-3417-9

Allan, R. P., M. Barlow, M. P. Byrne, A. Cherchi, H. Douville, H. J. Fowler, T. Y. Gan, A. G. Pendergrass, D. Rosenfeld, and A. L. Swann. (2020). Advances in understanding large-scale responses of the water cycle to climate change. *Annals of the New York Academy of Sciences 1472* (1): 49-75.

Alter, S. E., M. P. Simmonds, and J. R. Brandon. (2010). Forecasting the consequences of climate-driven shifts in human behavior on cetaceans. *Marine Policy 34* (5): 943–954. DOI:doi:10.1016/j.marpol.2010.01.026

Alves, F., A. Dinis, I. Cascao, and L. Freitas. (2010). Bryde's whale (*Balaenoptera brydei*) stable associations and dive profiles: New insights from foraging behavior. *Marine Mammal Science 26* (1): 202–212. DOI:doi: 10.1111/j.1748-7692.2009.00333

American Bird Conservancy. (2024). *ABC's Bird Library: Black-capped Petrel*. Retrieved February 6, 2024, from https://abcbirds.org/bird/black-capped-petrel/#:~:text=The%20Black%2Dcapped%20Petrel%20is,from%20forest%20fires%20and%20encroachment.

American Ornithologists' Union. (1998). *The AOU Check-List of North American Birds* (7th ed.). Washington, DC: American Ornithologists' Union.

Andersen, J. M., Y. F. Wiersma, G. B. Stenson, M. O. Hammil, A. Rosing-Asvid, and M. Skern-Maurizen. (2013). Habitat selection by hooded seals (*Cystophora cristata*) in the Northwest Atlantic Ocean. *Journal of Marine Science 70* (1): 173–185. DOI:10.1093/icesjms/fss133

Anderson, D. M., E. Fensin, C. J. Gobler, A. E. Hoeglund, K. A. Hubbard, D. M. Kulis, J. H. Landsberg, K. A. Lefebvre, P. Provoost, M. L. Richlen, J. L. Smith, A. R. Solow, and V. L. Trainer. (2021). Marine harmful algal blooms (HABs) in the United States: History, current status and future trends. *Harmful Algae 102* 101975. DOI:https://doi.org/10.1016/j.hal.2021.101975

Anderson, D. W., C. J. Henny, C. Godinez-Reyes, F. Gress, E. L. Palacios, K. Santos del Prado, and J. Bredy. (2007). *Size of the California Brown Pelican Metapopulation during a non-El Niño year*. Reston, VA: U.S. Geological Survey.

Andrady, A. L. (2011). Microplastics in the marine environment. *Marine Pollution Bulletin 62* 1595–1605. DOI:10.1016/j.marpolbul.2011.05.030

Andrzejaczek, S., A. C. Gleiss, C. B. Pattiaratchi, and M. G. Meekan. (2019). Patterns and drivers of vertical movements of the large fishes of the epipelagic. *Reviews in Fish Biology and Fisheries 29* (2): 335-354. DOI:10.1007/s11160-019-09555-1

Andrzejaczek, S., R. J. Schallert, K. Forsberg, N. S. Arnoldi, M. Cabanillas-Torpoco, W. Purizaca, and B. A. Block. (2021). Reverse diel vertical movements of oceanic manta rays off the northern coast of Peru and implications for conservation. *Ecological Solutions and Evidence 2* (1): e12051. DOI:https://doi.org/10.1002/2688-8319.12051

Animal Diversity Web. (2024). *Myotis sodalis, Indiana bat*. Retrieved May 30, 2024, from https://animaldiversity.org/accounts/Myotis_sodalis/.

Aoki, D. M., J. R. Perrault, S. L. Hoffmann, J. R. Guertin, A. Page-Karjian, B. A. Stacy, and D. Lowry. (2023). Forensic determination of shark species as predators and scavengers of sea turtles in Florida and Alabama. DOI:10.3354/meps14214

Archer, F. I., II and W. F. Perrin. (1999). *Stenella coeruleoalba*. *American Society of Mammalogists 603* 1–9.

Archer, S. K., S. A. Heppell, B. X. Semmens, C. V. Pattengill-Semmens, P. G. Bush, C. M. McCoy, and B. C. Johnson. (2012). Patterns of color phase indicate spawn timing at a Nassau grouper, *Epinephelus striatus* spawning aggregation. *Current Zoology 58* 73–83.

Archibald, D. W. and M. C. James. (2018). Prevalence of visible injuries to leatherback sea turtles *Dermochelys coriacea* in the Northwest Atlantic. *Endangered Species Research 37* 149–163. DOI:10.3354/esr00920

Arendt, M. D., J. A. Schwenter, A. L. Segars, J. I. Byrd, P. P. Maier, J. D. Whitaker, D. W. Owens, G. Blanvillain, J. M. Quattro, and M. A. Roberts. (2012). Catch rates and demographics of loggerhead sea turtles (*Caretta caretta*) captured from the Charleston, South Carolina, shipping channel during the period of mandatory use of turtle excluder devices (TEDs). *Fishery Bulletin 110* (1): 98–109.

Armstrong, J. L. and J. E. Hightower. (2002). Potential for restoration of the Roanoke River population of Atlantic sturgeon. *Journal of Applied Ichthyology 18* (4-6): 475–480. DOI:10.1046/j.1439-0426.2002.00389

Arnold, T., C. Mealey, H. Leahey, A. W. Miller, J. M. Hall-Spencer, M. Milazzo, and K. Maers. (2012). Ocean acidification and the loss of phenolic substances in marine plants. *PLoS ONE 7* (4): e35107–e35107. DOI:10.1371/journal.pone.0035107

Aronson, R., A. Bruckner, J. Moore, B. Precht, and E. Weil. (2008a). *Acropora cervicornis*. *International Union for Conservation of Nature 2009. International Union for Conservation of Nature Red List of Threatened Species. Version 2009.2.* Retrieved from http://www.iucnredlist.org/apps/redlist/details/133381/0.

Aronson, R., A. Bruckner, J. Moore, B. Precht, and E. Weil. (2008b). *Acropora palmata*. *International Union for Conservation of Nature 2009. International Union for Conservation of Nature Red List of Threatened Species. Version 2009.2.* Retrieved from http://www.iucnredlist.org/apps/redlist/details/133006/0.

Aronson, R., A. Bruckner, J. Moore, B. Precht, and E. Weil. (2008c). *Montastraea annularis*. *International Union for Conservation of Nature 2010. International Union for Conservation of Nature Red List of Threatened Species. Version 2010.1.* Retrieved from http://www.iucnredlist.org/apps/redlist/details/133134/0.

Aronson, R., A. Bruckner, J. Moore, B. Precht, and E. Weil. (2008d). *Mycetophyllia ferox*. *International Union for Conservation of Nature 2010. International Union for Conservation of Nature Red List of Threatened Species. Version 2010.1.* Retrieved from http://www.iucnredlist.org/apps/redlist/details/133356/0.

Aronson, R. B. and W. F. Precht. (2001). White-band disease and the changing face of Caribbean coral reefs. In J. W. Porter (Ed.), *The Ecology and Etiology of Newly Emerging Marine Diseases* (pp. 25-38). Dordrecht: Springer Netherlands.

Arroyo-Arce, S., I. Thomson, E. Harrison, S. Wilmott, and G. Baker. (2017). First record of jaguar (*Panthera onca*) predation on a loggerhead sea turtle (*Caretta caretta*) in Tortuguero National Park, Costa Rica. *Herpetology Notes 10* 17–18.

Aschettino, J., D. Engelhaupt, and A. Engelhaupt. (2024). *Mid-Atlantic Baleen Whale Monitoring, Virginia Beach, Virginia: 2022/23 Annual Progress Report*. Norfolk, VA: Naval Facilities Engineering Systems Command Atlantic.

Aschettino, J. M., D. Engelhaupt, A. Engelhaupt, M. Richlen, and M. Cotter. (2022). *Mid-Atlantic Baleen Whale Monitoring, Virginia Beach, Virginia: 2020/21 Annual Progress Report*. Prepared for U.S. Fleet Forces Command Submitted to Naval Facilities Engineering Systems Command Atlantic, Norfolk, Virginia, under Contract N62470-20-0016, Task Order 21F4005, issued to HDR Inc., Virginia Beach, Virginia.

Aschettino, J. M., D. T. Engelhaupt, A. G. Engelhaupt, A. Dimatteo, T. Pusser, M. F. Richlen, and J. T. Bell. (2020). Satellite telemetry reveals spatial overlap between vessel high-traffic areas and humpback whales (*Megaptera novaeangliae*) near the mouth of the Chesapeake Bay. *Frontiers in Marine Science 7*.

Atlantic Sturgeon Status Review Team. (2007). *Status Review of Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus)*. Gloucester, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Auster, P. J. and R. W. Langton. (1999). The Effects of Fishing on Fish Habitat. In: *Fish Habitat: Essential Fish Habitat and Rehabilitation* (pp. 150–187). Symposium 22: American Fisheries Society.

Avens, L., M. D. Ramirez, L. R. Goshe, J. M. Clark, A. B. Meylan, W. Teas, D. J. Shaver, M. H. Godfrey, and L. Howell. (2021). Hawksbill sea turtle life-stage durations, somatic growth patterns, and age at maturation. *Endangered Species Research 45* 127–145. DOI:10.3354/esr01123

Ayres, K. L., R. K. Booth, J. A. Hempelmann, K. L. Koski, C. K. Emmons, R. W. Baird, K. Balcomb-Bartok, M. B. Hanson, M. J. Ford, and S. K. Wasser. (2012). Distinguishing the impacts of inadequate prey and vessel traffic on an endangered killer whale (*Orcinus orca*) population. *PLoS ONE 7* (6): e36842. DOI:10.1371/journal.pone.0036842

Azzellino, A., S. Gaspari, S. Airoldi, and B. Nani. (2008). Habitat use and preferences of cetaceans along the continental slope and the adjacent pelagic waters in the western Ligurian Sea. *Deep Sea Research Part I: Oceanographic Research Papers 55* (3): 296–323. DOI:10.1016/j.dsr.2007.11.006

Bain, M. B. (1997). Atlantic and shortnose sturgeons of the Hudson River: common and divergent life history attributes. *Environmental Biology of Fishes 48* 347–358.

Baird, R., D. McSweeney, C. Bane, J. Barlow, D. Salden, L. Antoine, R. LeDuc, and D. Webster. (2006). Killer Whales in Hawaiian Waters: Information on Population Identity and Feeding Habits. *Pacific Science 60* (4): 523–530.

Baird, R. W. (2001). Status of harbour seals*, Phoca vitulina*, in Canada. *The Canadian Field-Naturalist 115* (4): 663–675.

Baird, R. W. (2005). Sightings of dwarf (*Kogia sima*) and pygmy (*K. breviceps*) sperm whales from the main Hawaiian Islands. *Pacific Science 59* 461–466.

Baird, R. W. (2009a). False killer whale *Pseudorca crassidens*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 405–406). Cambridge, MA: Academic Press.

Baird, R. W. (2009b). *A Review of False Killer Whales in Hawaiian Waters: Biology, Status, and Risk Factors*. Olympia, WA: Marine Mammal Commission.

Baird, R. W., A. M. Gorgone, D. J. McSweeney, D. B. Webster, D. R. Salden, M. H. Deakos, A. D. Ligon, G. Schorr, J. Barlow, and S. D. Mahaffy. (2008). False killer whales (*Psuedorca crassidens*) around the main Hawaiian Islands: Long-term site fidelity, inter-island movements, and association patterns. *Marine Mammal Science 24* (3): 591–612. DOI:10.1111/j.1748.7692.2008.00200

Baird, R. W., A. D. Ligon, S. K. Hooker, and A. M. Gorgone. (2001). Subsurface and nighttime behaviour of pantropical spotted dolphins in Hawai'i. *Canadian Journal of Zoology 79* (6): 988–996. DOI:10.1139/cjz-79-6-988

Baird, R. W., G. S. Schorr, D. L. Webster, D. J. McSweeney, M. B. Hanson, and R. D. Andrews. (2010). Movements and habitat use of satellite-tagged false killer whales around the main Hawaiian Islands. *Endangered Species Research 10* 107–121. DOI:10.3354/esr00258

Baird, R. W. and P. J. Stacey. (1991). Status of Risso's dolphin, *Grampus griseus*, in Canada. *Canadian Field-Naturalist 105* 233–242.

Baird, R. W., D. L. Webster, Z. Swaim, H. J. Foley, D. B. Anderson, and A. J. Read. (2015). *Spatial Use by Cuvier's Beaked Whales, Short-finned Pilot Whales, Common Bottlenose Dolphins, and Short-beaked Common Dolphins Satellite Tagged off Cape Hatteras, North Carolina, in 2014. Final*. Norfolk, VA: Fleet Forces Command, Naval Facilities Engineering Command Atlantic.

Baird, R. W., D. L. Webster, Z. Swaim, H. J. Foley, D. B. Anderson, and A. J. Read. (2016). *Spatial Use by Odontocetes Satellite Tagged off Cape Hatteras, North Carolina in 2015. Final Report*. Virginia Beach, VA: U.S. Fleet Forces Command.

Bajzak, C. E., M. O. Hammill, G. B. Stenson, and S. Prinsenberg. (2011). Drifting away: implications of changes in ice condition for a pack-ice-breeding phocid, the harp seal (*Pagophilus groenlandicus*). *Canadian Journal of Zoology 89* 1050–1062. DOI:10.1139/Z11-081

Baker, A. N. and B. Madon. (2007). Bryde's whales (*Balaenoptera cf. brydei*) in the Hauraki Gulf and northeastern New Zealand waters. *Science for Conservation 272* 4–14.

Baker, J., M. Baumgartner, E. A. Becker, P. Boveng, D. Dick, J. Fiechter, J. Forcada, K. A. Forney, R. Griffis, J. Hare, A. Hobday, D. Howell, K. Laidre, N. Mantua, L. Quakenbush, J. Santora, P. Spencer, C. Stock, K. Stafford, W. Sydeman, K. Van Houtan, and R. Waples. (2016). *Report of a Workshop on Best Approaches and Needs for Projecting Marine Mammal Distributions in a Changing Climate*. Santa Cruz, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Balazik, M. T. (2012). *Life History Analysis of James River Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus) with Implications for Management and Recovery of the Species.* Virginia Commonwealth University, Richmond, VA.

Balazik, M. T., S. P. McIninch, G. C. Garman, and R. J. Latour. (2012a). Age and Growth of Atlantic Sturgeon in the James River, Virginia, 1997–2011. *Transactions of the American Fisheries Society 141* (4): 1074-1080. DOI:10.1080/00028487.2012.676590

Balazik, M. T. and J. A. Musick. (2015). Dual annual spawning races in Atlantic sturgeon. *PLoS ONE 10* (5): e0128234. DOI:10.1371/journal.pone.0128234

Balazik, M. T., K. J. Reine, A. J. Spells, C. A. Fredrickson, M. L. Fine, G. C. Garman, and S. P. McIninch. (2012b). The potential for vessel interactions with adult Atlantic sturgeon in the James River, Virginia. *North American Journal of Fisheries Management 32* (6): 1062–1069. DOI:10.1080/02755947.2012.716016

Balazs, G. H. (1986). Fibropapillomas in Hawaiian green turtles. *Marine Turtle Newsletter 39* 1–3.

Balazs, G. H., P. Craig, B. R. Winton, and R. K. Miya. (1994). *Satellite telemetry of green turtles nesting at French Frigate Shoals, Hawaii, and Rose Atoll, American Samoa*. Presented at the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. Hilton Head, SC.

Baldwin, R., M. Gallagher, and K. Van Waerebeek. (1999). A review of cetaceans from waters off the Arabian Peninsula. In M. Fisher, S. A. Ghazanfur, & J. A. Soalton (Eds.), *The Natural History of Oman: A Festschrift for Michael Gallagher* (pp. 161–189). SV Kerkwerve, The Netherlands: Backhuys Publishers.

Barchi, J. R., J. M. Knowles, and J. A. Simmons. (2013). Spatial memory and stereotype of flight paths by big brown bats in cluttered surroundings. *The Journal of Experimental Biology 216* (6): 1053–1063. DOI:10.1242/jeb.073197

Barco, S., M. Law, B. Drummond, H. Koopman, C. Trapani, S. Reinheimer, S. Rose, W. M. Swingle, and A. Williard. (2016). Loggerhead turtles killed by vessel and fishery interaction in Virginia, USA, are healthy prior to death. *Marine Ecology Progess Series 555* 221–234.

Barco, S. and G. G. Lockhart. (2015). *Turtle Tagging and Tracking in Chesapeake Bay and Coastal Waters of Virginia: 2014 Annual Progress Report. Draft Report* (Contract No. N62470-10-D-3011, Task Orders 41 and 50, issued to HDR Inc.). Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Barco, S., W. McLellan, J. Allen, R. Asmutis, R. Mallon-Day, E. Meagher, D. A. Pabst, J. Robbins, R. Seton, W. M. Swingle, M. Weinrich, and P. Clapham. (2002). Population identity of humpback whales (*Megaptera novaeangliae*) in the waters of the U.S. mid-Atlantic states. *Journal of Cetacean Research and Management 4* (2): 135–141.

Barco, S. G., M. L. Burt, R. A. DiGiovanni JR, W. M. Swingle, and A. S. Williard. (2018a). Loggerhead turtle *Caretta caretta* density and abundance in Chesapeake Bay and the temperate ocean waters of the southern portion of the Mid-Atlantic Bight. *Endangered Species Research 37* 269-287. DOI:10.3354/esr00917

Barco, S. G., S. A. Rose, and G. G. Lockhart. (2017). *Turtle Tagging and Tracking in Chesapeake Bay and Coastal Waters of Virginia: 2016 Annual Progress Report*. Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Barco, S. G., S. A. Rose, G. G. Lockhart, and A. DiMatteo. (2018b). *Sea Turtle Tagging and Tracking in Chesapeake Bay and Coastal Waters of Virginia: 2017 Annual Progress Report*. Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Barco, S. G., W. M. Swingle, W. A. Mlellan, R. N. Harris, and D. A. Pabst. (1999). Local Abundance and Distribution of Bottlenose Dolphins (Tursiops truncatus) in the Nearshore Waters of Virginia Beach, Virginia. *Marine Mammal Science 15* (2): 394-408. DOI:https://doi.org/10.1111/j.1748-7692.1999.tb00809.x

Barlas, M. E. (1999). *The distribution and abundance of harbor seals (Phoca vitulina concolor) and gray seals (Halichoerus grypus) in southern New England, Winter 1998–Summer 1999.* (Master's thesis). Boston University, Boston, MA.

Barlow, J. (1994). Abundance of large whales in California coastal waters: A comparison of ship surveys in 1979–1980 and in 1991. *Report of the International Whaling Commission 44* 399–406.

Barlow, J. and B. L. Taylor. (2001). *Estimates of Large Whale Abundance off California, Oregon, Washington, and Baja California Based on 1993 and 1996 Ship Surveys*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Barragan, A. R., J. A. Espín, and R. Barragan. (2022). *Northwest Atlantic Leatherback (Dermochelys coriacea) Regional Action Plan for the Wider Caribbean Region* (Widecast Technical Report). Godfrey, IL: Wider Caribbean Sea Turtle Conservation Network.

Barrett, L. T., S. E. Swearer, and T. Dempster. (2019). Impacts of marine and freshwater aquaculture on wildlife: A global meta-analysis. *Reviews in Aquaculture 11* (4): 1022-1044. DOI:https://doi.org/10.1111/raq.12277

Barros, N. B. and A. A. Myrberg, Jr. (1987). Prey detection by means of passive listening in bottlenose dolphins (*Tursiops truncatus*). *The Journal of the Acoustical Society of America 82* S65.

Barros, N. B. and R. S. Wells. (1998). Prey and feeding patterns of resident bottlenose dolphins (*Tursiops truncatus*) in Sarasota Bay, Florida. *Journal of Mammalogy 79* (3): 1045–1059.

Basile, E. R., H. W. Avery, W. F. Bien, and J. M. Keller. (2011). Diamondback terrapins as indicator species of persistent organic pollutants: Using Barnegat Bay, New Jersey as a case study. *Chemosphere 82* (1): 137–144.

Bassett, B. L., J. A. Hostetler, E. Leone, C. P. Shea, B. D. Barbeau, G. L. Lonati, A. L. Panike, A. Honaker, and L. I. Ward-Geiger. (2020). Quantifying sublethal Florida manatee watercraft interactions by examining scars on manatee carcasses. *Endangered Species Research 43* 395-408.

Bauer, L. J., M. S. Kendall, and C. F. Jeffrey. (2008). Incidence of marine debris and its relationships with benthic features in Gray's Reef National Marine Sanctuary, Southeast USA. *Marine Pollution Bulletin 56* (3): 402–413. DOI:10.1016/j.marpolbul.2007.11.001

Baum, E. (1997). *Maine Atlantic Salmon: A National Treasure*. Hermon, ME: Atlantic Salmon Unlimited.

Baum, J., E. Medina, J. A. Musick, and M. Smale. (2015). *Carcharhinus longimanus. The International Union for Conservation of Nature Red List of Threatened Species 2015: e.T39374A85699641*. Retrieved July 21, 2016, from http://www.iucnredlist.org/details/39374/0.

Baum, J. K., R. A. Myers, D. G. Kehler, B. Worm, S. J. Harley, and P. A. Doherty. (2003). Collapse and conservation of shark populations in the northwest Atlantic. *Science 299* (5605): 389–392.

Baumgartner, M. (2009). *Right Whale Diving and Foraging Behavior in the Southwestern Gulf of Maine* (Marine Mammals & Biological Oceanography Annual Reports: FY09). Woods Hole, MA: Woods Hole Oceanographic Institution.

Baumgartner, M., R. Ji, and C. Chen. (2009). *Physical and Biological Controls of Copepod Aggregation and Baleen Whale Distribution* (Marine Mammals & Biological Oceanography Annual Reports: FY09). Woods Hole, MA and New Bedford, MA: Office of Naval Research.

Baumgartner, M. F. (1997). The distribution of Risso’s dolphin (*Grampus griseus*) with respect to the physiography of the northern Gulf of Mexico. *Marine Mammal Science 13* (4): 614–638.

Baumgartner, M. F. and B. R. Mate. (2003). Summertime foraging ecology of North Atlantic right whales. *Marine Ecology Progress Series 264* 123–135.

Baumgartner, M. F. and B. R. Mate. (2005). Summer and fall habitat of North Atlantic right whales (*Eubalaena glacialis*) inferred from satellite telemetry. *Canadian Journal of Fisheries and Aquatic Sciences 62* 527–543.

Baumgartner, M. F., K. D. Mullin, L. N. May, and T. D. Leming. (2001). Cetacean habitats in the northern Gulf of Mexico. *Fishery Bulletin 99* 219–239.

Beason, R. (2004). *What Can Birds Hear?* Lincoln, NE: University of Nebraska.

Beatson, E. (2007). The diet of pygmy sperm whales, *Kogia breviceps*, stranded in New Zealand: Implications for conservation. *Reviews in Fish Biology and Fisheries 17* (2–3): 295–303. DOI:10.1007/s11160-007-9039-9

Becciu, P., M. H. Menz, A. Aurbach, S. A. Cabrera-Cruz, C. E. Wainwright, M. Scacco, M. Ciach, L. B. Pettersson, I. Maggini, and G. M. Arroyo. (2019). Environmental effects on flying migrants revealed by radar. *Ecography 42* (5): 942-955.

Becking, L. E., T. C. J. M. van Bussel, A. O. Debrot, and M. J. A. Christianen. (2014). First record of a Caribbean green turtle (*Chelonia mydas*) grazing on invasive seagrass (*Halophila stipulacea*). *Caribbean Journal of Science 48* (2–3): 162–163.

Bellido, J. J., J. J. Castillo, M. A. Farfan, J. L. Mons, and R. Real. (2007). First records of hooded seals (*Cystophora cristata*) in the Mediterranean Sea. *Journal of the Marine Biological Association of the U.K. - Biodiversity Records* 1–2.

Benkwitt, C. E., M. A. Albins, K. L. Buch, K. E. Ingeman, T. L. Kindinger, T. J. Pusack, C. D. Stallings, and M. A. Hixon. (2017). Is the lionfish invasion waning? Evidence from The Bahamas. *Coral Reefs 36* (4): 1255-1261. DOI:10.1007/s00338-017-1620-7

Benoit-Bird, K. J. (2004). Prey caloric value and predator energy needs: Foraging predictions for wild spinner dolphins. *Marine Biology 145* (3): 435–444. DOI:10.1007/s00227-004-1339-1

Benoit-Bird, K. J. and W. W. L. Au. (2003). Prey dynamics affect foraging by a pelagic predator (*Stenella longirostris*) over a range of spatial and temporal scales. *Behavioral Ecology and Sociobiology 53* 364–373. DOI:10.1007/s00265-003-0585-4

Benoit-Bird, K. J. and W. W. L. Au. (2004). Diel migration dynamics of an island-associated sound-scattering layer. *Deep-Sea Research I 51* 707–719.

Benoit-Bird, K. J., W. W. L. Au, R. E. Brainard, and M. O. Lammers. (2001). Diel horizontal migration of the Hawaiian mesopelagic boundary community observed acoustically. *Marine Ecology Progress Series 217* 1–14.

Bergström, L., L. Kautsky, T. Malm, R. Rosenberg, M. Wahlberg, N. Åstrand Capetillo, and D. Wilhelmsson. (2014). Effects of offshore wind farms on marine wildlife–A generalized impact assessment. *Environmental Research Letters 9* (3): 12. DOI:10.1088/1748-9326/9/3/034012

Bermuda Government Department of Environment and Natural Resources. (2021). *Cahow Recovery Program for Bermuda's Endangered National Bird 2020-2021 Breeding Season Report*. Hamilton, Bermuda: Bermuda Government Department of Environment and Natural Resources.

Bernard, H. J. and S. B. Reilly. (1999). Pilot whales, *Globicephala* Lesson, 1828. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 245–280). San Diego, CA: Academic Press.

Berrow, S. D. and E. Rogan. (1996). Stomach contents of harbor porpoises and dolphins in Irish waters. Lugano, Switzerland.

Berry, K. A., M. E. Peixoto, and S. S. Sadove. (2000). *Occurrence, Distribution and Abundance of Green Turtles, Chelonia mydas, in Long Island New York: 1986–1987* (Proceedings of the Eighteenth International Sea Turtle Symposium). Mazatlan, Sinaloa, Mexico: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Berta, A., J. L. Sumich, and K. M. Kovacs. (2006). *Marine Mammals: Evolutionary Biology* (2nd ed.). Burlington, MA: Elsevier.

Best, B. D., P. N. Halpin, A. J. Read, E. Fujioka, C. P. Good, E. A. LaBrecque, R. S. Schick, J. J. Roberts, L. J. Hazen, S. S. Qian, D. L. Palka, L. P. Garrison, and W. A. McLellan. (2012). Online cetacean habitat modeling system for the U.S. East Coast and Gulf of Mexico. *Endangered Species Research 18* 1–15. DOI:10.3354/ers00430

Best, P. B. (1996). Evidence of migration by Bryde's whales from the offshore population in the southeast Atlantic. *Reports of the International Whaling Commission 46* 315–322.

Bester, A., D. Priddel, and N. Klomp. (2011). Diet and foraging behaviour of the providence petrel *Pterodroma solandri*. *Marine Ornithology 39* 163–172.

Bester, C. (1999). *Biological profiles: Scalloped hammerhead*. Retrieved from https://www.floridamuseum.ufl.edu/fish/discover/species-profiles/sphyrna-lewini/.

Bester, C. (2012). *Biological profiles: Nassau grouper*. Retrieved from http://www.flmnh.ufl.edu/fish/Gallery/Descript/Nassaugrouper/Nassaugrouper.html.

Bettridge, S., C. S. Baker, J. Barlow, P. J. Clapham, M. Ford, D. Gouveia, D. K. Mattila, R. M. Pace, III, P. E. Rosel, G. K. Silber, and P. R. Wade. (2015). *Status Review of the Humpback Whale (Megaptera novaeangliae) under the Endangered Species Act* (NOAA Technical Memorandum NMFS-SWFSC-540). La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Beuter, K. J., R. Weiss, and B. Frankfurt. (1986). *Properties of the auditory system in birds and the effectiveness of acoustic scaring signals*. Presented at the Bird Strike Committee Europe, 18th Meeting Part I, 26–30 May 1986. Copenhagen, Denmark.

Bianchi, D., E. D. Galbraith, D. A. Carozza, K. Mislan, and C. A. Stock. (2013). Intensification of open-ocean oxygen depletion by vertically migrating animals. *Nature Geoscience 6* (7): 545-548.

Bianchi, D. and K. A. S. Mislan. (2016). Global patterns of diel vertical migration times and velocities from acoustic data. *Limnology and Oceanography 61* (1): 353-364.

Bilkovic, D. M., K. Havens, D. Stanhope, and K. Angstadt. (2014). Derelict fishing gear in Chesapeake Bay, Virginia: Spatial patterns and implications for marine fauna. *Marine Pollution Bulletin 80* (1-2): 114–123.

BirdLife International. (2008a). *Bermuda Petrel is being conserved through translocation and provision of artificial nest-sites*. Retrieved May 11, 2017, from http://datazone.birdlife.org/sowb/casestudy/bermuda-petrel-is-being-conserved-through-translocation-and-provision-of-artificial-nest-sites.

BirdLife International. (2008b, March 25). *Bermuda Petrel returns to Nonsuch Island after 400 years*. Retrieved May 11, 2017, from http://birdguides.com/webzine/article.asp?a=1294.

BirdLife International. (2012). *Spotlight on seabirds*. Retrieved May 11, 2017, from http://datazone.birdlife.org/sowb/spotseabirds.

BirdLife International. (2023a). *Species factsheet: Charadrius melodus*. Retrieved August 24, 2023, from http://datazone.birdlife.org/species/factsheeVpiping-plover-charadrius-melodus.

BirdLife International. (2023b). *Species factsheet: Stema dougallii*. Retrieved August 24, 2023, from http://datazone.birdlife.orglspecies/factsheeUroseate-tern-sterna-dougallii.

BirdLife International. (2024). *Black-capped Petrel Pterodroma hasitata: Species Factsheet*. Retrieved February 6, 2024, from http://datazone.birdlife.org/species/factsheet/black-capped-petrel-pterodroma-hasitata.

Biscoito, M., M. Segonzac, A. Almeida, D. Desbruyeres, P. Geistdoerfer, M. Turnipseed, and C. Van Dover. (2002). Fishes from the hydrothermal vents and cold seeps - An update. *CBM-Cahiers de Biologie Marine 43* (3-4): 359-362.

Bjorge, A. and K. A. Tolley. (2009). Harbor Porpoise, *Phocoena phocoena*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 530–532). Cambridge, MA: Academic Press.

Bjorndal, K. A. (1997). Foraging ecology and nutrition of sea turtles. In P. L. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles* (pp. 199–231). Boca Raton, FL: CRC Press.

Bjorndal, K. A. (2003). Roles of loggerhead sea turtles in marine ecosystems. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead Sea Turtles* (pp. 235–254). Washington, DC: Smithsonian Books.

Bjorndal, K. A. and A. B. Bolten. (1988). Growth rates of immature green turtles, *Chelonia mydas*, on feeding grounds in the southern Bahamas. *Copeia 1988* (3): 555–564.

Bjorndal, K. A., A. B. Bolten, B. Koike, B. A. Schroeder, D. J. Shaver, W. G. Teas, and W. N. Witzell. (2001). Somatic growth function for immature loggerhead sea turtles, *Caretta caretta*, in southeastern U.S. waters. *Fishery Bulletin 99* 240–246.

Bjorndal, K. A., A. B. Bolten, and H. R. Martins. (2000). Somatic growth model of juvenile loggerhead sea turtles *Caretta caretta*: Duration of pelagic stage. *Marine Ecology Progress Series 202* 265–272.

Bloch, D. and L. Lastein. (1993). Morphometric segregation of long-finned pilot whales in eastern and western North Atlantic. *Ophelia 38* 55–68.

Bloodworth, B. and D. K. Odell. (2008). Kogia breviceps. *American Society of Mammalogists 819* 1–12. DOI:DOI:10.1644/819.1

Boehlert, G. W. and A. B. Gill. (2010). Environmental and ecological effects of ocean renewable energy development: A current synthesis. *Oceanography 23* (2): 68–81.

Bogomolni, A. L., K. R. Pugliares, S. M. Sharp, K. Patchett, C. T. Harry, J. M. LaRocque, K. M. Touhey, and M. Moore. (2010). Mortality trends of stranded marine mammals on Cape Cod and southeastern Massachusetts, USA, 2000 to 2006. *Diseases Of Aquatic Organisms 88* 143–155. DOI:10.3354/dao02146

Bohn, K. M., C. F. Moss, and G. S. Wilkinson. (2006). Correlated evolution between hearing sensitivity and social calls in bats. *Biology Letters 2* (4): 561–564. DOI:10.1098/rsbl.2006.0501

Bolten, A. B. (2003). Variation in sea turtle life history patterns: Neritic vs. oceanic developmental stages. In P. L. Lutz, J. A. Musick, & J. Wyneken (Eds.), *The Biology of Sea Turtles* (Vol. II, pp. 243–258). Boca Raton, FL: CRC Press.

Boman, E. M., T. Bervoets, M. De Graaf, J. Dewenter, A. Maitz, M. P. M. Zu Schlochtern, J. Stapel, A. C. Smaal, and L. A. Nagelkerke. (2019). Diet and growth of juvenile queen *conch Lobatus gigas* (Gastropoda: Strombidae) in native, mixed and invasive seagrass habitats. *Marine Ecology Progress Series 621* 143-154.

Bonanno, G. and M. Orlando-Bonaca. (2020). Marine plastics: What risks and policies exist for seagrass ecosystems in the Plasticene? *Marine Pollution Bulletin 158* 111425. DOI:https://doi.org/10.1016/j.marpolbul.2020.111425

Bonde, R. K. and T. J. O'Shea. (1989). Sowerby's beaked whale (*Mesoplodon bidens*) in the Gulf of Mexico. *Journal of Mammalogy 70* 447–449.

Borchert, S. M., M. J. Osland, N. M. Enwright, and K. T. Griffith. (2018). Coastal wetland adaptation to sea level rise: Quantifying potential for landward migration and coastal squeeze. *Journal of Applied Ecology 55* (6): 2876–2887. DOI:https://doi.org/10.1111/1365-2664.13169

Bort, J., M. Richlen, T. McDonald, and J. Bell. (2023). Opportunistic Offshore Sighting of a Tricolored Bat (*Perimyotis subflavus*). *Southeastern Naturalist Notes 22* (1): 9-12.

Bost, C. A., C. Cotte, F. Bailleul, Y. Cherel, J. B. Charrassin, C. Guinet, D. G. Ainley, and H. Weimerskirch. (2009). The importance of oceanographic fronts to marine birds and mammals of the southern oceans. *Journal of Marine Systems 78* 363–376.

Botton, M. L., R. E. Loveland, and T. R. Jacobsen. (1994). Site selection by migratory shorebirds in Delaware Bay, and its relationship to beach characteristics and abundance of horseshoe-crab (*Limulus polyphemus*) eggs. *Auk 111* (3): 605–616.

Boulon, R., M. Chiappone, R. Halley, W. Jaap, B. Keller, B. Kruczynski, M. Miller, and C. Rogers. (2005). *Atlantic Acropora Status Review Document Report to National Marine Fisheries Service, Southeast Regional Office*.

Bovery, C. M. and J. Wyneken. (2015). Seasonal variation in sea turtle density and abundance in the southeast Florida current and surrounding waters. *PLoS ONE 10* (12): e0145980.

Bowen, B. W., A. L. Bass, S.-M. Chow, M. Bostrom, K. A. Bjorndal, A. B. Bolten, T. Okuyama, B. M. Bolker, S. Epperly, E. Lacasella, D. Shaver, M. Dodd, S. R. Hopkins-Murphy, J. A. Musick, M. Swingle, K. Rankin-Baransky, W. Teas, W. N. Witzell, and P. H. Dutton. (2004). Natal homing in juvenile loggerhead turtles *(Caretta caretta)*. *Molecular Ecology 13* 3797–3808. DOI:doi: 10.1111/j.1365-294X.2004.02356

Bowen, W. D., J. I. McMillan, and W. Blanchard. (2007). Reduced population growth of gray seals at Sable Island: Evidence from pup production and age of primiparity. *Marine Mammal Science 23* (1): 48–64. DOI:10.1111/j.1748-7692.2006.00085

Bowen, W. D. and D. B. Siniff. (1999). Distribution, population biology, and feeding ecology of marine mammals. In J. E. Reynolds, III & S. A. Rommel (Eds.), *Biology of Marine Mammals* (pp. 423–484). Washington, DC: Smithsonian Institution Press.

Boyer, J. N., J. W. Fourqurean, and R. D. Jones. (1999). Seasonal and long-term trends in the water quality of Florida Bay (1989–1997). *Estuaries 22* (2): 417–430.

Bradshaw, C. J. A., K. Evans, and M. A. Hindell. (2006). Mass cetacean strandings—A plea for empiricism. *Conservation Biology 20* (2): 584–586. DOI:10.1111/j.1523-1739.2006.00329.x

Brainard, R. E., C. Birkeland, C. M. Eakin, P. McElhany, M. W. Miller, M. Patterson, and G. A. Piniak. (2011). *Status Review Report of 82 Candidate Coral Species Petitioned Under the U.S. Endangered Species Act* (NOAA Technical Memorandum NMFS-PIFSC-27). Honolulu, HI: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Brame, A. B., T. R. Wiley, J. K. Carlson, S. V. Fordham, R. D. Grubbs, J. Osborne, R. M. Scharer, D. M. Bethea, and G. R. Poulakis. (2019). Biology, ecology, and status of the smalltooth sawfish *Pristis pectinata* in the USA. *Endangered Species Research 39* 9-23.

Branch, T. A. (2007). Abundance of Antarctic blue whales south of 60° S from three complete circumpolar sets of surveys. *Journal of Cetacean Research and Management 9* (3): 253–262.

Brandt, L. A., J. S. Beauchamp, B. M. Jeffery, M. S. Cherkiss, and F. J. Mazzotti. (2016). Fluctuating water depths affect American alligator (*Alligator mississippiensis*) body condition in the Everglades, Florida, USA. *Ecological Indicators 67* 441–450.

Brautigam, A. and K. L. Eckert. (2006). *Turning the Tide: Exploitation, Trade and Management of Marine Turtles in the Lesser Antilles, Central America, Columbia, and Venezuela*. Cambridge, United Kingdom: Traffic International.

Breece, M. W., D. A. Fox, K. J. Dunton, M. G. Frisk, A. Jordaan, M. J. Oliver, and C. Kurle. (2016). Dynamic seascapes predict the marine occurrence of an endangered species: Atlantic Sturgeon *Acipenser oxyrinchus oxyrinchus*. *Methods in Ecology and Evolution 7* (6): 725–733. DOI:10.1111/2041-210x.12532

Breece, M. W., M. J. Oliver, M. A. Cimino, and D. A. Fox. (2013). Shifting distributions of adult Atlantic sturgeon amidst post-industrialization and future impacts in the Delaware River: a maximum entropy approach. *PLoS ONE 8* (11): e81321. DOI:10.1371/journal.pone.0081321

Bresette, M., J. Gorham, and B. Peery. (1998). Site fidelity and size frequencies of juvenile green turtles (*Chelonia mydas*) utilizing near shore reefs in St. Lucie County, Florida. *Marine Turtle Newsletter 82* 5–7.

Bresette, M., D. Singewald, and E. De Maye. (2006). Recruitment of post-pelagic green turtles (*Chelonia mydas*) to nearshore reefs on Florida's east coast. In M. Frick, A. Panagopoulou, A. F. Rees, & K. Williams (Eds.), *Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation: Book of Abstracts* (pp. 288). Athens, Greece: National Marine Fisheries Service Southeast Fisheries Science Center, International Sea Turtle Society.

Brierley, A. S. (2014). Diel vertical migration. *Current Biology 24* (22): R1074-R1076.

Briggs-Gonzalez, V., C. Bonenfant, M. Basille, M. Cherkiss, J. Beauchamp, and F. Mazzotti. (2017). Life histories and conservation of long-lived reptiles, an illustration with the American Crocodile (*Crocodylus acutus*). *Journal of Animal Ecology 86* 1102–1113.

Britton, A. (2009). *Alligator mississippiensis (Daudin, 1801)*. Retrieved from http://crocodilian.com/cnhc/csp_amis.htm.

Brooke, M. (2004). Albatrosses and Petrels Across the World (pp. 520). New York, NY: Oxford University Press.

Broughton, K. (2012). *Office of National Marine Sanctuaries Science Review of Artificial Reefs*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service, National Marine Sanctuary Program.

Brower, L. (1996). Monarch butterfly orientation: Missing pieces of a magnificent puzzle. *The Journal of Experimental Biology 199* (Pt 1): 93-103.

Brown, D. M., J. Robbins, P. L. Sieswerda, C. Ackerman, J. M. Aschettino, S. Barco, T. Boye, R. A. DiGiovanni, K. Durham, A. Engelhaupt, A. Hill, L. Howes, K. F. Johnson, L. Jones, C. D. King, A. H. Kopelman, M. Laurino, S. Lonergan, S. D. Mallette, M. Pepe, C. Ramp, K. Rayfield, M. Rekdahl, H. C. Rosenbaum, R. Schoelkopf, D. Schulte, R. Sears, J. E. F. Stepanuk, J. E. Tackaberry, M. Weinrich, E. C. M. Parsons, and J. Wiedenmann. (2022). Site fidelity, population identity and demographic characteristics of humpback whales in the New York Bight apex. *Journal of the Marine Biological Association of the United Kingdom 102* (1-2): 157-165. DOI:10.1017/S0025315422000388

Brown, D. M., J. Robbins, P. L. Sieswerda, R. Schoelkopf, and E. C. M. Parsons. (2018). Humpback whale (*Megaptera novaeangliae*) sightings in the New York-New Jersey Harbor Estuary. *Marine Mammal Science 34* (1): 250-257. DOI:https://doi.org/10.1111/mms.12450

Brown, D. M., P. L. Sieswerda, and E. C. M. Parsons. (2019). Potential encounters between humpback whales (*Megaptera novaeangliae*) and vessels in the New York Bight apex, USA. *Marine Policy 106* 103527. DOI:https://doi.org/10.1016/j.marpol.2019.103527

Brown, J. J. and G. W. Murphy. (2010). Atlantic sturgeon vessel-strike mortalities in the Delaware estuary. *Fisheries 35* (2): 72–83.

Brown, J. W. and J. Harshman. (2008). *Pelecaniformes*. Retrieved from http://tolweb.org/Pelecaniformes/57152/2008.06.27.

Brown, M. W., D. Fenton, K. Smedbol, C. Merriman, K. Robichaud-Leblanc, and J. D. Conway. (2009). *Recovery Strategy for the North Atlantic Right Whale (Eubalaena glacialis) in Atlantic Canadian Waters* (Species at Risk Act Recovery Strategy Series). Ottawa, ON: Fisheries and Oceans Canada.

Brownell, W. and J. Stevely. (1981). The biology, fisheries, and management of the Queen Conch, *Strombus gigas*. *Marine Fisheries Review 43* (7): 1-12.

Bruckner, A. W. (2003). *Proceedings of the Caribbean Acropora Workshop: Potential Application of the U.S. Endangered Species Act as a Conservation Strategy*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

Brunell, A., V. Deem, B. Bankovich, F. Bled, and F. Mazzotti. (2023). Effects of translocation on American crocodile movements and habitat use in South Florida. *The Journal of Wildlife Management 87* (e22427): 1–20. DOI:10.1002/jwmg.22427

Brusca, R. C. and G. J. Brusca. (2003). Phylum Cnidaria. In R. C. Brusca & G. J. Brusca (Eds.), *Invertebrates* (pp. 219–283). Sunderland, MA: Sinauer Associates, Inc.

Bryant, D., L. Burke, J. McManus, and M. Spalding. (1998). *Reefs at Risk: A Map-Based Indicator of Threats to the World's Coral Reefs*. Washington, DC: World Resources Institute.

Buckland, S. T., D. Bloch, K. L. Cattanach, T. Gunnlaugsson, K. Hoydal, S. Lens, and J. Sigurjonsson. (1993). Distribution and abundance of long-finned pilot whales in the North Atlantic, estimated from NASS-87 and NASS-89 data. *Reports of the International Whaling Commission Special Issue 14* 33–49.

Buehler, D. A. (2000). *Bald Eagle (Haliaeetus leucocephalus)*. *The Birds of North America Online.* Retrieved 3, from http://bna.birds.cornell.edu/bna/species/506.

Bugoni, L., T. S. Neves, N. O. Leite, Jr., D. Carvalho, G. Sales, R. W. Furness, C. E. Stein, F. V. Peppes, B. B. Giffoni, and D. S. Monteiro. (2008). Potential bycatch of seabirds and turtles in hook-and-line fisheries of the Itaipava Fleet, Brazil. *Fisheries Research 90* 217–224.

Bunch, A. J., K. B. Carlson, F. J. Hoogakker, L. V. Plough, and H. K. Evans. (2021). Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus* Mitchill, 1815) early life stage consumption evidenced by high-throughput DNA sequencing. *Journal of Applied Ichthyology 37* (1): 12-19. DOI:https://doi.org/10.1111/jai.14153

Bunting, P., A. Rosenqvist, R. M. Lucas, L.-M. Rebelo, L. Hilarides, N. Thomas, A. Hardy, T. Itoh, M. Shimada, and C. M. Finlayson. (2018). The Global Mangrove Watch—A New 2010 Global Baseline of Mangrove Extent. *Remote Sensing 10* (10): 1669.

Bureau of Ocean Energy Management. (2013). *Revised Environmental Assessment for Commercial Wind Lease Issuance and Site Assessment Activities on the Atlantic Outer Continental Shelf Offshore Rhode Island and Massachusetts*. Washington, DC: U.S. Department of the Interior, Bureau of Ocean Energy Management.

Bureau of Ocean Energy Management. (2019). *Tracking Offshore Occurrence of Common Terns, Endangered Roseate Terns, and Threatened Piping Plovers with VHF Arrays*. Washington, DC: U.S. Department of the Interior, Bureau of Ocean Energy Management, Environmental Studies Program, and U.S. Department of the Interior, U.S. Fish and Wildlife Service, Division of Migratory Birds.

Bureau of Ocean Energy Management. (2021). *Atlantic Marine Assessment Program for Protected Species: FY15 – FY19*. Washington, DC: U.S. Department of the Interior, Bureau of Ocean Energy Management Headquarters.

Bureau of Ocean Energy Management Office of Renewable Energy Programs. (2022). *Coastal Virginia Offshore Wind Commercial Project Draft Environmental Impact Statement Volume 1*. Washington, DC: U.S. Department of the Interior, Bureau of Ocean Energy Management, Office of Renewable Energy Programs.

Burger, A. E., C. L. Hitchcock, and G. K. Davoren. (2004). Spatial aggregations of seabirds and their prey on the continental shelf off SW Vancouver Island. *Marine Ecology Progress Series 283* 279–292.

Burger, A. E. and M. Simpson. (1986). Diving depths of Atlantic puffins and common murres. *Auk 103* (4): 828–830.

Burger, J. and M. Gochfeld. (1988). Nest-site selection and temporal patterns in habitat use of roseate terns. *The Auk 105* (3): 433–438.

Burger, J., C. Gordon, J. Lawrence, J. Newman, G. Forcey, and L. Vlietstra. (2011). Risk evaluation for federally listed (roseate tern, piping plover) or candidate (red knot) bird species in offshore waters: A first step for managing the potential impacts of wind facility development on the Atlantic Outer Continental Shelf. *Renewable Energy 36* (1): 338–351. DOI:10.1016/j.renene.2010.06.048

Burkett, E. E., N. A. Rojek, A. E. Henry, M. J. Fluharty, L. Comrack, P. R. Kelly, A. C. Mahaney, and K. M. Fien. (2003). *Status Review of Xantus's Murrelet (Synthliboramphus) in California*. Sacramento, CA: California Department of Fish and Game, Habitat Conservation Planning Branch.

Burkholder, J., D. Eggleston, H. Glasgow, C. Brownie, R. Reed, G. Janowitz, M. Posey, G. Mella, C. Kinder, R. Corbett, D. Toms, T. Alphin, N. Deamer, and J. Springer. (2004). Comparative impacts of two major hurricane seasons on the Neuse River and western Pamlico Sound ecosystems. *Proceedings of the National Academy of Sciences 101* (25): 9291–9296. DOI:10.1073/pnas.0306842101

Burns, J. J. (2009). Harbor seal and spotted seal *Phoca vitulina* and *P. largha*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 533–542). Cambridge, MA: Academic Press.

Burns, T. J., H. Davidson, and M. W. Kennedy. (2016). Large-scale investment in the excavation and “camouflaging” phases by nesting Leatherback Turtles (*Dermochelys coriacea*). *Canadian Journal of Zoology 94* (6): 443–448.

Burt, M. L., L. A. S. Scott-Hayward, and D. L. Borchers. (2014). *Analysis of Aerial Surveys Conducted in Coastal Waters of Maryland and Virginia, Including Chesapeake Bay, 2011–2013: Loggerhead Turtles. Draft Report.* Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Buskey, E. J., H. K. White, and A. J. Esbaugh. (2016). Impact of oil spills on marine life in the Gulf of Mexico: Effects on plankton, nekton, and deep-sea benthos. *Oceanography 29* (3): 174-181.

Byers, J. E. and J. H. Grabowski. (2013). Soft-sediment communities. In M. D. Bertness, J. F. Bruno, B. R. Silliman, & J. J. Stachowicz (Eds.), *Marine Community Ecology and Conservation*. Sunderland, MA: Sinauer Associates.

Caillouet, C. W., B. J. Gallaway, and N. F. Putman. (2016). Kemp's Ridley Sea Turtle saga and setback: Novel analyses of cumulative hatchlings released and time-lagged annual nests in Tamaulipas, Mexico. *Chelonian Conservation and Biology 15* (1): 115–131.

Caillouet, C. W., Jr., R. A. Hart, and J. M. Nance. (2008). Growth overfishing in the brown shrimp fishery of Texas, Louisiana, and adjoining Gulf of Mexico EEZ. *Fisheries Research 92* 289–302.

Calambokidis, J., E. Falcone, A. Douglas, L. Schlender, and J. Huggins. (2009). *Photographic Identification of Humpback and Blue Whales off the U.S. West Coast: Results and Updated Abundance Estimates from 2008 Field Season*. La Jolla, CA: Southwest Fisheries Science Center, and Olympia, WA: Cascadia Research Collective.

Calambokidis, J., G. H. Steiger, J. M. Straley, L. M. Herman, S. Cerchio, D. R. Salden, J. Urban R., J. K. Jacobsen, O. von Ziegesar, K. C. Balcomb, C. M. Gabriele, M. E. Dahlheim, S. Uchida, G. Ellis, Y. Miyamura, P. Ladron De Guevara, M. Yamaguchi, F. Sato, S. A. Mizroch, L. Schlender, K. Rasmussen, J. Barlow, and T. J. Quinn, II. (2001). Movements and population structure of humpback whales in the North Pacific. *Marine Mammal Science 17* (4): 769–794.

Caldwell, D. K. and M. C. Caldwell. (1989). Pygmy sperm whale, *Kogia breviceps* (de Blainville, 1838): Dwarf sperm whale *Kogia simus* Owen, 1866. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 4, pp. 234–260). San Diego, CA: Academic Press.

Caldwell, M. (2001). *Social and genetic structure of bottlenose dolphin (Tursiops truncatus) in Jacksonville, Florida.* (Dissertation). University of Miami, Miami, FL.

California Department of Fish and Game. (2010). *State and Federally Listed Endangered and Threatened Animals of California*. Sacramento, CA: California Natural Resources Agency, Department of Fish and Game, Biogeographic Data Branch.

Calleson, C. S. and R. K. Frohlich. (2007). Slower boat speeds reduce risks to manatees. *Endangered Species Research 3* 295–304. DOI:10.3354/esr00056

Calvert, A. M., D. L. Amirault, F. Shaffer, R. Elliot, A. Hanson, J. McKnight, and P. D. Taylor. (2006). Population assessment of an endangered shorebird: The piping plover (*Charadrius melodus*) in eastern Canada. *Avian Conservation and Ecology 1* (4): n.p.

Calvert, J. and C. McGonigle. (2020). Chapter 26 - Benthic community structure at a remote temperate rocky reef in the Gulf of Maine, Cashes Ledge. In P. T. Harris & E. Baker (Eds.), *Seafloor Geomorphology as Benthic Habitat (Second Edition)* (pp. 461–471). Amsterdam, Netherlands: Elsevier.

Campbell-Malone, R., S. G. Barco, P.-Y. Daoust, A. R. Knowlton, W. A. McLellan, D. S. Rotstein, and M. J. Moore. (2008). Gross and histologic evidence of sharp and blunt trauma in North Atlantic right whales (Eubalaena glacialis) killed by vessels. *Journal of Zoo and Wildlife Medicine 39* (1): 37–55, 19.

Campbell, C. (2016, July 15, 2016). Port of Baltimore's first container ship through expanded Panama Canal carries hope for future. *The Baltimore Sun*. Retrieved February 28, 2017, from http://www.baltimoresun.com/business/bs-bz-panama-canal-expansion-20160715-story.html.

Campbell, M. D., A. G. Pollack, K. Thompson, T. Switzer, W. B. Driggers, E. R. Hoffmayer, S. Keenan, C. Gardner, D. Hanisko, K. R. Rademacher, and K. Overly. (2021). Rapid spatial expansion and population increase of invasive lionfish (*Pterois* spp.) observed on natural habitats in the northern Gulf of Mexico. *Biological Invasions 24* (1): 93–105. DOI:10.1007/s10530-021-02625-1

Cañadas, A., R. Sagarminaga, and S. García-Tiscar. (2002). Cetacean distribution related with depth and slope in the Mediterranean waters off southern Spain. *Deep-Sea Research I 49* 2053–2073.

Cardona-Maldonado, M. A. and A. A. Mignucci-Giannoni. (1999). Pygmy and dwarf sperm whales in Puerto Rico and the Virgin Islands, with a review of *Kogia* in the Caribbean. *Caribbean Journal of Science 35* (1–2): 29–37.

Carlson, J. K., S. J. B. Gulak, C. A. Simpfendorfer, R. D. Grubbs, J. G. Romine, and G. H. Burgess. (2014). Movement patterns and habitat use of smalltooth sawfish, *Pristis pectinata*, determined using pop-up satellite archival tags. *Aquatic Conservation: Marine and Freshwater Ecosystems 24* (1): 104-117. DOI:https://doi.org/10.1002/aqc.2382

Carlson, J. K. and J. Osborne. (2012). *Relative abundance of smalltooth sawfish (Pristis pectinata) based on the Everglades National Park Creel Survey*. National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center, Panama City Laboratory. NOAA Technical Memorandum NMFS-SEFSC-626.

Carlson, J. K. and C. A. Simpfendorfer. (2015). Recovery potential of smalltooth sawfish, Pristis pectinata, in the United States determined using population viability models. *Aquatic Conservation: Marine and Freshwater Ecosystems 25* (2): 187-200. DOI:https://doi.org/10.1002/aqc.2434

Carr, A. (1986). Rips, FADS, and little loggerheads: Years of research have told us much about the behavioral ecology of sea turtles, but mysteries remain. *BioScience 36* (2): 92–100.

Carr, A. (1987). New perspectives on the pelagic stage of sea turtle development. *Conservation Biology 1* (2): 103–121.

Carr, A. and A. B. Meylan. (1980). Evidence of passive migration of green turtle hatchlings in *Sargassum*. *Copeia 1980* (2): 366–368.

Carr, S. H., T. Carr, and F. A. Chapman. (1996). First observations of young-of-year Gulf of Mexico sturgeon (*Acipenser oxyrinchus de sotoi*) in the Suwannee River, Florida. *Gulf of Mexico Science 141* 44–46.

Carretta, J. V., K. A. Forney, M. S. Lowry, J. Barlow, J. Baker, D. Johnston, B. Hanson, R. L. Brownell, Jr., J. Robbins, D. Mattila, K. Ralls, M. M. Muto, D. Lynch, and L. Carswell. (2010). *U.S. Pacific Marine Mammal Stock Assessments: 2009*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Carter, H. R. and K. J. Kuletz. (1995). Mortality of Marbled Murrelets Due to Oil Pollution in North America. In C. J. Ralph, G. L. Hunt, Jr., M. G. Raphael, & J. F. Piatt (Eds.), *Ecology and Conservation of the Marbled Murrelet* (Vol. General Technical Report PSW-152, pp. 261–269). Washington, DC: U.S. Department of Agriculture Forest Service.

Carter, H. R., S. G. Sealy, E. E. Burkett, and J. F. Piatt. (2005). Biology and conservation of Xantus's Murrelet: Discovery, taxonomy, and distribution. *Marine Ornithology 33* 81–87.

Casale, P. and S. Ceriani. (2020). Sea turtle populations are overestimated worldwide from remigration intervals: correction for bias. *Endangered Species Research 41* 141–150. DOI:10.3354/esr01019

Castelblanco-Martinez, D. N., C. Nourisson, E. Quintana-Rizzo, J. Padilla-Saldivar, and J. J. Schmitter-Soto. (2012). Potential effects of human pressure and habitat fragmentation on population viability of the Antillean manatee *Trichechus manatus manatus*: A predictive model. *Endangered Species Research 18* 129–287. DOI:10.3354/esr00439

Castro, J. I. (1983). *The Sharks of North American Waters*. College Station, TX: Texas A&M University Press.

Castro, P. and M. E. Huber. (2000). Marine prokaryotes, protists, fungi, and plants. In *Marine Biology* (3rd ed., pp. 83–103). New York, NY: McGraw-Hill.

Cates, K., D. P. DeMaster, R. L. Brownell, Jr., and G. Silber. (2017). *Strategic Plan to Mitigate the Impacts of Ship Strikes on Cetacean Populations: 2017–2020*. Cambridge, United Kingdom: International Whaling Commission.

Causey, B., J. Delaney, E. Diaz, D. Dodge, J. Garcia, J. Higgins, B. Keller, R. Kelty, W. Jaap, C. Matos, G. Schmahl, C. Rogers, M. Miller, and D. Turgeon. (2002). Status of coral reefs in the U.S. Caribbean and Gulf of Mexico: Florida,Texas, Puerto Rico, U.S. Virgin Islands, and Navassa. In C. Wilkinson (Ed.), *Status of Coral Reefs of the World: 2002* (pp. 251–276). Cape Ferguson, Australia: Australian Institue for Marine Science.

Ceccarelli, D. M. (2009). *Impacts of Plastic Debris on Australian Marine Wildlife*. Canberra, Australia: Department of the Environment, Water, Heritage and the Arts.

Center for Biological Diversity. (2024). *Saving the Tricolored Bat*. Retrieved May 30, 2024, from https://www.biologicaldiversity.org/species/mammals/tricolored_bat/index.html.

Ceriani, S., P. Casale, M. Brost, E. Leone, and B. Witherington. (2019). Conservation implications of sea turtle nesting trends: elusive recovery of a globally important loggerhead population. *Ecosphere 10* (11): 1–19. DOI:10.1002/ecs2.2936

Ceriani, S. A., S. Murasko, D. S. Addison, D. Anderson, G. Curry, N. A. Desjardin, S. F. Eastman, D. R. Evans, N. Evou, M. M. P. B. Fuentes, M. H. Godfrey, K. M. Hart, P. Hillbrand, S. E. Hirsch, C. R. Mott, K. L. Mansfield, K. T. Mazzarella, S. V. Norris, S. M. Pate, K. F. Phillips, K. W. Rusenko, B. M. Shamblin, A. Stevenson, K. A. Sloan, A. D. Tucker, R. C. Welsh, and P. Casale. (2023). Monitoring population-level foraging distribution of a marine migratory species from land: strengths and weaknesses of the isotopic approach on the Northwest Atlantic loggerhead turtle aggregation. *Frontiers in Marine Science 10* 1-22. DOI:10.3389/fmars.2023.1189661

Ceriani, S. A., J. F. Weishampel, L. M. Ehrhart, K. L. Mansfield, and M. B. Wunder. (2017). Foraging and recruitment hotspot dynamics for the largest Atlantic loggerhead turtle rookery. *Scientific Reports 7* (16894): 1–13. DOI:10.1038/s41598-017-17206-3

Cetacean and Turtle Assessment Program. (1982). *A Characterization of Marine Mammals and Turtles in the Mid- and North Atlantic Areas of the U.S. Outer Continental Shelf*. Kingston, RI: University of Rhode Island, Graduate School of Oceanography.

Chaloupka, M., G. H. Balazs, and T. M. Work. (2009). Rise and fall over 26 years of a marine epizootic in Hawaiian green sea turtles. *Journal of Wildlife Diseases 45* (4): 1138–1142.

Chaloupka, M., T. M. Work, G. H. Balazs, S. K. K. Murakawa, and R. Morris. (2008). Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982–2003). *Marine Biology 154* 887–898. DOI:10.1007/s00227-008-0981-4.

Chambault, P., F. Roquet, S. Benhamou, A. Baudena, E. Pauthenet, B. de Thoisy, M. Bonola, V. Dos Reis, R. Crasson, M. Brucker, Y. Le Maho, and D. Chevallier. (2017). The Gulf Stream frontal system: A key oceanographic feature in the habitat selection of the leatherback turtle? *Deep Sea Research Part I: Oceanographic Research Papers 123* 35-47. DOI:https://doi.org/10.1016/j.dsr.2017.03.003

Chapman, F. A. and S. H. Carr. (1995). Implications of early life stages in the natural history of the Gulf of Mexico sturgeon, *Acipenser oxyrinchus de sotoi*. *Environmental Biology of Fishes 43* 407–413.

Chapman, J. W., R. L. Nesbit, L. E. Burgin, D. R. Reynolds, A. D. Smith, D. R. Middleton, and J. K. Hill. (2010). Flight orientation behaviors promote optimal migration trajectories in high-flying insects. *Science 327* (5966): 682-685.

Chapman, R. and J. A. Seminoff. (2016). *Status of Loggerhead Turtles (Caretta caretta) within Nations of the Inter-American Convention for the Protection and Conservation of Sea Turtles*. Falls Church, VA: Inter-American Convention for the Protection and Conservation of Sea Turtles.

Chaput, G. (2012). Overview of the status of Atlantic salmon (*Salmo salar*) in the North Atlantic and trends in marine mortality. *ICES Journal of Marine Science: Journal du Conseil 69* (9): 1538–1548. DOI:10.1093/icesjms/fss013

Chavez-Rosales, S., D. L. Palka, L. P. Garrison, and E. A. Josephson. (2019). Environmental predictors of habitat suitability and occurrence of cetaceans in the western North Atlantic Ocean. *Scientific Reports 9* (1): 5833. DOI:10.1038/s41598-019-42288-6

Chen, C., G. Wang, S.-P. Xie, and W. Liu. (2019). Why does global warming weaken the Gulf Stream but intensify the Kuroshio? *Journal of Climate 32* (21): 7437-7451. DOI:https://doi.org/10.1175/JCLI-D-18-0895.1

Cherkiss, M. S., F. J. Mazzotti, L. Hord, and M. Aldecoa. (2014). Remarkable movements of an American crocodile (*Crocodylus acutus*) in Florida. *Southeastern Naturalist 13* (4): N52–N56.

Christiansen, F., N. F. Putman, R. Farman, D. M. Parker, M. R. Rice, J. J. Polovina, G. H. Balazs, and G. C. Hays. (2016). Spatial variation in directional swimming enables juvenile sea turtles to reach and remain in productive waters. *Marine Ecology Progress Series 557* 247–259.

Cipriano, F. (2009). Atlantic white-sided dolphin, *Lagenorhynchus acutus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 56–58). Cambridge, MA: Academic Press.

Clabough, E. B. D., E. Kaplan, D. Hermeyer, T. Zimmerman, J. Chamberlin, and S. Wantman. (2022). The secret life of baby turtles: A novel system to predict hatchling emergence, detect infertile nests, and remotely monitor sea turtle nest events. *PLoS ONE 17* (10): e0275088. DOI:10.1371/journal.pone.0275088

Clapham, P. J. and D. K. Mattila. (1990). Humpback whale songs as indicators of migration routes. *Marine Mammal Science 6* (2): 155–160.

Clapham, P. J. and J. G. Mead. (1999). *Megaptera novaeangliae*. *American Society of Mammalogists 604* 1–9.

Clapham, P. J., S. B. Young, and R. L. Brownell, Jr. (1999). Baleen whales: Conservation issues and the status of the most endangered populations. *Mammal Review 29* 35–60.

Clark, C. W. (1995). Application of U.S. Navy Underwater Hydrophone Arrays for Scientific Research on Whales. *Report of the International Whaling Commission 45* 210–212.

Clark, K. E., L. J. Niles, and J. Burger. (1993). Abundance and distribution of migrant shorebirds in Delaware Bay. *Condor 95* (3): 694–705. DOI:10.2307/1369612

Clark, R., J. Taylor, C. Buckel, and L. Kracker. (2014). *Fish and Benthic Communities of the Flower Garden Banks National Marine Sanctuary: Science to Support Sanctuary Management* (NOAA Technical Memorandum NOS NCCOS). Silver Spring, MD: NOAA National Centers for Coastal Ocean Science.

Clarke, M. C., M. E. Mach, and R. G. Martone. (2014). *Cumulative effects in marine ecosystems: Scientific perspectives on its challenges and solutions* (WWF-Canada and Center For Ocean Solutions). Vancouver, Canada: World Wide Fund for Nature and the Center for Ocean Solutions.

Clarke, M. R. (1986). Cephalopods in the diet of odonotocetes. In M. M. Bryden & R. J. Harrison (Eds.), *Research on Dolphins* (pp. 281–321). Oxford, United Kingdom: Clarendon Press.

Clarke, M. R. (1996). Cephalopods as prey III. Cetaceans. *Philosophical Transactions of the Royal Society of London 351* 1053–1065.

Clarke, M. R. and T. K. Kristensen. (1980). Cephalopod beaks from the stomachs of two northern bottlenosed whales (*Hyperoodon ampullatus*). *Journal of the Marine Biological Association of the United Kingdom 60* 151–156.

Clavero, M., L. Brotons, P. Pons, and D. Sol. (2009). Prominent role of invasive species in avian biodiversity loss. *Biological Conservation 142* (10): 2043–2049. DOI:10.1016/j.biocon.2009.03.034

Closek, C. J., S. Sunagawa, M. K. DeSalvo, Y. M. Piceno, T. Z. DeSantis, E. L. Brodie, M. X. Weber, C. R. Voolstra, G. L. Anderson, and M. Medina. (2014). Coral transcriptome and bacterial community profiles reveal distinct Yellow Band Disease states in *Orbicella faveolata*. *The ISME Journal 8* 2411–2422.

Cloyed, C. S., E. E. Hieb, M. K. Collins, K. P. DaCosta, and R. H. Carmichael. (2019). Linking use of ship channels by West Indian manatees (*Trichechus manatus*) to seasonal migration and habitat use. *Frontiers in Marine Science 6*. DOI:10.3389/fmars.2019.00318

Clyne, H., R. Leaper, and J. Kennedy. (1999). *Computer simulation of interactions between the North Atlantic right whale (Eubalaena glacialis) and shipping*. Presented at the Thirteenth Annual Conference of the European Cetacean Society. Valencia, Spain.

Coakes, A., S. Gowans, P. Simard, J. Giard, C. Vashro, and R. Sears. (2005). Photographic identification of fin whales (*Balaenoptera physalus*) off the Atlantic coast of Nova Scotia, Canada. *Marine Mammal Science 21* (2): 323–327.

Cohen, J. B. and C. Gratto-Trevor. (2011). Survival, site fidelity, and the population dynamics of piping plovers in Saskatchewan. *Journal of Field Ornithology 82* (4): 379–394. DOI:10.1111/j.1557-9263.2011.00341

Cole, T. V. N., P. Duley, M. Foster, A. Henry, and D. D. Morin. (2016). *2015 right whale aerial surveys of the Scotian Shelf and Gulf of St. Lawrence*. Northeast Fisheries Science Center Reference Document 16-02.

Cole, T. V. N., P. K. Hamilton, A. Henry, P. A. Duley, R. M. Pace, III, B. N. White, and T. R. Frasier. (2013). Evidence of a North Atlantic right whale *Eubalaena glacialis* mating ground. *Endangered Species Research 21* 55–64.

Coles, P. J. (2001). *Identifying beaked whales at sea in North Atlantic waters* (A report on the whales, dolphins and seabirds of the Bay of Biscay and English Channel). Portsmouth, United Kingdom: Organization Cetacea (ORCA).

Collazo, J. A., R. Boulon, and T. L. Tallevast. (1992). Abundance and growth patterns of *Chelonia mydas* in Culebra, Puerto Rico. *Journal of Herpetology 26* (3): 293–300.

Collazo, J. A., M. J. Krachey, K. H. Pollock, F. J. Pérez-Aguilo, J. P. Zegarra, and A. A. Mignucci-Giannoni. (2019). Population estimates of Antillean manatees in Puerto Rico: an analytical framework for aerial surveys using multi-pass removal sampling. *Journal of Mammalogy 100* (4): 1340-1349. DOI:10.1093/jmammal/gyz076

Collier, K. J., P. K. Probert, and M. Jeffries. (2016). Conservation of aquatic invertebrates: concerns, challenges and conundrums. *Aquatic Conservation: Marine and Freshwater Ecosystems 26* (5): 817-837.

Collins, M. R., S. G. Rogers, T. I. J. Smith, and M. L. Moser. (2000). Primary factors affecting sturgeon populations in the southeastern United States: Fishing mortality and degradation of essential habitats. *Bulletin of Marine Science 66* (3): 917–928.

Committee on Environment and Natural Resources. (2010). *Scientific assessment of hypoxia in U.S. coastal waters*. Washington, DC: Interagency Working Group on Harmful Algal Blooms, Hypoxia, and Human Health of the Joint Subcommittee on Ocean Science and Technology.

Committee on Taxonomy. (2023). *List of Marine Mammal Species and Subspecies (updated June 2023)*. Retrieved March 12, 2024, from https://marinemammalscience.org/science-and-publications/list-marine-mammal-species-subspecies/.

Committee on the Status of Endangered Wildlife in Canada. (2011). *Assessment and Status Report on the Northern Bottlenose Whale Hyperoodon ampullatus in Canada – 2011. Species at Risk Public Registry.* Retrieved July 7, 2016, from http://www.sararegistry.gc.ca/default.asp?lang=EN&n=3BF95D10-1.

Compagno, L. J. V. (1984). *FAO species catalogue. Sharks of the world. An annotated and illustrated catalogue of shark species known to date. Part 2. Carcharhiniformes* (FAO Fisheries Synopsis No. 125). Tiburon, CA: San Francisco State University.

Conant, T. A., P. H. Dutton, T. Eguchi, S. P. Epperly, C. C. Fahy, M. H. Godfrey, S. L. MacPherson, E. E. Possardt, B. A. Schroeder, J. A. Seminoff, M. L. Snover, C. M. Upite, and B. E. Witherington. (2009). *Loggerhead sea turtle (Caretta caretta) 2009 status review under the U.S. Endangered Species Act* (Report of the loggerhead biological review team to the National Marine Fisheries Service, August 2009). Silver Spring, MD: Loggerhead Biological Review Team.

Congdon, B. C., C. A. Erwin, D. R. Peck, G. B. Baker, M. C. Double, and P. O'Neill. (2007). Vulnerability of seabirds on the Great Barrier Reef to climate change. In J. E. Johnson & P. A. Marshall (Eds.), *Climate Change and the Great Barrier Reef: A Vulnerability Assessment* (pp. 427–463). Townsville, Australia: Great Barrier Reef Marine Park Authority and Australian Greenhouse Office.

Constantine, D. (2003). Geographic translocation of bats: Known and potential problems. *Emerging Infectious Diseases 9* (1): 17–21.

Cook, C. (2016). *Impacts Of Invasive Phragmites Australis On Diamondback Terrapin Nesting.* (Master's thesis). College of William and Mary, Williamsburg, VA.

Cook, T. R., M. Hamann, L. Pichegru, F. Bonadonna, D. Grémillet, and P. G. Ryan. (2011). GPS and time-depth loggers reveal underwater foraging plasticity in a flying diver, the Cape Cormorant. *Marine Biology 159* (2): 373–387. DOI:10.1007/s00227-011-1815-3

Cordes, E. E., D. C. Bergquist, and C. R. Fisher. (2009). Macro-ecology of Gulf of Mexico cold seeps. *Annual Review of Marine Science 1* (1): 143–168. DOI:10.1146/annurev.marine.010908.163912

Corn, M. L. (2010). *Deepwater Horizon Oil Spill: Coastal Wetland and Wildlife Impacts and Response*. Collingdale, PA: Diane Publishing Co.

Cornell Lab of Ornithology. (2002, January 1). *Red Phalarope: Phalaropus fulicarius*. *Birds of North America.* Retrieved May 30, 2017, from https://birdsna.org/Species-Account/bna/species/redpha1/introduction.

Cornell Lab of Ornithology. (2009). *All About Birds, Sandpipers, Phalaropes, and Allies (Order: Charadiiformes, Family: Scolopacidae)*. Retrieved 2010, from http://www.allaboutbirds.org/guide/browse_tax.aspx?family=53.

Cornell Lab of Ornithology. (2011). *All About Birds, Peregrine Falcon*. Retrieved 2011, from http://www.allaboutbirds.org/guide/Peregrine_Falcon/id?gclid=CLK8m7eW4KoCFQfd4AoduAzf6A.

Cornell Lab of Ornithology. (2013, December 4). *Red Knot: Calidris canutus*. *Birds of North America.* Retrieved May 11, 2017, from https://birdsna.org/Species-Account/bna/species/563/articles/introduction.

Cornell Lab of Ornithology. (2014). *Roseate Tern (Sterna dougallii)*. *Birds of North America.* Retrieved May 17, 2017, from https://birdsna.org/Species-Account/bna/species/370/articles/introduction.

Cornish, A. and A. M. Eklund. (2003). *Epinephelus striatus*. *IUCN 2012 Red List of Threatened Species. Version 2012.1.* Retrieved from http://www.iucnredlist.org/details/full/7862/0.

Costa, A. P. B., W. Mcfee, L. A. Wilcox, F. I. Archer, and P. E. Rosel. (2022). The common bottlenose dolphin (*Tursiops truncatus*) ecotypes of the western North Atlantic revisited: an integrative taxonomic investigation supports the presence of distinct species. *Zoological Journal of the Linnean Society 196* (4): 1608–1636. DOI:10.1093/zoolinnean/zlac025

Cotter, M. P. (2019). *Aerial Surveys for Protected Marine Species in the Norfolk Canyon Region: 2018–2019 Final Report*. Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Coyne, M. S., M. E. Monaco, and A. M. Landry, Jr. (2000). *Kemp's ridley habitat suitability index model* [Abstract]. Presented at the Eighteenth International Sea Turtle Symposium. Mazatlán, Mexico. Retrieved from http://www.nmfs.noaa.gov/pr/pdfs/species/turtlesymposium1998.pdf.

Craddock, J. E., P. T. Polloni, B. Hayward, and F. Wenzel. (2009). Food habits of Atlantic white-sided dolphins (*Lagenorhynchus acutus*) off the coast of New England. *Fishery Bulletin 107* (3): 384–394.

Craft, N. M., B. Russell, and S. Travis. (2001). *Identification of Gulf Sturgeon Spawning Habitats and Migratory Patterns in the Yellow and Escambia River Systems*. Tallahassee, FL: Florida Marine Research Institute, Florida Fish and Wildlife Conservation Commission.

Crain, C. M., B. S. Halpern, M. W. Beck, and C. V. Kappel. (2009). Understanding and Managing Human Threats to the Coastal Marine Environment. In R. S. Ostfeld & W. H. Schlesinger (Eds.), *The Year in Ecology and Conservation Biology, 2009* (pp. 39–62). Oxford, United Kingdom: Blackwell Publishing.

Crichton, K. A., J. D. Wilson, A. Ridgwell, F. Boscolo-Galazzo, E. H. John, B. S. Wade, and P. N. Pearson. (2023). What the geological past can tell us about the future of the ocean's twilight zone. *Nature Communications 14* (1): 2376. DOI:10.1038/s41467-023-37781-6

Croll, D. A., C. W. Clark, J. Calambokidis, W. T. Ellison, and B. R. Tershy. (2001). Effect of anthropogenic low-frequency noise on the foraging ecology of *Balaenoptera* whales. *Animal Conservation 4* 13–27.

Croquer, A., C. Bastidas, and D. Lipscomb. (2006). Folliculinid ciliates: A new threat to Caribbean corals? *Diseases of Aquatic Organisms 69* 75–78.

Crowe, L. M., M. W. Brown, P. J. Corkeron, P. K. Hamilton, C. Ramp, S. Ratelle, A. S. M. Vanderlaan, and T. V. N. Cole. (2021). In plane sight: a mark-recapture analysis of North Atlantic right whales in the Gulf of St. Lawrence. *Endangered Species Research 46* 227-251.

Crowell, S. E., A. M. Wells-Berlin, C. E. Carr, G. H. Olsen, R. E. Therrien, S. E. Ynnuzzi, and D. R. Ketten. (2015). A comparison of auditory brainstem responses across diving bird species. *Journal of Comparative Physiology A 201* (8): 803–815. DOI:10.1007/s00359-015-1024-5

Cryan, P. and A. Brown. (2007). Migration of bats past a remote island offers clues toward the problem of bat fatalities at wind turbines. *Biological Conservation* 1–11. DOI:10.1016/j.biocon.2007.05.019

CSA Ocean Sciences Inc. (2023). *Assessment of Impacts to Marine Mammals, Sea Turtles, and ESA-Listed Fish Species Revolution Wind Offshore Wind Farm*. Providence, RI: Revolution Wind, LLC.

CT.gov. (2023, July 24). *Connecticut Northern Long-eared bat Observations by Town*. Retrieved May 30, 2024, from https://portal.ct.gov/-/media/deep/nddb/nolongearedbat-map.pdf.

Cuevas, E., M. d. l. A. Liceaga-Correa, A. Uribe-Martínez, S. A. Gallegos-Fernández, F. Moncada-Gavilán, R. J. González-Díaz-Mirón, M. C. López-Castro, V. Guzmán-Hernández, and M. López. (2022). Marine turtle hotspots in the Gulf of Mexico and Mesoamerican Reef: Strengthening management and preparedness. *Frontiers in Marine Science 9* 1–11. DOI:10.3389/fmars.2022.1059678

Cummings, E., R. McAlarney, W. McLellan, and D. A. Pabst. (2016). *Aerial Surveys for Protected Species in the Jacksonville Operating Area: 2015 Annual Progress Report*. Virginia Beach, VA: U.S. Fleet Forces Command.

Cummings, E. W., D. A. Pabst, J. E. Blum, S. G. Barco, S. J. Davis, V. G. Thayer, N. Adimey, and W. A. McLellan. (2014). Spatial and temporal patterns of habitat use and mortality of the Florida manatee (*Trichechus manatus latirostris*) in the mid-Atlantic states of North Carolina and Virginia from 1991 to 2012. *Aquatic Mammals 40* (2): 126–138. DOI:10.1578/AM.40.2.2014.126

Cummings, W. C. (1985). Bryde's whale, *Balaenoptera edeni* Anderson, 1878. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 3, pp. 137–154). San Diego, CA: Academic Press.

Currie, J. J., S. H. Stack, and G. D. Kaufman. (2017). Modelling whale-vessel encounters: The role of speed in mitigating collisions with humpback whales (*Megaptera novaeangliae*). *Journal of Cetacean Research Management 17* 57–63.

Curry, B. E. and J. Smith. (1997). Phylogeographic structure of the bottlenose dolphin (*Tursiops truncatus*): Stock identification and implications for management. In A. E. Dizon, S. J. Chivers, & W. F. Perrin (Eds.), *Molecular Genetics of Marine Mammals* (pp. 227–247). Lawrence, KS: Society for Marine Mammalogy.

Curtis, K. J. and A. Schneider. (2011). Understanding the demographic implications of climate change: estimates of localized population predictions under future scenarios of sea-level rise. *Population and Environment 33* (1): 28–54. DOI:10.1007/s11111-011-0136-2

D'Vincent, C. G., R. M. Nilson, and R. E. Hanna. (1985). Vocalization and coordinated feeding behavior of the humpback whale in southeastern Alaska. *The Scientific Reports of the Whales Research Institute 36* 41–47.

Dadswell, M. J. (2006). A review of the status of Atlantic sturgeon in Canada, with comparisons to populations in the United States and Europe. *Fisheries 31* (5): 218–229.

Dadswell, M. J., B. D. Taubert, T. S. Squiers, D. Marchette, and J. Buckley. (1984). *Synopsis of biological data on shortnose sturgeon, Acipenser brevirostrum LeSueur 1818* (FAO Fisheries Synopsis No. 140). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Dahl, T. E. and S. M. Stedman. (2013). *Status and trends of wetlands in the coastal watersheds of the Conterminous United States 2004 to 2009*. Washington, DC and Silver Spring, MD: U.S. Department of the Interior, Fish and Wildlife Service, and National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Dahlheim, M. E. and J. E. Heyning. (1999). Killer whale, *Orcinus orca* (Linnaeus, 1758). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 281–322). San Diego, CA: Academic Press.

Dalebout, M. L., D. E. Ruzzante, H. Whitehead, and N. I. Oien. (2006). Nuclear and mitochondrial markers reveal distinctiveness of a small population of bottlenose whale (*Hyperoodon ampullatus*) in the western North Atlantic. *Molecular Ecology 15* 3115–3129. DOI:10.1111/j.1365-294X-2006.03004

Daly-Engel, T. S., K. D. Seraphin, K. N. Holland, J. P. Coffey, H. A. Nance, R. J. Toonen, and B. W. Bowen. (2012). Global phylogeography with mixed-marker analysis reveals male-mediated dispersal in the endangered scalloped hammerhead shark (*Sphyrna lewini*). *PLoS ONE 7* (1): e29986. DOI:10.1371/journal.pone.0029986

Danish, S. T. Hassan, M. A. Baloch, N. Mahmood, and J. Zhang. (2019). Linking economic growth and ecological footprint through human capital and biocapacity. *Sustainable Cities and Society 47* 101516. DOI:https://doi.org/10.1016/j.scs.2019.101516

Dannheim, J., L. Bergström, S. N. R. Birchenough, R. Brzana, A. R. Boon, J. W. P. Coolen, J.-C. Dauvin, I. De Mesel, J. Derweduwen, A. B. Gill, Z. L. Hutchison, A. C. Jackson, U. Janas, G. Martin, A. Raoux, J.

Reubens, L. Rostin, J. Vanaverbeke, T. A. Wilding, D. Wilhelmsson, and S. Degraer. (2019). Benthic effects of offshore renewables: identification of knowledge gaps and urgently needed research. *ICES Journal of Marine Science 77* (3): 1092–1108. DOI:10.1093/icesjms/fsz018

Daoust, P.-Y., E. L. Couture, T. Wimmer, and L. Bourque. (2017). *Incident Report: North Atlantic Right Whale Mortality Event In the Gulf of St. Lawrence, 2017*. Saskatoon, Canada: Canadian Wildlife Health Cooperative, Marine Animal Response Society, and Fisheries and Oceans Canada.

Davenport, J. (1988). Do diving leatherbacks pursue glowing jelly? *British Herpetological Society Bulletin 24* 20–21.

Davies, J. L. (1957). The geography of the gray seal. *Journal of Mammalogy 38* (3): 297–310.

Davies, K. T. A., M. W. Brown, P. K. Hamilton, A. R. Knowlton, C. T. Taggart, and A. S. M. Vanderlaan. (2019). Variation in North Atlantic right whale *Eubalaena glacialis* occurrence in the Bay of Fundy, Canada, over three decades. *Endangered Species Research 39* 159-171.

Davis, G. E. (1982). A century of natural change in coral distribution at the Dry Tortugas: A comparison of reef maps from 1881 and 1976. *Bulletin of Marine Science 32* (2): 608–623.

Davis, G. E., M. F. Baumgartner, J. M. Bonnell, J. Bell, C. Berchok, J. Bort Thornton, S. Brault, G. Buchanan, R. A. Charif, D. Cholewiak, C. W. Clark, P. Corkeron, J. Delarue, K. Dudzinski, L. Hatch, J. Hildebrand, L. Hodge, H. Klinck, S. Kraus, B. Martin, D. K. Mellinger, H. Moors-Murphy, S. Nieukirk, D. P. Nowacek, S. Parks, A. J. Read, A. N. Rice, D. Risch, A. Širović, M. Soldevilla, K. Stafford, J. E. Stanistreet, E. Summers, S. Todd, A. Warde, and S. M. Van Parijs. (2017). Long-term passive acoustic recordings track the changing distribution of North Atlantic right whales (*Eubalaena glacialis*) from 2004 to 2014. *Scientific Reports 7* (1): 13460. DOI:10.1038/s41598-017-13359-3

Davis, G. E., M. F. Baumgartner, P. J. Corkeron, J. Bell, C. Berchok, J. M. Bonnell, J. Bort Thornton, S. Brault, G. A. Buchanan, D. M. Cholewiak, C. W. Clark, J. Delarue, L. T. Hatch, H. Klinck, S. D. Kraus, B. Martin, D. K. Mellinger, H. Moors-Murphy, S. Nieukirk, D. P. Nowacek, S. E. Parks, D. Parry, N. Pegg, A. J. Read, A. N. Rice, D. Risch, A. Scott, M. S. Soldevilla, K. M. Stafford, J. E. Stanistreet, E. Summers, S. Todd, and S. M. Van Parijs. (2020). Exploring movement patterns and changing distributions of baleen whales in the western North Atlantic using a decade of passive acoustic data. *Global Change Biology 26* (9): 4812-4840. DOI:https://doi.org/10.1111/gcb.15191

Davis, R. W., W. E. Evans, and B. Würsig, (Eds.). (2000). *Cetaceans, Sea Turtles and Seabirds in the Northern Gulf of Mexico: Distribution, Abundance and Habitat Associations*. New Orleans, LA: U.S. Geological Survey, Biological Resource Division; and Minerals Management Service, Gulf of Mexico Region.

Davis, R. W. and G. S. Fargion. (1996). *Distribution and Abundance of Marine Mammals in the North-central and Western Gulf of Mexico*. Galveston, TX: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Davis, R. W., G. S. Fargion, N. May, T. D. Leming, M. Baumgartner, W. E. Evans, L. J. Hansen, and K. Mullin. (1998). Physical habitat of cetaceans along the continental slope in the north-central and western Gulf of Mexico. *Marine Mammal Science 14* (3): 490–507.

Davis, R. W., N. Jaquet, D. Gendron, U. Markaida, G. Bazzino, and W. Gilly. (2007). Diving behavior of sperm whales in relation to behavior of a major prey species, the jumbo squid, in the Gulf of California, Mexico. *Marine Ecology Progress Series 333* 291–302.

Davis, R. W., J. G. Ortega-Ortiz, C. A. Ribic, W. E. Evans, D. C. Biggs, P. H. Ressler, R. B. Cady, R. R. Leben, K. D. Mullin, and B. Würsig. (2002). Cetacean habitat in the northern oceanic Gulf of Mexico. *Deep-Sea Research I 49* 121–142.

Dawes, C. J. (1998). *Marine Botany* (2nd ed.). New York, NY: John Wiley and Sons, Inc.

Day, R. D., R. D. McCauley, Q. P. Fitzgibbon, and J. M. Semmens. (2016). Seismic air gun exposure during early-stage embryonic development does not negatively affect spiny lobster *Jasus edwardsii* larvae (Decapoda: Palinuridae). *Scientific Reports 6* (22723): 9. DOI:10.1038/srep22723

Dearborn, D. C., A. D. Anders, and P. G. Parker. (2001). Sexual dimorphism, extrapair fertilizations, and operational sex ratio in great frigatebirds (*Fregata minor*). *Behavioral Ecology 12* (6): 746–752. DOI:10.1093/beheco/12.6.746

Debich, A. J., S. Baumann-Pickering, A. Širović, J. A. Hildebrand, A. M. Brewer, K. E. Frasier, R. T. Gresalfi, S. T. Herbert, S. C. Johnson, A. C. Rice, L. M. Varga, S. M. Wiggins, L. E. W. Hodge, J. E. Stanistreet, and A. J. Read. (2016). *Passive Acoustic Monitoring for Marine Mammals in the Virginia Capes Range Complex October 2012–April 2015.* Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Deepwater Horizon Marine Mammal Injury Quantification Team. (2015). Models and analyses for the quantification of injury to Gulf of Mexico cetaceans from the Deepwater Horizon oil spill. *DWH Marine Mammal NRDA Technical Working Group Report*.

Deepwater Horizon Natural Resource Damage Assessment Trustees. (2016). *Deepwater Horizon Oil Spill: Final Programmatic Damage Assessment and Restoration Plan and Final Programmatic Environmental Impact Statement*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Defenders of Wildlife. (2015a). *A Petition to List the Giant Manta Ray (Manta birostris), Reef Manta Ray (Manta alfredi), and Caribbean Manta Ray (Manta c.f. birostris) as Endangered, or Alternatively as Threatened, Species Pursuant to the Endangered Species Act and for the Concurrent Designation of Critical Habitat*. Denver, CO: Defenders of Wildlife.

Defenders of Wildlife. (2015b). *A Petition to List the Oceanic Whitetip Shark (Carcharhinus longimanus) as an Endangered, or Alternatively as a Threatened, Species Pursuant to the Endangered Species Act and for the Concurrent Designation of Critical Habitat*. Denver, CO: Defenders of Wildlife.

DeHart, P. A. P. (2002). *The distribution and abundance of harbor seals (Phoca vitulina concolor) in the Woods Hole region.* (Unpublished thesis). Boston University, Boston, MA.

Dellinger, T., V. Zekovic, and M. Radeta. (2022). Long-term monitoring of in-water abundance of juvenile pelagic loggerhead sea turtles (*Caretta caretta*): Population trends in relation to North Atlantic oscillation and nesting. *Frontiers in Marine Science 9* 1–14. DOI:10.3389/fmars.2022.877636

Denkinger, J., M. Parra, J. P. Muñoz, C. Carrasco, J. C. Murillo, E. Espinosa, F. Rubianes, and V. Koch. (2013). Are boat strikes a threat to sea turtles in the Galapagos Marine Reserve? *Ocean & Coastal Management 80* 29–35.

Denzinger, A. and H. U. Schnitzler. (2013). Bat guilds, a concept to classify the highly diverse foraging and echolocation behaviors of microchiropteran bats. *Frontiers in Physiology 4* 164. DOI:10.3389/fphys.2013.00164

Deslauriers, D. and J. D. Kieffer. (2012). The effects of temperature on swimming performance of juvenile shortnose sturgeon (*Acipenser brevirostrum*). *Journal of Applied Ichthyology 28* (2): 176–181. DOI:10.1111/j.1439-0426.2012.01932

Deutsch, C. J., J. P. Reid, R. K. Bonde, D. E. Easton, H. I. Kochman, and T. J. O'Shea. (2003). Seasonal movements, migratory behavior, and site fidelity of West Indian manatees along the Atlantic coast of the United States. *Wildlife Monographs 151* 1–77.

Diaz, R. J. and R. Rosenberg. (2008). Spreading dead zones and consequences for marine ecosystems. *Science 321* (5891): 926–929.

DiMatteo, A., G. Lockhart, and S. Barco. (2021). Normalizing home ranges of immature Kemp's ridley turtles (*Lepidochelys kempii*) in an important estuarine foraging area to better assess their spatial distribution. *Marine Biology Research 17* 1–15. DOI:10.1080/17451000.2021.1896004

DiMatteo, A., G. Lockhart, and S. Barco. (2022). Habitat models and assessment of habitat partitioning for Kemp's ridley and loggerhead marine turtles foraging in Chesapeake Bay (USA). *Endangered Species Research 47* 91–107. DOI:doi.org/10.3354/esr01168

DiMatteo, A., J. J. Roberts, D. T. Jones-Farrand, L. Garrison, K. Hart, R. D. Kenney, W. A. McLellan, K. Lomac-MacNair, D. Palka, M. E. Rickard, K. E. Roberts, A. M. Zoidis, and L. Sparks. (2024). Sea turtle density surface models along the United States Atlantic coast. *Endangered Species Research 53* 227-245. DOI:10.3354/esr01298

Dobson, A. L. F. and J. Madeiros. (2009). Threats facing Bermuda's breeding seabirds: Measures to assist future breeding success. In T. D. Rich, C. Arizmendi, D. W. Demarest, & C. Thompson (Eds.), *Tundra to Tropics: Connecting Birds, Habitats and People: Proceedings of the Fourth International Partners in Flight Conference* (pp. 223–226). Hamilton, Bermuda: Partners in Flight.

Dodd, C. K., Jr. (1988). *Synopsis of the Biological Data on the Loggerhead Sea Turtle, Caretta caretta (Linnaeus 1758)*. Washington, DC: U.S. Fish and Wildlife Service.

Dodge, K. L., B. Galuardi, T. J. Miller, and M. E. Lutcavage. (2014). Leatherback Turtle Movements, Dive Behavior, and Habitat Characteristics in Ecoregions of the Northwest Atlantic Ocean. *PLoS ONE 9* (3): e91726.

Doksaeter, L., E. Olsen, L. Nottestad, and A. Ferno. (2008). Distribution and feeding ecology of dolphins along the Mid-Atlantic Ridge between Iceland and the Azores. *Deep Sea Research II 55* 243–253.

Dolar, M. L. L. (2009). Fraser's dolphin, *Lagenodelphis hosei*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 485–487). Cambridge, MA: Academic Press.

Donahue, M. A. and W. L. Perryman. (2009). Pygmy killer whale, *Feresa attenuata*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 938–939). Cambridge, MA: Academic Press.

Donaton, J., K. Durham, R. Cerrato, J. Schwerzmann, and L. H. Thorne. (2019). Long-term changes in loggerhead sea turtle diet indicate shifts in the benthic community associated with warming temperatures. *Estuarine, Coastal and Shelf Science 218* 139–147. DOI:10.1016/j.ecss.2018.12.008

Donovan, G. P. (1991). A Review of International Whaling Commission Stock Boundaries. *Reports of the International Whaling Commission 13* 39–68.

Dooling, R. (2002). *Avian Hearing and the Avoidance of Wind Turbines*. College Park, MD: University of Maryland.

Dooling, R. J. and A. N. Popper. (2000). Hearing in birds and reptiles. In R. J. Dooling, R. R. Fay, & A. N. Popper (Eds.), *Comparative Hearing in Birds and Reptiles* (Vol. 13, pp. 308–359). New York, NY: Springer-Verlag.

Dorcas, M. E., J. D. Willson, R. N. Reed, R. W. Snow, M. R. Rochford, M. A. Miller, W. E. Meshaka, P. T. Andreadis, F. J. Mazzotti, and C. M. Romagosa. (2012). Severe mammal declines coincide with proliferation of invasive Burmese pythons in Everglades National Park. *Proceedings of the National Academy of Sciences 109* (7): 2418–2422.

dos Santos, R. A. and M. Haimovici. (2001). Cephalopods in the diet of marine mammals stranded or incidentally caught along southeastern and southern Brazil (21–34°S). *Fisheries Research 52* 99–112.

Dowling, Z. (2018). *NOT GONE WITH THE WIND: Addressing Effects of Offshore Wind Development on Bat Species in the Northeastern United States.* (Doctor of Philosophy Dissertation). University of Massachusetts Amherst, Amherst, MA.

Drake, V. A. and D. R. Reynolds. (2012). Migrations of pest and beneficial insects *Radar entomology: observing insect flight and migration*. CABI.

Draud, M., M. Bossert, and S. Zimnavoda. (2004). Predation on hatchling and juvenile diamondback terrapins (*Malaclemys terrapin*) by the Norway rat (*Rattus norvegicus*). *Journal of Herpetology 38* (3): 467–470.

Duarte, C. M., N. Marbà, D. Krause-Jensen, and M. Sánchez-Camacho. (2007). Testing the predictive power of seagrass depth limit models. *Estuaries and Coasts 30* (4): 652–656. DOI:10.1007/BF02841962

Duffield, D. A., S. H. Ridgway, and L. H. Cornell. (1983). Hematology distinguishes coastal and offshore forms of dolphins (*Tursiops*). *Canadian Journal of Zoology 61* 930–933.

Duffy, D. C. (1986). Foraging at patches: Interactions between common and roseate terns. *Ornis Scandinavica 17* (4): 47–52.

Dugan, J. E., D. M. Hubbard, I. F. Rodil, D. L. Revell, and S. Schroeter. (2008). Ecological effects of coastal armoring on sandy beaches. *Marine Ecology 29* (s1): 160–170. DOI:https://doi.org/10.1111/j.1439-0485.2008.00231.x

Dunbar, E. A. (2015). *Depth Preferences of Sturgeon In Critical Habitat.* (B.S., Marine Science). The Honors College, University of Maine, Orono, ME.

Duncan, E. M., A. C. Broderick, W. J. Fuller, T. S. Galloway, M. H. Godfrey, M. Hamann, C. J. Limpus, P. K. Lindeque, A. G. Mayes, L. C. Omeyer, D. Santillo, R. T. E. Snape, and B. J. Godley. (2018). Microplastic ingestion ubiquitous in marine turtles. *Global Change Biology 25* 744–752. DOI:10.1111/gcb.14519

Duncan, K. M. and K. N. Holland. (2006). Habitat use, growth rates and dispersal patterns of juvenile scalloped hammerhead sharks, *Sphyrna lewini,* in a nursery habitat. *Marine Ecology Progress Series 312* 211–221. DOI:10.3354/meps312211

Dunton, K. J., A. Jordaan, K. A. McKown, D. O. Conover, and M. G. Frisk. (2010). Abundance and distribution of Atlantic sturgeon (*Acipenser oxyrinchus*) within the Northwest Atlantic Ocean, determined from five fishery-independent surveys. *Fishery Bulletin 108* (4).

Dustan, P. and J. C. Halas. (1987). Changes in the reef-coral community of Carysfort Reef, Key Largo, Florida: 1974 to 1982. *Coral Reefs 6* (2): 91–106. DOI:10.1007/BF00301378

Dutil, J. D. and J. M. Coutu. (1988). Early marine life of Atlantic salmon, *Salmo salar,* postsmolts in the Northern Gulf of St. Lawrence. *Fishery Bulletin 86* (2): 197–212.

Eckert, K. and K. Hart. (2021). *Threat Assessment: Northwest Atlantic Leatherback Sea Turtles, Dermochelys coriacea, with Special Emphasis on Trinidad & Tobago and the Guianas* (Widecast Technical Report). Godfrey, IL: Wider Caribbean Sea Turtle Conservation Network.

Eckert, S. A. (2002). Distribution of juvenile leatherback sea turtle, *Dermochelys coriacea,* sightings. *Marine Ecology Progress Series 230* 289–293.

Eckert, S. A., K. L. Eckert, P. Ponganis, and G. L. Kooyman. (1989). Diving and foraging behavior of leatherback sea turtles (*Dermochelys coriacea*). *Canadian Journal of Zoology 67* 2834–2840.

Edmonds, N. J., C. J. Firmin, D. Goldsmith, R. C. Faulkner, and D. T. Wood. (2016). A review of crustacean sensitivity to high amplitude underwater noise: Data needs for effective risk assessment in relation to UK commercial species. *Marine Pollution Bulletin 108* 5–11. DOI:10.1016/j.marpolbul.2016.05.006

Edmunds, P. J. and R. Elahi. (2007). The demographics of a 15-year decline in cover of the Caribbean reef coral *Montastraea annularis*. *Ecological Monographs 77* (1): 3–18. DOI:10.1890/05-1081

Edwards, E. F., C. Hall, T. J. Moore, C. Sheredy, and J. V. Redfern. (2015). Global distribution of fin whales (*Balaenoptera physalus*) in the post-whaling era (1980–2012). *Mammal Review 45* 197–214. DOI:10.1111/mam.12048

Edwards, H. H. (2013). Potential impacts of climate change on warmwater megafauna: The Florida manatee example (*Trichechus manatus latirostris*). *Climatic Change 121* (4): 727–738. DOI:10.1007/s10584-013-0921-2

Edwards, R. E., K. J. Sulak, M. T. Randall, and C. B. Grimes. (2003). Movements of Gulf sturgeon *(Acipenser oxyrinchus desotoi)* in nearshore habitat as determined by acoustic telemetry. *Gulf of Mexico Science 21* (1): 59–70.

Ehrhardt, N. M. and V. K. W. Deleveaux. (2007). The Bahamas' Nassau grouper (*Epinephelus striatus*) fishery – Two assessment methods applied to a data-deficient coastal population. *Fisheries Research 87* (1): 17–27.

Ehrlich, P. R., D. S. Dobkin, and D. Wheye. (1988). *The Birder's Handbook: A Field Guide to the Natural History of North American Birds*. New York, NY: Simon & Schuster, Inc.

Eisenberg, J. F. and J. Frazier. (1983). A leatherback turtle (*Dermochelys coriacea*) feeding in the wild. *Journal of Herpetology 17* (1): 81–82.

Elliott-Smith, E., M. Bidwell, A. Holland, and S. Haig. (2015). *Data from the 2011 International Piping Plover Census*. Reston, VA: U.S. Geological Survey.

Ellis, J. C., M. J. Shulman, M. Wood, J. D. Witman, and S. Lozyniak. (2007). Regulation of intertidal food webs by avian predators on New England rocky shores. *Ecology 88* (4): 853–863. DOI:10.1890/06-0593

Elphick, J. (2007). *Atlas of Bird Migration: Tracing the Great Journeys of the World's Birds*. Buffalo, NY: Firefly Books.

Elsey, R. M., N. Kinler, V. Lance, and W. Parke Moore, III. (2006). *Effects of Hurricanes Katrina and Rita on alligators (Alligator mississippiensis) in Louisiana*. Presented at the 18th Working Meeting of the International Union, for Conservation of Nature-Species Survival Commission, Crocodile Specialist Group. Montelimar, France.

Elsey, R. M. and A. R. Woodward. (2010). American Alligator, *Alligator mississippiensis*. In S. C. Manolis & C. Stevenson (Eds.), *Crocodiles. Status Survey and Conservation Action Plan* (3rd ed., pp. 1–4). Gland, Switzerland and Cambridge,United Kingdom: Crocodile Specialist Group, International Union for Conservation of Nature.

Engelhaupt, A., T. A. Jefferson, J. M. Aschettino, and J. T. Bell. (2022). Distribution, abundance and sighting patterns of multiple stocks of bottlenose dolphins (*Tursiops truncatus*) in coastal Virginia waters. *Journal of Cetacean Research Management 23* (1): 109–125. DOI:10.47536/jcrm.v23i1.222

Engelhaupt, D. T., T. Pusser, J. M. Aschettino, A. G. Engelhaupt, M. P. Cotter, M. F. Richlen, and J. T. Bell. (2020). Blue whale (*Balaenoptera musculus*) sightings off the coast of Virginia. *Marine Biodiversity Records 13* (1): 6. DOI:10.1186/s41200-020-00189-y

Engeman, R. M., D. Addison, and J. C. Griffin. (2016). Defending against disparate marine turtle nest predators: Nesting success benefits from eradicating invasive feral swine and caging nests from raccoons. *Oryx 50* (02): 289–295.

Engeman, R. M., R. W. Byrd, J. Dozier, M. A. McAlister, J. O. Edens, E. M. Kierepka, T. J. Smyser, and N. Myers. (2019). Feral swine harming insular sea turtle reproduction: The origin, impacts, behavior and elimination of an invasive species. *Acta Oecologica 99* 1–42. DOI:10.1016/j.actao.2019.103442

Enticott, J. and D. Tipling. (1997). *Seabirds of the World: The Complete Reference* (1st ed.). Mechanicsburg, PA: Stackpole Books.

Environmental Protection in the Caribbean. (2015). *Radar Surveys for the Endangered Black-capped Petrel on Dominica, West Indies*. Green Cove Springs, FL: Environmental Protection in the Caribbean.

Epperly, S. P., J. Braun, and A. J. Chester. (1995a). Aerial surveys for sea turtles in North Carolina inshore waters. *Fishery Bulletin 93* 254–261.

Epperly, S. P., J. Braun, A. J. Chester, F. A. Cross, J. V. Merriner, and P. A. Tester. (1995b). Winter distribution of sea turtles in the vicinity of Cape Hatteras and their interactions with the summer flounder trawl fishery. *Bulletin of Marine Science 56* (2): 547–568.

Epperly, S. P., J. Braun, and A. Veishlow. (1995c). Sea turtles in North Carolina waters. *Conservation Biology 9* (2): 384–394.

Epple, A. L., J. T. Daniel, and A. M. Costidis. (2023). *Virginia Sea Turtle & Marine Mammal Stranding Network 2022 Grant Report* (Final Report to the Virginia Coastal Zone Management Program, NOAA CZM Grant NA21NOS4190152, Task 49. VAQF Scientific Report Volume 2023 No. 01). Virginia Beach, VA: Virginia Aquarium Foundation Stranding Response Program.

Erdle, S. Y., J. L. Davis, and K. G. Sellner (Eds.). (2006). *Management, Policy, Science, and Engineering of Nonstructural Erosion Control in the Chesapeake Bay: Proceedings of the 2006 Living Shoreline Summit*. Edgewater, MD: Chesapeake Research Consortium Publication.

Erdman, D. S. (1970). Marine mammals from Puerto Rico to Antigua. *Journal of Mammalogy 51* 636–639.

Erdman, D. S., J. Harms, and M. Marcial-Flores. (1973). Cetacean records from the northeastern Caribbean region. *Cetology 17* 1–14.

Erickson, D. L., A. Kahnle, M. J. Millard, E. A. Mora, M. Bryja, A. Higgs, J. Mohler, M. DuFour, G. Kenney, J. Sweka, and E. K. Pikitch. (2011). Use of pop-up satellite archival tags to identify oceanic-migratory patterns for adult Atlantic Sturgeon, *Acipenser oxyrinchus oxyrinchus* Mitchell, 1815. *Journal of Applied Ichthyology 27* (2): 356–365. DOI:https://doi.org/10.1111/j.1439-0426.2011.01690.x

Eriksson, C. and H. Burton. (2003). Origins and biological accumulation of small plastic particles in fur seals from Macquarie Island. *Ambio 32* (6): 380–384.

Evans, W. E. (1994). Common dolphin, white-bellied porpoise—*Delphinus delphis* Linnaeus, 1758. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 191–224). New York, NY: Academic Press.

Eversole, C. B., S. E. Henke, D. B. Wester, B. M. Ballard, and R. L. Powell. (2015). Responses of American Alligators (*Alligator mississippiensis*) to Environmental Conditions: Implications for Population and Ecosystem Monitoring. *Herpetologica 71* (1): 37–45.

Fais, A., N. Aguilar Soto, M. Johnson, C. Perez-Gonzales, P. J. O. Miller, and P. T. Madsen. (2015). Sperm whale echolocation behavior reveals a directed, prior-based search strategy informed by prey distribution. *Behavioural Ecology and Sociobiology 69* (4): 663–674. DOI:10.1007/s00265-015-1877-1

Falcone, E. A., G. S. Schorr, A. B. Douglas, J. Calambokidis, E. Henderson, M. F. McKenna, J. Hildebrand, and D. Moretti. (2009). Sighting characteristics and photo-identification of Cuvier's beaked whales (*Ziphius cavirostris*) near San Clemente Island, California: A key area for beaked whales and the military? *Marine Biology 156* 2631–2640.

Farmer, N. A., L. P. Garrison, C. Horn, M. Miller, T. Gowan, R. D. Kenney, M. Vukovich, J. R. Willmott, J. Pate, D. Harry Webb, T. J. Mullican, J. D. Stewart, K. Bassos-Hull, C. Jones, D. Adams, N. A. Pelletier, J. Waldron, and S. Kajiura. (2022). The distribution of manta rays in the western North Atlantic Ocean off the eastern United States. *Scientific Reports 12* (1): 6544. DOI:10.1038/s41598-022-10482-8

Farrell, S. (2021, December 9). *Water Quality Parameters and Terrapins in the Chesapeake Bay*. Retrieved January 24, 2023, from https://storymaps.arcgis.com/stories/2e9ffd6f13df42938105619a51716d29/print.

Fast, M. D., M. S. Sokolowski, K. J. Dunton, and P. R. Bowser. (2009). *Dichelesthium oblongum* (Copepoda: Dichelesthiidae) infestation in wild-caught Atlantic sturgeon, *Acipenser oxyrinchus oxyrinchus*. *ICES Journal of Marine Science 66* (10): 2141–2147. DOI:doi: 10.1093/icesjms/fsp186

Fautin, D., P. Dalton, L. S. Incze, J. Leong, C. Pautzke, A. Rosenberg, P. Sandifer, G. Sedberry, J. W. Tunnell, I. Abbott, R. E. Brainard, M. Brodeur, L. E. Eldredge, M. Feldman, F. Moretzsohn, P. S. Vroom, M. Wainstein, and N. Wolff. (2010). An overview of marine biodiversity in United States waters. *PLoS ONE 5* (8): e11914. DOI:10.1371/journal.pone.0011914

Fay, C., M. Bartron, S. Craig, A. Hecht, J. Pruden, R. Saunders, T. Sheehan, and J. Trial. (2006). *Status Review for Anadromous Atlantic Salmon (Salmo salar) in the United States*. Silver Spring, MD: U.S. Fish and Wildlife Service; and National Marine Fisheries Service.

Federal Emergency Management Agency. (2012). *Draft Environmental Assessment for Canaveral Port Authority Port Wide Interoperable Communications Infrastructure Project Cape Canaveral, Brevard County, Florida*. Atlanta, GA: U.S. Department of Homeland Security, Federal Emergency Management Agency.

Feitosa, L. M., A. P. B. Martins, T. Giarrizzo, W. Macedo, I. L. Monteiro, R. Gemaque, J. L. S. Nunes, F. Gomes, H. Schneider, I. Sampaio, R. Souza, J. B. Sales, L. F. Rodrigues-Filho, L. Tchaicka, and L. F. Carvalho-Costa. (2018). DNA-based identification reveals illegal trade of threatened shark species in a global elasmobranch conservation hotspot. *Scientific Reports 8* (1): 3347. DOI:10.1038/s41598-018-21683-5

Ferguson, M. C., C. Curtice, J. Harrison, and S. M. Van Parijs. (2015). Biologically important areas for cetaceans within U.S. waters – Overview and rationale. *Aquatic Mammals (Special Issue) 41* (1): 2–16. DOI:10.1578/am.41.1.2015.2

Fergusson, I. K., L. J. V. Compagno, and M. A. Marks. (2000). Predation by white sharks *Carcharodon carcharias* (Chondrichthyes: Lamnidae) upon chelonians, with new records from the Mediterranean Sea and a first record of the ocean sunfish *Mola mola* (Osteichthyes: Molidae) as stomach contents. *Environmental Biology of Fishes 58* 447–453.

Fertl, D., A. Acevedo-Gutiérrez, and F. L. Darby. (1996). A report of killer whales (*Orcinus orca*) feeding on a carcharhinid shark in Costa Rica. *Marine Mammal Science 12* (4): 606–611.

Fertl, D., T. A. Jefferson, I. B. Moreno, A. N. Zerbini, and K. D. Mullin. (2003). Distribution of the Clymene dolphin, *Stenella clymene*. *Mammal Review 33* 253–271.

Fertl, D. and S. Leatherwood. (1997). Cetacean interactions with trawls: A preliminary review. *Journal of Northwest Atlantic Fishery Science 22* 219–248.

Fertl, D., A. J. Schiro, and D. Peake. (1997). Coordinated feeding by Clymene dolphins (*Stenella clymene*) in the Gulf of Mexico. *Aquatic Mammals 23* (2): 111–112.

Fertl, D. and B. Würsig. (1995). Coordinated feeding by Atlantic spotted dolphins (*Stenella frontalis*) in the Gulf of Mexico. *Aquatic Mammals 21* 3–5.

Figgener, C., D. Chacón-Chaverri, M. P. Jensen, and H. Feldhaar. (2016). Paternity re-visited in a recovering population of Caribbean leatherback turtles (*Dermochelys coriacea*). *Journal of Experimental Marine Biology and Ecology 475* 114–123.

Finkbeiner, E. M., B. P. Wallace, J. E. Moore, R. L. Lewison, L. B. Crowder, and A. J. Read. (2011). Cumulative estimates of sea turtle bycatch and mortality in USA fisheries between 1990 and 2007. *Biological Conservation 144* (11): 2719–2727.

Fish, M. R., I. M. Cote, J. A. Horrocks, B. Mulligan, A. R. Watkinson, and A. P. Jones. (2008). Construction setback regulations and sea-level rise: Mitigating sea turtle nesting beach loss. *Ocean & Coastal Management 51* (4): 330–341.

Fishman, J., K. MacKinnon, and S. Baker. (2009). *Crocodylus acutus*. *University of Michigan Animal Diversity Web.* Retrieved from http://animaldiversity.ummz.umich.edu/site/accounts/information/Crocodylus_acutus.html.

Flávio, H., R. Seitz, D. Eggleston, J. C. Svendsen, and J. Støttrup. (2023). Hard-bottom habitats support commercially important fish species: A systematic review for the North Atlantic Ocean and Baltic Sea. *PeerJ 11* e14681.

Fleming, A. H., C. T. Clark, J. Calambokidis, and J. Barlow. (2016). Humpback whale diets respond to variance in ocean climate and ecosystem conditions in the California Current. *Global Change Biology 22* (3): 1214–1224. DOI:10.1111/gcb.13171

Florida Bat Conservancy. (2024). *Accidental Bat Species in Florida*. Retrieved May 30, 2024, from https://www.floridabats.org/accidental-species.html.

Florida Fish and Wildlife Conservation Commission. (2009). *American crocodile, Crocodylus acutus*. Retrieved 5, from http://myfwc.com/crocodile/.

Florida Fish and Wildlife Conservation Commission. (2017). *Bald Eagle: General Information*. Retrieved May 11, 2017, from http://myfwc.com/wildlifehabitats/managed/baldeagle/information/.

Florida Fish and Wildlife Conservation Commission. (2018). *Index Nesting Beach Survey Totals (1989–2017)*. Retrieved October 19, 2018, from http://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/.

Florida Fish and Wildlife Conservation Commission. (2022). *Manatee Mortality Event Along The East Coast: 2020-ongoing*. Retrieved June 27, 2024, from https://myfwc.com/research/manatee/rescue-mortality-response/ume/.

Florida Fish and Wildlife Conservation Commission. (2023a). *American Alligator Alligator mississippiensis*. Retrieved August 29, 2023, from https://myfwc.com/wildlifehabitats/profiles/reptiles/alligator/.

Florida Fish and Wildlife Conservation Commission. (2023b). *Diamondback Terrapin Malaclemys terrapin*. Retrieved August 29, 2023, from https://myfwc.com/wildlifehabitats/profiles/reptiles/freshwater-turtles/diamondback-terrapin/.

Florida Fish and Wildlife Conservation Commission. (2023c). *Index Nesting Beach Survey Totals (1989-2022)*. Retrieved August 18, 2023, from https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/.

Florida Fish and Wildlife Conservation Commission. (2024). *Index Nesting Beach Survey Totals (1989-2023)*. Retrieved August 13, 2024, from https://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/.

Florida Museum of Natural History. (2017). *Gulf Sturgeon, Acipenser oxyrinchus desotoi*. Retrieved April 5, 2017, from https://www.flmnh.ufl.edu/fish/discover/species-profiles/acipenser-oxyrinchus-desotoi/.

Florida Museum of Natural History. (2018a). *Discover Fishes: Sawfish biology*. Retrieved June 23, 2023, from https://www.floridamuseum.ufl.edu/discover-fish/sawfish/biology/.

Florida Museum of Natural History. (2018b). *Pristis pristis*. Retrieved January 17, 2023, from.

Foderaro, L. W. (2015, July 21). *Group Petitions to Save a Prehistoric Fish From Modern Construction*. *New York Times.* Retrieved March 1, 2016, from http://www.nytimes.com/2015/07/22/nyregion/group-petitions-to-save-a-prehistoric-fish-from-modern-construction.html?_r=0.

Foley, A. M., B. A. Schroeder, R. Hardy, S. L. MacPherson, and M. Nicholas. (2014). Long-term behavior at foraging sites of adult female loggerhead sea turtles (*Caretta caretta*) from three Florida rookeries. *Marine Biology 161* (6): 1251–1262. DOI:10.1007/s00227-014-2415-9

Foley, A. M., B. A. Stacy, R. F. Hardy, C. P. Shea, K. E. Minch, and B. A. Schroeder. (2019). Characterizing watercraft-related mortality of sea turtles in Florida. *The Journal of Wildlife Management 83* (5): 1057–1072. DOI:10.1002/jwmg.21665

Foley, H., Z. Swaim, D. Waples, and A. Read. (2015). *Deep Divers and Satellite Tagging Projects in the Virginia Capes OPAREA - Cape Hatteras, NC: January 2014–December 2014*. Virginia Beach, VA: U.S. Fleet Forces Command.

Foley, H. J., R. C. Holt, R. E. Hardee, P. B. Nilsson, K. A. Jackson, A. J. Read, D. A. Pabst, and W. A. McLellan. (2011). Observations of a western North Atlantic right whale (*Eubalaena glacialis*) birth offshore of the protected southeast U.S. critical habitat. *Marine Mammal Science 27* (3): E234–E240. DOI:10.1111/j.1748-7692.2010.00452

Foley, H. J., K. Pacifici, R. W. Baird, D. L. Webster, Z. T. Swaim, and A. J. Read. (2021). Residency and movement patterns of Cuvier's beaked whales *Ziphius cavirostris* off Cape Hatteras, North Carolina, USA. *Marine Ecology Progress Series 660* 203–216. DOI:10.3354/meps13593

Folkow, L. P., E. S. Nordoy, and A. S. Blix. (2004). Distribution and diving behaviour of harp seals (*Pagophilus groenlandicus*) from the Greenland sea stock. *Polar Biology 27* 281–298.

Fonseca, M. and A. Malhotra. (2012). *Boat Wakes and Their Influence on Erosion in the Atlantic Intracoastal Waterway, North Carolina*. Beaufort, NC: National Oceanic and Atmospheric Administration, National Ocean Service, National Centers for Coastal Ocean Science.

Food and Agriculture Organization of the United Nations. (2005). *Review of the State of World Marine Fishery Resources*. Rome, Italy: Food and Agriculture Organization Fisheries Department, Fishery Resources Division, Marine Resources Service.

Food and Agriculture Organization of the United Nations. (2009). *The State of World Fisheries and Aquaculture*. Rome, Italy: Food and Agriculture Organization Fisheries Department, Fishery Resources Division, Marine Resources Service.

Food and Agriculture Organization of the United Nations. (2012). *Species Fact Sheets, Sphyrna lewini*. Retrieved from http://www.fao.org/fishery/species/2028/en.

Food and Agriculture Organization of the United Nations. (2013). *Report of the Fourth FAO Expert Advisory Panel for the Assessment of Proposals to Amend Appendices I and II of CITES Concerning Commercially-Exploited Aquatic Species*. Rome, Italy: Food and Agriculture Organization Fisheries Department, Fishery Resources Division, Marine Resources Service.

Ford, J. K. B., G. M. Ellis, D. R. Matkin, K. C. Balcomb, D. Briggs, and A. B. Morton. (2005). Killer whale attacks on minke whales: Prey capture and antipredator tactics. *Marine Mammal Science 21* (4): 603–618.

Fossette, S., S. Ferraroli, H. Tanaka, Y. Ropert-Coudert, N. Arai, K. Sato, Y. Naito, Y. Le Maho, and J. Georges. (2007). Dispersal and dive patterns in gravid leatherback turtles during the nesting season in French Guiana. *Marine Ecology Progress Series 338* 233–247.

Foster, A. M. and J. P. Clugston. (1997). Seasonal migration of Gulf sturgeon in the Suwannee River, Florida. *Transactions of the American Fisheries Society 126* 302–308.

Foster, N. L., C. B. Paris, J. T. Kool, I. B. Baums, J. R. Stevens, J. A. Sanchez, C. Bastidas, C. Agudelo, P. Bush, O. Day, R. Ferrari, P. Gonzalez, S. Gore, R. Guppy, M. A. McCartney, C. McCoy, J. Mendes, A. Srinivasan, S. Steiner, M. J. Vermeij, E. Weil, and P. J. Mumby. (2012). Connectivity of Caribbean coral populations: Complementary insights from empirical and modelled gene flow. *Molecular Ecology 21* (5): 1143–1157. DOI:10.1111/j.1365-294X.2012.05455

Foster, T. and J. P. Gilmour. (2016). Seeing red: Coral larvae are attracted to healthy-looking reefs. *Marine Ecology Progress Series 559* 65–71. DOI:10.3354/meps11902

Fox, A. G., E. S. Stowe, K. J. Dunton, and D. L. Peterson. (2018). Seasonal occurrence of Atlantic sturgeon (*Acipenser oxyrinchus oxyrinchus*) in the St. Johns River, Florida. *Fishery Bulletin 116* 219+.

Fox, A. G., E. S. Stowe, and D. L. Peterson. (2016). *Occurrence and Movements of Shortnose and Atlantic Sturgeon in the St. Johns River, Florida* (Final Report to the United States Army Corps of Engineers and the United States Navy). Athens, GA: University of Georgia.

Fox, D. A., J. E. Hightower, and F. M. Parauka. (2000). Gulf sturgeon spawning migration and habitat in the Choctawhatchee River system, Alabama-Florida. *Transactions of the American Fisheries Society 129* 811–826.

Fox, D. A., J. E. Hightower, and F. M. Parauka. (2002). Estuarine and nearshore marine habitat use by Gulf sturgeon from the Choctawhatchee River System, Florida. *American Fisheries Society Symposium 28* 111–126.

Fraser, P. J. (1987). Atlantic salmon, *Salmo salar L*., feed in Scottish coastal waters. *Aquaculture Research 18* (3): 243–247.

Freiwald, A., A. Rogers, J. Hall-Spencer, J. M. Guinotte, A. J. Davies, C. Yesson, C. S. Martin, and L. V. Weatherdon. (2021). *Global distribution of cold-water corals (version 5.1)*. Retrieved from https://data.unep-wcmc.org/pdfs/3/WCMC_001_Global_Distribution_of_Cold-water_Corals.pdf?1617121038.

Frick, M. G., C. A. Quinn, and C. K. Slay. (1999). *Dermochelys coriacea* (leatherback sea turtle), *Lepidochelys kempii* (Kemp's ridley sea turtle), and *Caretta caretta* (loggerhead sea turtle): pelagic feeding. *Herpetological Review 30* (3): 165.

Fricke, R., W. Eschmeyer, and J. D. Fong. (2023). *Genera/Species by Family/Subfamily in: Eschmeyer's Catalog of Fishes*. Retrieved January 24, 2023, from https://researcharchive.calacademy.org/research/ichthyology/catalog/SpeciesByFamily.asp.

Friedlaender, A. S., E. L. Hazen, D. P. Nowacek, P. N. Halpin, C. Ware, M. T. Weinrich, T. Hurst, and D. Wiley. (2009). Diel changes in humpback whale *Megaptera novaeangliae* feeding behavior in response to sand lance *Ammodytes* spp. behavior and distribution. *Marine Ecology Progress Series 395* 91–100. DOI:10.3354/meps08003

Frost, M. T., S. Dye, B. Stoker, P. Buckley, and J. Baxter. (2017). *Marine Climate Change Impacts: 10 Years' Experience of Science to Policy Reporting. Summary Report*. London, United Kingdom: Marine Climate Change Impacts Partnership.

Fuentes, M. M. P. B., D. A. Pike, A. Dimatteo, and B. P. Wallace. (2013). Resilience of marine turtle regional management units to climate change. *Global Change Biology 19* (5): 1399–1406.

Fukuoka, T., M. Yamane, C. Kinoshita, T. Narazaki, G. J. Marshall, K. J. Abernathy, N. Miyazaki, and K. Sato. (2016). The feeding habit of sea turtles influences their reaction to artificial marine debris. *Scientific Reports 6* 28015. DOI:10.1038/srep28015

Fullard, K. J., G. Early, M. P. Heide-Jorgensen, D. Bloch, A. Rosing-Asvid, and W. Amos. (2000). Population structure of long-finned pilot whales in the North Atlantic: A correlation with sea surface temperature? *Molecular Ecology 9* 949–958.

Fulling, G. L. and D. Fertl. (2003). *Kogia distribution in the northern Gulf of Mexico*. Presented at the Workshop on the Biology of *Kogia*. Greensboro, NC.

Fulling, G. L., K. D. Mullin, and C. W. Hubard. (2003). Abundance and distribution of cetaceans in outer continental shelf waters of the U.S. Gulf of Mexico. *Fishery Bulletin 101* 923–932.

Furness, R. W. (2003). Impacts of fisheries on seabird communities. *Scientia Marina 67* 33–45.

Gallagher, A. J., E. S. Orbesen, N. Hammerschlag, and J. E. Serafy. (2014). Vulnerability of oceanic sharks as pelagic longline bycatch. *Global Ecology and Conservation 1* 50–59. DOI:https://doi.org/10.1016/j.gecco.2014.06.003

Gallaway, B. J., C. W. Caillouet, P. T. Plotkin, W. J. Gazey, J. G. Cole, and S. W. Raborn. (2013). *Kemp's Ridley Stock Assessment Project* (Final Report to the Gulf States Marine Fisheries Commission). Ocean Springs, MS: Gulf States Marine Fisheries Commission.

Gallaway, B. J., W. J. Gazey, T. Wibbels, E. Bevan, D. J. Shaver, and J. George. (2016). Evaluation of the status of the Kemp's ridley sea turtle after the 2010 Deepwater Horizon oil spill. *Gulf of Mexico Science 33* (2): 192–205. DOI:10.18785/goms.3302.06

Gannier, A. and E. Praca. (2007). SST fronts and the summer sperm whale distribution in the north-west Mediterranean Sea. *Journal of the Marine Biological Association of the United Kingdom 87* (01): 187. DOI:10.1017/s0025315407054689

Gannier, A. and K. L. West. (2005). Distribution of the rough-toothed dolphin (*Steno bredanensis*) around the Windward Islands (French Polynesia). *Pacific Science 59* 17–24.

Gannon, D. P., A. J. Read, J. E. Craddock, K. M. Fristrup, and J. R. Nicolas. (1997). Feeding ecology of long-finned pilot whales *Globicephala melas* in the western North Atlantic. *Marine Ecology Progress Series 148* 1–10.

Gannon, J. G., K. M. Scolardi, J. E. Reynolds, III, J. K. Koelsch, and T. J. Kessenich. (2007). Habitat selection by manatees in Sarasota Bay, Florida. *Marine Mammal Science 23* (1): 133–143. DOI:DOI:10.1111/j.1748-7692.2006.00096

García-Sais, J. R., J. Sabater-Clavell, R. Esteves, and M. Carlo. (2012). *Fishery independent survey of commercially exploited fish and shellfish populations from mesophotic reefs within the Puerto Rican EEZ*. San Juan, PR: Caribbean Fisheries Management Council.

Gardner, B., L. A. Garner, D. T. Cobb, and C. E. Moorman. (2016). Factors affecting occupancy and abundance of American alligators at the northern extent of their range. *Journal of Herpetology 50* (4): 541–547.

Gargan, P. G., G. Forde, N. Hazon, D. J. F. Russell, and C. D. Todd. (2012). Evidence for sea lice-induced marine mortality of Atlantic salmon (*Salmo salar*) in western Ireland from experimental releases of ranched smolts treated with emamectin benzoate. *Canadian Journal of Fisheries and Aquatic Sciences 69* (2): 343–353. DOI:DOI:10.1139/f2011-155

Garrison, E. G., J. Ortega-Ortiz, G. Rappucci, L. Aichinger Dias, K. Mullin, and J. Litz. (2023). *Gulf of Mexico Marine Assessment Program for Protected Species (GOMMAPPS): Marine Mammals. Volume 3: Appendix D: Gulf of Mexico Sea Turtle Spatial Density Models* (Report No.: OCS Study BOEM

2023-042). New Orleans, LA: U.S. Department of the Interior, Bureau of Ocean Energy Management.

Garrison, L. P., J. Ortega-Ortiz, and G. Rappucci. (2020). *Abundance of marine mammals in the waters of the U.S. Gulf of Mexico in the summer of 2017 and 2018* (PRBD Contribution # PRBD-2020-07). Southeast Fisheries Science Center, Protected Resources and Biodiversity Division.

Gaskin, D. E. (1977). *Harbour porpoise, Phocoena phocoena (L.), in the western approaches to the Bay of Fundy 1969–75* (Report of the International Whaling Commission). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Gaskin, D. E. (1992). Status of the harbour porpoise, *Phocoena phocoena*, in Canada. *Canadian Field-Naturalist 106* (1): 36–54.

Gaskins, L. C., A. B. Paxton, and B. R. Silliman. (2020). Megafauna in Salt Marshes. *Frontiers in Marine Science 7*. DOI:10.3389/fmars.2020.561476

Geburzi, J. C. and M. L. McCarthy. (2018). *How do they do it?–Understanding the success of marine invasive species*. Presented at the YOUMARES 8–Oceans Across Boundaries: Learning from each other: Proceedings of the 2017 conference for YOUng MARine RESearchers in Kiel, Germany. Kiel, Germany.

Gerstein, E. R., J. E. Blue, and S. E. Forysthe. (2005). The acoustics of vessel collisions with marine mammals. *Proceedings of OCEANS 2005 MTS/IEEE 2* 1190–1197. DOI:10.1109/OCEANS.2005.1639917

Gil-Agudelo, D. L., G. W. Smith, and E. Weil. (2006). The white band disease type II pathogen in Puerto Rico. *Revista de Biología Tropical 54 (Supplement 3)* 59–67.

Gilbert, J. R. and N. Guldager. (1998). *Status of Harbor and Gray Seal Populations in Northern New England*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Gill, F. B. (1995). *Ornithology* (2nd ed.). New York, NY: W.H. Freeman and Company.

Gilliam, D. S. and B. K. Walker. (2011). *Benthic Habitat Characterization for the South Florida Ocean Measurement Facility (SFOMF) - Protected Stony Coral Species Assessment*. Fort Lauderdale, FL: Nova Southeastern University Oceanographic Center.

Ginsburg, R. N. and E. A. Shinn. (1964). Distribution of the reef building community in Florida and the Bahamas. *Bulletin of the American Association of Petroleum Geologists 48* 527.

Gitschlag, G. R. (1996). Migration and diving behavior of Kemp's ridley (Garman) sea turtles along the U.S. southeastern Atlantic coast. *Journal of Experimental Marine Biology and Ecology 205* 115–135.

Glandon, H. L. and T. J. Miller. (2016). No effect of high pCO_2 on juvenile blue crab, *Callinectes sapidus*, growth and consumption despite positive responses to concurrent warming. *ICES Journal of Marine Science 74* (4): 1201–1209. DOI:10.1093/icesjms/fsw171

Glazer, R. A. and J. A. Kidney. (2004). Habitat associations of adult queen conch (*Strombus gigas* L.) in an unfished Florida Keys back reef: Applications to essential fish habitat. *Bulletin of Marine Science 75* (2): 205-224.

Gobler, C. J. (2020). Climate change and harmful algal blooms: Insights and perspective. *Harmful Algae 91* 101731. DOI:https://doi.org/10.1016/j.hal.2019.101731

Gochfeld, M. (1983). The roseate tern: World distribution and status of a threatened species. *Biological Conservation 25* 103–125.

Godley, B. J., D. R. Thompson, S. Waldron, and R. W. Furness. (1998). The trophic status of marine turtles as determined by stable isotope analysis. *Marine Ecology Progress Series 166* 277–284.

Goetz, J., J. Hardesty-Norris, and J. Wheeler. (2012). *Conservation Action Plan for the Black-capped Petrel (Pterodroma hasitata)*. Washington, DC: International Black-capped Petrel Conservation Group.

Goldbogen, J. A., J. Calambokidis, R. E. Shadwick, E. M. Oleson, M. A. McDonald, and J. A. Hildebrand. (2006). Kinematics of foraging dives and lunge-feeding in fin whales. *The Journal of Experimental Biology 209* (Pt 7): 1231–1244. DOI:10.1242/jeb.02135

Goldbogen, J. A., E. L. Hazen, A. S. Friedlaender, J. Calambokidis, S. L. DeRuiter, A. K. Stimpert, B. L. Southall, and D. Costa. (2015). Prey density and distribution drive the three-dimensional foraging strategies of the largest filter feeder. *Functional Ecology 29* (7): 951–961. DOI:10.1111/1365-2435.12395

Gonçalves Neto, A., J. A. Langan, and J. B. Palter. (2021). Changes in the Gulf Stream preceded rapid warming of the Northwest Atlantic Shelf. *Communications Earth & Environment 2* (1): 74. DOI:10.1038/s43247-021-00143-5

Gonzalez-Terrazas, T. P., J. C. Koblitz, T. H. Fleming, R. A. Medellín, E. K. Kalko, H. U. Schnitzler, and M. Tschapka. (2016). How nectar-feeding bats localize their food: Echolocation behavior of *leptonycteris yerbabuenae* approaching cactus flowers. *PLoS ONE 11* (9): e0163492. DOI:10.1371/journal.pone.0163492

Gonzalez, L. A. (2011). State of the Bay: Water and Sediment Quality. In *A Characterization of the Galveston Bay Ecosystem* (Third ed., pp. 1–45). Austin, TX: Texas Commission on Environmental Quality, Galveston Bay Estuary Program.

Goold, J. C. (2000). A diel pattern in vocal activity of short-beaked common dolphins, *Delphinus delphis*. *Marine Mammal Science 16* (1): 240–244.

Goreau, T. F. (1959). The ecology of Jamaican coral reefs I. Species composition and zonation. *Ecology 40* (1): 67–90.

Goreau, T. F. and J. W. Wells. (1967). The shallow-water Scleractinia of Jamaica: Revised list of species and their vertical distribution range. *Bulletin of Marine Science 17* (2): 442–453.

Gorham, J., D. Clark, M. Bresette, D. Bagley, C. Keske, S. Traxler, B. Witherington, B. Shamblin, and C. Nairn. (2014). Characterization of a subtropical Hawksbill sea turtle (Eretmocheyles imbricata) assemblage utilizing shallow water natural and artificial habitats in the Florida Keys. *PLoS ONE 9* 1-–16. DOI:10.1371/journal.pone.0114171

Gowan, T. A., H. H. Edwards, A. M. Krzystan, J. Martin, and J. A. Hostetler. (2023). *2021–2022 Statewide Abundance Estimates for the Florida Manatee*. St. Petersburg, FL: Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute.

Gowan, T. A., J. G. Ortega-Ortiz, J. A. Hostetler, P. K. Hamilton, A. R. Knowlton, K. A. Jackson, R. C. George, C. R. Taylor, and P. J. Naessig. (2019). Temporal and demographic variation in partial migration of the North Atlantic right whale. *Scientific Reports 9* (1): 1-11. DOI:10.1038/s41598-018-36723-3

Gowan, T. A., M. D. Tringali, J. A. Hostetler, J. Martin, L. I. Ward-Geiger, and J. M. Johnson. (2021). A hidden Markov model for estimating age-specific survival when age and size are uncertain. *Ecology 102* (8): e03426. DOI:https://doi.org/10.1002/ecy.3426

Gowans, S. (2009). Bottlenose whales *Hyperoodon ampullatus* and *H. planifrons*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 129–131). Cambridge, MA: Academic Press.

Gowans, S. and H. Whitehead. (1995). Distribution and habitat partitioning by small odontocetes in the Gully, a submarine canyon on the Scotian Shelf. *Canadian Journal of Zoology 73* 1599–1608.

Gower, J. and S. King. (2020). The distribution of pelagic Sargassum observed with OLCI. *International Journal of Remote Sensing 41* (15): 5669–5679. DOI:10.1080/01431161.2019.1658240

Gower, J., E. Young, and S. King. (2013). Satellite images suggest a new *Sargassum* source region in 2011. *Remote Sensing Letters 4* (8): 764–773. DOI:10.1080/2150704X.2013.796433

Grant, G. S. and D. Ferrell. (1993). Leatherback turtle, *Dermochelys coriacea* (Reptilia: *Dermochelidae*): Notes on near-shore feeding behavior and association with cobia. *Brimleyana 19* 77–81.

Gratto-Trevor, C., D. Amirault-Langlais, D. Catlin, F. Cuthbert, J. Fraser, S. Maddock, E. Roche, and F. Shaffer. (2012). Connectivity in piping plovers: Do breeding populations have distinct winter distributions? *The Journal of Wildlife Management 76* (2): 348–355. DOI:doi:10.1002/jwmg.261

Gredzens, C. and D. J. Shaver. (2020). Satellite Tracking Can Inform Population-Level Dispersal to Foraging Grounds of Post-nesting Kemp's Ridley Sea Turtles. *Frontiers in Marine Science 7* 1–15. DOI:10.3389/fmars.2020.00559

Green, G. A., J. J. Brueggeman, R. A. Grotefendt, C. E. Bowlby, M. L. Bonnell, and K. C. Balcomb, III. (1992). *Cetacean Distribution and Abundance off Oregon and Washington, 1989–1990*. Los Angeles, CA: U.S. Department of the Interior, Minerals Management Service.

Green, T. W., D. H. Slone, E. D. Swain, M. S. Cherkiss, M. Lohmann, F. J. Mazzotti, and K. G. Rice. (2014). Evaluating effects of Everglades restoration on American crocodile populations in South Florida using a spatially-explicit, stage-based population model. *Wetlands 34* (1): 213–224.

Griffin, D. B., S. R. Murphy, M. G. Frick, A. C. Broderick, J. W. Coker, M. S. Coyne, M. G. Dodd, M. H. Godfrey, B. J. Godley, and L. A. Hawkes. (2013). Foraging habitats and migration corridors utilized by a recovering subpopulation of adult female loggerhead sea turtles: Implications for conservation. *Marine Biology 160* (12): 3071–3086.

Griffin, L., C. Griffin, J. Finn, R. Prescott, M. Faherty, B. Still, and A. Danylchuk. (2019). Warming seas increase cold-stunning events for Kemp's ridley sea turtles in the northwest Atlantic. *PLoS ONE 14* 1–15. DOI:10.1371/journal.pone.0211503

Griffin, R. B. and N. J. Griffin. (2003). Distribution, habitat partitioning, and abundance of Atlantic spotted dolphins, bottlenose dolphins, and loggerhead sea turtles on the eastern Gulf of Mexico continental shelf. *Gulf of Mexico Science 1* 23–34.

Griffith, A. W. and C. J. Gobler. (2020). Harmful algal blooms: A climate change co-stressor in marine and freshwater ecosystems. *Harmful Algae 91* 101590. DOI:https://doi.org/10.1016/j.hal.2019.03.008

Grober-Dunsmore, R., V. Bonito, and T. K. Frazer. (2006). Potential inhibitors to recovery of *Acropora palmata* populations in St. John, U.S. Virgin Islands. *Marine Ecology Progress Series 321* 123–132. DOI:10.3354/meps321123

Gubbins, C. (2002). Association patterns of resident bottlenose dolphins (*Tursiops truncatus*) in a South Carolina estuary. *Aquatic Mammals 28* (11): 24–31.

Gubbins, C., M. Caldwell, S. G. Barco, K. Rittmaster, N. Bowles, and V. Thayer. (2003). Abundance and sighting patterns of bottlenose dolphins (*Tursiops truncatus*) at four northwest Atlantic coastal sites. *Journal of Cetacean Research and Management 5* (2): 141–147.

Guertin, S. (2019). *Marine Debris: Impacts on Ecosystems and Species*. Washington, DC: U.S. Department of the Interior, Office of Congressional and Legislative Affairs.

Guins, M., D. Rees, and A. Lay. (2023). *Pinniped Time-lapse Camera Surveys Southern Chesapeake Bay and Eastern Shore, Virginia: 2019-2022. Final Report*. Norfolk, VA: U.S. Fleet Forces Command.

Gulick, A. G., K. A. Ewen, C. G. Pollock, and Z. M. Hillis-Starr. (2022). Trends in abundance and reproductive success of the hawksbill turtle nesting population at Buck Island Reef National Monument, St. Croix, US Virgin Islands. *Endangered Species Research 48* 191–198. DOI:10.3354/esr01199

Gulland, F. M., J. Baker, M. Howe, E. LaBrecque, L. Leach, S. E. Moore, R. R. Reeves, and P. O. Thomas. (2022). A Review of Climate Change Effects on Marine Mammals in United States Waters: Past Predictions, Observed Impacts, Current Research and Conservation Imperatives. *Climate Change Ecology* 100054.

Hackney, A. D. (2010). *Conservation biology of the diamondback terrapin in North America: Policy status, nest predation, and managing island populations*. Clemson, SC: Clemson University.

Haelters, J., F. Kerckhof, T. Jauniaux, and S. Degraer. (2012). The grey seal (*Halichoerus grypus*) as a predator of harbor porpoises (*Phocoena phocoena*)? *Aquatic Mammals 38* (4): 343–353. DOI:10.1578/AM.38.4.2012.343

Hage, S. R., T. Jiang, S. W. Berquist, J. Feng, and W. Metzner. (2013). Ambient noise induces independent shifts in call frequency and amplitude within the Lombard effect in echolocating bats. *Proceedings of the National Academy of Sciences 110* (10): 4063–4068. DOI:10.1073/pnas.1211533110

Hager, C. (2015). *Telemetry tracking of Atlantic sturgeon in the lower Chesapeake Bay: Final Report for 2014*. Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Hager, C., J. Kahn, C. Watterson, J. Russo, and K. Hartman. (2014). Evidence of Atlantic Sturgeon Spawning in the New York River System. *Transactions of the American Fisheries Society 143* (5): 1217–1219. DOI:10.1080/00028487.2014.925971

Hager, C. and N. Mathias. (2018). *Operation of the Navy's Telemetry Array in the Lower Chesapeake Bay: Annual Progress Report for 2017. Final Report.* Prepared for U.S. Fleet Forces Command and Commander, Navy Region Mid-Atlantic. Submitted to Naval Facilities Engineering Command Atlantic, Norfolk, Virginia, under Contract No. N62470-15-D-8006, Task Order 31, issued to HDR Inc., Virginia Beach, Virginia.

Hager, C., C. Watterson, and J. Kahn. (2016). *Migration patterns of the Pamunkey River spawning stock [Presentation]*. Presented at the Atlantic and Shortnose Sturgeon 2016 Workshop. Sheperdstown, WV.

Haig, S. M. and E. Elliott-Smith. (2004). *Piping Plover*. *The Birds of North America.* Retrieved 3, from http://bna.birds.cornell.edu/bna/species.

Hain, J. H. W., G. R. Carter, S. C. Kraus, C. A. Mayo, and H. E. Winn. (1982). Feeding behavior of the humpback whale, *Megaptera novaeangliae*, in the western North Atlantic [North Eastern States (USA)]. *Fishery Bulletin 80* (2): 259-268.

Hain, J. H. W., R. K. Edel, H. E. Hays, S. K. Katona, and J. D. Roanowics. (1981). *General distribution of cetaceans in the continental shelf waters of the northeastern United States* (A characterization of marine mammals and turtles in the mid-and north-Atlantic areas of the U.S. outer continental shelf). Washington, DC: Bureau of Land Management.

Hain, J. H. W., M. J. Ratnaswamy, R. D. Kenney, and H. E. Winn. (1992). The fin whale, *Balaenoptera physalus*, in waters of the northeastern United States continental shelf. *Reports of the International Whaling Commission 42* 653–670.

Hall, A. and D. Thompson. (2009). Gray seal, *Halichoerus grypus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 500–503). Cambridge, MA: Academic Press.

Halley, R., G. P. Dennis, D. Weaver, and F. C. Coleman. (2005). *National Oceanic and Atmospheric Administration Coral Reef Conservation Grants 2002: Habitat Characterization of Pulley Ridge and the Florida Middle Grounds. Part II: Characterization of the Pulley Ridge Coral and Fish Fauna* (Final Report to The National Oceanic and Atmospheric Administration Coral Reef Conservation Grant Program). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Halpern, B. S., M. Frazier, J. Afflerbach, J. S. Lowndes, F. Micheli, C. O'Hara, C. Scarborough, and K. A. Selkoe. (2019). Recent pace of change in human impact on the world's ocean. *Scientific Reports 9* (1): 11609. DOI:10.1038/s41598-019-47201-9

Halpern, B. S., M. Frazier, J. Potapenko, K. S. Casey, K. Koenig, C. Longo, J. S. Lowndes, R. C. Rockwood, E. R. Selig, K. A. Selkoe, and S. Walbridge. (2015). Spatial and temporal changes in cumulative human impacts on the world's ocean. *Nature Communications 6* (1): 7615. DOI:10.1038/ncomms8615

Halpern, B. S., K. L. McLeod, A. A. Rosenberg, and L. B. Crowder. (2008). Managing for cumulative impacts in ecosystem-based management through ocean zoning. *Ocean & Coastal Management 51* (3): 203–211. DOI:10.1016/j.ocecoaman.2007.08.002

Halpin, P. N., A. J. Read, E. Fujioka, B. D. Best, B. Donnelly, L. J. Hazen, C. Kot, K. Urian, E. LaBrecque, and A. Dimatteo. (2009). OBIS-SEAMAP: The world data center for marine mammal, sea bird, and sea turtle distributions. *Oceanography 22* (2): 104-115.

Hamazaki, T. (2002). Spatiotemporal prediction models of cetacean habitats in the mid-western North Atlantic Ocean (from Cape Hatteras, North Carolina, U.S.A. to Nova Scotia, Canada). *Marine Mammal Science 18* (4): 920–939.

Hamilton, S. and G. B. Baker. (2019). Technical mitigation to reduce marine mammal bycatch and entanglement in commercial fishing gear: lessons learnt and future directions. *Reviews in Fish Biology and Fisheries 29* (2): 223-247. DOI:10.1007/s11160-019-09550-6

Hammill, M. O., W. D. Bowen, and B. Sjare. (2010). Status of harbor seals (*Phoca vitulina*) in Atlantic Canada. *North Atlantic Marine Mammal Commission Scientific Publications 8* 175–190.

Hammill, M. O., C. E. den Heyer, and W. D. Bowen. (2014a). *Grey Seal Population Trends in Canadian Waters, 1960-2014*. Ottawa, Canada: Fisheries and Oceans Canada.

Hammill, M. O. and J. F. Gosselin. (1995). Grey seal (*Halichoerus grypus*) from the Northwest Atlantic: Female reproductive rates, age at first birth, and age of maturity in males. *Canadian Journal of Fisheries and Aquatic Sciences 52* 2757–2761.

Hammill, M. O., K. M. Lydersen, K. M. Kovacs, and B. Sjare. (1997). Estimated fish consumption by hooded seals (*Cystophora cristata*) in the Gulf of St. Lawrence. *Journal of Northwest Atlantic Fishery Science 22* 249–258.

Hammill, M. O. and G. B. Stenson. (2006). *Abundance of Northwest Atlantic hooded seals (1960 — 2005)*. Ottawa, Canada: Fisheries and Oceans Canada.

Hammill, M. O., G. B. Stenson, A. Mosnier, and T. Doniol-Valcroze. (2021). *Trends in abundance of harp seals, Pagophilus groenlandicus, in the Northwest Atlantic, 1952–2019*. Ottawa, Canada: Fisheries and Oceans Canada, Canadian Science Advisory Secretariat Research Document 2021/006.

Hammill, M. O., G. B. Stenson, R. A. Myers, and W. T. Stobo. (1998). Pup production and population trends of the grey seal (*Halichoerus grypus*) in the Gulf of St. Lawrence. *Canadian Journal of Fisheries and Aquatic Sciences 55* 423–430.

Hammill, M. O., G. B. Stenson, D. P. Swain, and H. P. Benoit. (2014b). Feeding by grey seals on endangered stocks of Atlantic cod and white hake. *ICES Journal of Marine Science 71* (6): 1332–1341. DOI:10.1093/icesjms/fsu123

Han, G., E. Colbourne, P. Pepin, and Y. Xie. (2015). Statistical Projections of Ocean Climate Indices off Newfoundland and Labrador. *Atmosphere-Ocean 53* (5): 556–570. DOI:10.1080/07055900.2015.1047732

Han, G., Z. Ma, Z. Long, W. Perrie, and J. Chassé. (2019). Climate Change on Newfoundland and Labrador Shelves: Results From a Regional Downscaled Ocean and Sea-Ice Model Under an A1B Forcing Scenario 2011–2069. *Atmosphere-Ocean 57* (1): 3–17. DOI:10.1080/07055900.2017.1417110

Handley, C. O., Jr. (1966). A synopsis of the genus *Kogia* (pygmy sperm whales). In K. S. Norris (Ed.), *Whales, Dolphins, and Porpoises* (pp. 62–69). Berkeley, CA: University of California Press.

Haney, J. C. (1987). Aspects of the Pelagic Ecology and Behavior of the Black-capped Petrel (*Pterodroma hasitata*). *The Wilson Bulletin 99* (2): 153–168.

Hansen, L. J., K. D. Mullin, T. A. Jefferson, and G. P. Scott. (1996). *Visual Surveys Aboard Ships and Aircraft* (Distribution and Abundance of Marine Mammals in the Northcentral and Western Gulf of Mexico). New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Hansen, L. J., K. D. Mullin, and C. L. Roden. (1995). *Estimates of Cetacean Abundance in the Northern Gulf of Mexico from Vessel Surveys*. Miami, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Hansen, L. P. and M. L. Windsor. (2006). Interactions between aquaculture and wild stocks of Atlantic salmon and other diadromous fish species: Science and management, challenges and solutions. *ICES Journal of Marine Science 63* (7): 1159–1161. DOI:10.1016/J.ICEJMS.2006.05.003

Harris, D. E., B. Lelli, and G. Jakush. (2002). Harp seal records from the southern Gulf of Maine: 1997–2001. *Northeastern Naturalist 9* (3): 331–340.

Harris, D. E., B. Lelli, G. Jakush, and G. Early. (2001). Hooded seal (*Cystophora cristata*) records from the southern Gulf of Maine. *Northeastern Naturalist 8* 427–434.

Harris, J. E., D. C. Parkyn, and D. J. Murie. (2005). Distribution of Gulf of Mexico sturgeon in relation to benthic invertebrate prey resources and environmental parameters in the Suwannee River estuary, Florida. *Transactions of the American Fisheries Society 134* 975–990.

Harrison, J., M. C. Ferguson, L. New, J. Cleary, C. Curtice, S. DeLand, E. Fujioka, P. N. Halpin, R. B. T. Moore, and S. M. Van Parijs. (2023). Biologically Important Areas II for cetaceans within U.S. and adjacent waters - Updates and the application of a new scoring system. *Frontiers in Marine Science 10*. DOI:doi.org/10.3389/fmars.2023.1081893

Harrison, P. (1983). *Seabirds, an Identification Guide*. Boston, MA: Houghton Mifflin Company.

Hart, K. M., A. R. Iverson, I. Fujisaki, M. M. Lamont, D. Bucklin, and D. J. Shaver. (2018). Sympatry or syntopy? Investigating drivers of distribution and co-occurrence for two imperiled sea turtle species in Gulf of Mexico neritic waters. *Ecology and Evolution 8* 12656–12669. DOI:10.1002/ece3.4691

Hart, K. M. and D. S. Lee. (2006). The Diamondback Terrapin: The Biology, Ecology, Cultural History, and Conservation Status of an Obligate Estuarine Turtle. In R. Greenberg, J. E. Maldonado, S. Droege, & M. V. McDonald (Eds.), *Terrestrial Vertebrates of Tidal Marshes: Evolution, Ecology, and Conservation*. Camarillo, CA: Cooper Ornithological Society.

Hart, K. M., P. J. Schofield, and D. R. Gregoire. (2012). Experimentally derived salinity tolerance of hatchling Burmese pythons (*Python molurus bivittatus*) from the Everglades, Florida (USA). *Journal of Experimental Marine Biology and Ecology 413* 56–59.

Hart, K. M., C. F. White, A. R. Iverson, and N. Whitney. (2016). Trading shallow safety for deep sleep: Juvenile green turtles select deeper resting sites as they grow. *Endangered Species Research 31* 61–73.

Hartman, D. S. (1979). *Ecology and Behavior of the Manatee (Trichechus manatus) in Florida* (Special Publication). Pittsburgh, PA: American Society of Mammalogists.

Hatase, H., K. Sato, M. Yamaguchi, K. Takahashi, and K. Tsukamoto. (2006). Individual variation in feeding habitat use by adult female green sea turtles (*Chelonia mydas*): Are they obligately neritic herbivores? *Oecologia 149* 52–64. DOI:10.1007/s00442-006-0431-2

Hatch, S. K., E. E. Connelly, T. J. Divoll, I. J. Stenhouse, and K. A. Williams. (2013). Offshore observations of Eastern red bats (*Lasiurus borealis*) in the mid-Atlantic United States using multiple survey methods. *PLoS ONE 8* (12): e83803. DOI:10.1371/journal.pone.0083803

Haugland, M., J. C. Holst, M. Holm, and L. P. Hansen. (2006). Feeding of Atlantic salmon (*Salmo salar* L.) post-smolts in the Northeast Atlantic. *ICES Journal of Marine Science 63* 1488–1500.

Hauksson, E. and V. Bogason. (1997). Comparative feeding of grey (*Halichoerus grypus*) and common seals (*Phoca vitulina*) in coastal waters of Iceland, with a note on the diet of hooded (*Cystophora cristata*) and harp seals (*Phoca groenlandica*). *Journal of Northwest Atlantic Fishery Science 22* 125–135.

Hawkes, L. A., A. C. Broderick, M. S. Coyne, M. H. Godfrey, L.-F. Lopez-Jurado, P. Lopez-Suarez, S. E. Merino, N. Varo-Cruz, and B. J. Godley. (2006). Phenotypically linked dichotomy in sea turtle foraging requires multiple conservation approaches. *Current Biology 16* 990–995.

Hayes, S. A., E. Josephson, K. Maze-Foley, and P. E. Rosel (Eds.). (2019). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments - 2018*. Woods Hole, MA: National Marine Fisheries Service Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-NE-258.

Hayes, S. A., E. Josephson, K. Maze-Foley, P. E. Rosel, B. Byrd, S. Chavez-Rosales, T. V. Cole, L. P. Garrison, J. Hatch, and A. Henry (Eds.). (2020). *U.S. Atlantic and Gulf of Mexico marine mammal stock assessments-2019*. Woods Hole, MA: National Marine Fisheries Service Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-NE-264.

Hayes, S. A., E. Josephson, K. Maze-Foley, P. E. Rosel, B. Byrd, S. Chavez-Rosales, T. V. N. Cole, L. Engleby, L. P. Garrison, J. Hatch, A. Henry, S. C. Horstman, J. Litz, M. C. Lyssikatos, K. D. Mullin, C. Orphanides, R. M. Pace, D. L. Paka, M. Soldevilla, and F. W. Wenzel. (2018). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments – 2017* (NOAA Technical Memorandum NMFS-NE-245). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Hayes, S. A., E. Josephson, K. Maze-Foley, P. E. Rosel, J. McCordic, and J. Wallace (Eds.). (2023). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2022*. Woods Hole, MA: National Marine Fisheries Service Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-NE-304.

Hayes, S. H., E. Josephson, K. Maze-Foley, P. E. Rosel, and J. Turek (Eds.). (2021). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments 2020*. NOAA Technical Memorandum NMFS-NE-271. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Hayes, S. H., E. Josephson, K. Maze-Foley, P. E. Rosel, and J. Wallace (Eds.). (2022). *U.S. Atlantic and Gulf of Mexico marine mammal stock assessments 2021*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center. NOAA Technical Memorandum NMFS-NE-288.

Hays, G. C., J. D. R. Houghton, C. Isaacs, R. S. King, C. Lloyd, and P. Lovell. (2004). First records of oceanic dive profiles for leatherback turtles, *Dermochelys coriacea*, indicate behavioural plasticity associated with long-distance migration. *Animal Behaviour 67* 733–743. DOI:10.1016/j.anbehav.2003.08.011

Hazen, E. L., A. Friedlaender, M. A. Thompson, C. R. Ware, M. T. Weinrich, P. N. Halpin, and D. N. Wiley. (2009). Fine-scale prey aggregations and foraging ecology of humpback whales *Megaptera novaeangliae*. *Marine Ecology Progress Series 395* 75–89. DOI:10.3354/meps08108

HDR Environmental Operations and Construction Inc. (2013). *Benthic habitat characterization of Naval Air Station, Key West, Florida*. Norfolk, VA: Naval Facilities Engineering Command Atlantic.

Hedger, R. D., D. Hatin, J. J. Dodson, F. Martin, D. Fournier, F. Caron, and F. G. Whoriskey. (2009). Migration and swimming depth of Atlantic salmon kelts *Salmo salar* in coastal zone and marine habitats. *Marine Ecology Progress Series 392* 179–192. DOI:10.3354/meps08227

Heide-Jorgensen, M. P., D. Bloch, E. Stefansson, B. Mikkelsen, L. H. Ofstad, and R. Dietz. (2002). Diving behavior of long-finned pilot whales *Globicephala melas* around the Faroe Islands. *Wildlife Biology 8* (4): 307–313.

Heileman, S. and R. Mahon. (2008). *XV-49 Caribbean Sea: TAG: LME #12* (The UNEP Large Marine Ecosystem Report: A Perspective on Changing Conditions in LMEs of the World's Regional Seas). Nairobi, Kenya: United Nations Environmental Programme.

Heileman, S. and N. Rabalais. (2008). *XV-50 Gulf of Mexico: TAG: LME #5* (The UNEP Large Marine Ecosystem Report: A Perspective on Changing Conditions in LMEs of the World's Regional Seas). Nairobi, Kenya: United Nations Environmental Programme.

Heise, R. J., W. T. Slack, S. T. Ross, and M. A. Dugo. (2004). Spawning and associated movement patterns of Gulf sturgeon in the Pascagoula River drainage, Mississippi. *Transactions of the American Fisheries Society 133* 221–230.

Heithaus, M. R. and L. M. Dill. (2009). Feeding strategies and tactics. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1100–1103). Cambridge, MA: Academic Press.

Henry, A. G., M. Garron, D. Morin, A. Smith, A. Reid, W. Ledwell, and T. V. N. Cole. (2021). *Serious injury and mortality determinations for baleen whale stocks along the Gulf of Mexico, United States East Coast, and Atlantic Canadian Provinces, 2014-2018* (Northeast Fisheries Science Center Reference Document 21-07). National Marine Fisheries Service, Northeast Fisheries Science Center.

Herbert-Read, J. E., A. Thornton, D. J. Amon, S. N. R. Birchenough, I. M. Côté, M. P. Dias, B. J. Godley, S. A. Keith, E. McKinley, L. S. Peck, R. Calado, O. Defeo, S. Degraer, E. L. Johnston, H. Kaartokallio, P. I. Macreadie, A. Metaxas, A. W. N. Muthumbi, D. O. Obura, D. M. Paterson, A. R. Piola, A. J. Richardson, I. R. Schloss, P. V. R. Snelgrove, B. D. Stewart, P. M. Thompson, G. J. Watson, T. A. Worthington, M. Yasuhara, and W. J. Sutherland. (2022). A global horizon scan of issues impacting marine and coastal biodiversity conservation. *Nature Ecology & Evolution 6* (9): 1262-1270. DOI:10.1038/s41559-022-01812-0

Hersh, S. L. and D. A. Duffield. (1990). Distinction between northwest Atlantic offshore and coastal bottlenose dolphins based on hemoglobin profile and morphometry. In S. Leatherwood & R. R. Reeves (Eds.), *The Bottlenose Dolphin* (pp. 129–139). San Diego, CA: Academic Press.

Hersh, S. L. and D. K. Odell. (1986). Mass stranding of Fraser's dolphin, *Lagenodelphis hosei*, in the western North Atlantic. *Marine Mammal Science 2* (1): 73–76.

Heyning, J. E. (1989). Cuvier's beaked whale, *Ziphius cavirostris* G. Cuvier, 1823. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 4, pp. 289–308). San Diego, CA: Academic Press.

Hickmott, L. S. (2005). *Diving behaviour and foraging ecology of Blainville's and Cuvier's beaked whales in the Northern Bahamas.* (Unpublished master's thesis). University of St. Andrews, Scotland, United Kingdom.

Hidalgo-Ruz, V., L. Gutow, R. C. Thompson, and M. Thiel. (2012). Microplastics in the marine environment: A review of methods used for identification and quantification. *Environmental Science and Technology 46* 3060–3075. DOI:10.1021/es2031505

Hieb, E. E., R. H. Carmichael, A. Aven, C. Nelson-Seely, and N. Taylor. (2017). Sighting demographics of the West Indian manatee *Trichechus manatus* in the north-central Gulf of Mexico supported by citizen-sourced data. *Endangered Species Research 32* 321–332.

Hildebrand, J. A., K. E. Frasier, S. Baumann-Pickering, S. M. Wiggins, K. P. Merkens, L. P. Garrison, M. S. Soldevilla, and M. A. McDonald. (2019). Assessing Seasonality and Density From Passive Acoustic Monitoring of Signals Presumed to be From Pygmy and Dwarf Sperm Whales in the Gulf of Mexico. *Frontiers in Marine Science 6*. DOI:10.3389/fmars.2019.00066

Hill, A. N., C. Karniski, J. Robbins, T. Pitchford, S. Todd, and R. Asmutis-Silvia. (2017). Vessel collision injuries on live humpback whales, *Megaptera novaeangliae*, in the southern Gulf of Maine. *Marine Mammal Science 33* (2): 558–573. DOI:10.1111/mms.12386

Hill, M. (1998). Spongivory on Caribbean reefs releases corals from competition with sponges. *Oecologia 117* (1–2): 143–150.

Hillis, Z. (1990). *Buck Island Reef National Sea Turtle Research Program*. Presented at the Tenth Annual Workshop on Sea Turtle Biology and Conservation. Hilton Head, SC.

Hilton, E. J., B. Kynard, M. T. Balazik, A. Z. Horodysky, and C. B. Dillman. (2016). Review of the biology, fisheries, and conservation status of the Atlantic Sturgeon, (*Acipenser oxyrinchus oxyrinchus* Mitchill, 1815). *Journal of Applied Ichthyology 32* (S1): 30–66. DOI:https://doi.org/10.1111/jai.13242

Hirth, H. F. (1997). *Synopsis of the Biological Data on the Green Turtle Chelonia mydas (Linnaeus 1758)*. Washington, DC: U.S. Fish and Wildlife Service.

Hislop, J. R. G. and R. G. J. Shelton. (1993). Marine predators and prey of Atlantic salmon (*Salmo salar L*.). In D. Mills (Ed.), *Salmon in the Sea and New Enhancement Strategies* (pp. 104–118). Oxford, United Kingdom: Fishing News Books.

Hislop, J. R. G. and A. F. Youngson. (1984). *A note on the stomach contents of salmon caught by longline north of the Faroe Islands in March* (International Council for the Exploration of the Sea Conference and Meeting Documents). Aberdeen, United Kingdom: Department of Agriculture and Fisheries Marine Laboratory.

Hochscheid, S. (2014). Why we mind sea turtles' underwater business: A review on the study of diving behavior. *Journal of Experimental Marine Biology and Ecology 450* 118–136.

Hodge, L. and A. Read. (2013). *Passive Acoustic Monitoring for Marine Mammals at Site A in Jacksonville, FL, August 2010–January 2011*. Norfolk, VA: Marine Physical Laboratory, Scripps Institution of Oceanography, and Duke University Marine Laboratory.

Hodge, L. and A. Read. (2015). *Passive Acoustic Monitoring for Marine Mammals at Site E in Onslow Bay, October 2012 – June 2013*. Norfolk, VA: U.S. Department of the Navy.

Hodge, L. and A. Read. (2018). *Passive acoustic monitoring for marine mammals at Site E in Onslow Bay, October 2016 – June 2017*. Norfolk, VA: U.S. Department of the Navy.

Hodge, L., J. Stanistreet, and A. Read. (2014). *Passive Acoustic Monitoring for Cetaceans in Navy OPAREAS off the U.S. Atlantic Coast, January 2013–December 2013*. Norfolk, VA: Duke University Marine Laboratory and the U.S. Department of the Navy.

Hodge, L., J. Stanistreet, and A. Read. (2015). *Annual Report 2014: Passive Acoustic Monitoring for Marine Mammals off Virginia, North Carolina, and Florida Using High-Frequency Acoustic Recording Packages*. Virginia Beach, VA: U.S. Department of the Navy.

Hodge, L., J. Stanistreet, and A. Read. (2016). *Annual Report 2015: Passive Acoustic Monitoring for Marine Mammals off Virginia, North Carolina, and Florida Using High-Frequency Acoustic Recording Packages*. Virginia Beach, VA: HDR, Inc.

Hodge, L. E. W. (2011). *Monitoring Marine Mammals in Onslow Bay, North Carolina, Using Passive Acoustics.* (Doctoral Dissertation in Philosophy). Duke University, Durham, NC. Retrieved from https://dukespace.lib.duke.edu.

Hodge, L. E. W., J. T. Bell, and A. Kumar. (2013). The influence of habitat and time of day on the occurrence of odontocete vocalizations in Onslow Bay, North Carolina. *Marine Mammal Science 29* (4): E411–E427. DOI:10.1111/mms.12006

Hoegh-Guldberg, O. and J. F. Bruno. (2010). The impact of climate change on the world's marine ecosystems. *Science 328* (5985): 1523–1528. DOI:10.1126/science.1189930

Hoelzel, A. R. (2002). *Marine Mammal Biology: An Evolutionary Approach*. Malden, MA: Blackwell Publishing.

Hoelzel, A. R., E. M. Dorsey, and S. J. Stern. (1989). The foraging specializations of individual minke whales. *Animal Behaviour 38* 786–794.

Hoffmayer, E. R., J. S. Franks, W. B. Driggers, and P. W. Howey. (2013). Diel vertical movements of a scalloped hammerhead, *Sphyrna lewini*, in the northern Gulf of Mexico. *Bulletin of Marine Science 89* (2): 551–557.

Holland, R. A., J. L. Kirschvink, T. G. Doak, and M. Wikelski. (2008). Bats use magnetite to detect the earth's magnetic field. *PLoS ONE 3* (2): e1676. DOI:10.1371/journal.pone.0001676

Holland, R. A., K. Thorup, M. J. Vonhof, W. W. Cochran, and M. Wikelski. (2006). Navigation: Bat orientation using Earth's magnetic field. *Nature 444* (7120): 702. DOI:10.1038/444702a

Holloway-Adkins, K. G. (2006). Juvenile green turtles (*Chelonia mydas*) forage on high-energy, shallow reef on the east coast of Florida. In M. Frick, A. Panagopoulou, A. F. Rees, & K. Williams (Eds.), *Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation: Book of Abstracts*. Athens, Greece: National Marine Fisheries Service Southeast Fisheries Science Center, International Sea Turtle Society.

Homer, C., J. Dewitz, S. Jin, G. Xian, C. Costello, P. Danielson, L. Gass, M. Funk, J. Wickham, S. Stehman, R. Auch, and K. Riitters. (2020). Conterminous United States land cover change patterns 2001–2016 from the 2016 National Land Cover Database. *ISPRS Journal of Photogrammetry and Remote Sensing 162* 184–199. DOI:https://doi.org/10.1016/j.isprsjprs.2020.02.019

Hooker, S. K., R. W. Baird, and A. Fahlman. (2009). Could beaked whales get the bends? Effect of diving behaviour and physiology on modelled gas exchange for three species: *Ziphius cavirostris, Mesoplodon densirostris* and *Hyperoodon ampullatus*. *Respiratory Physiology & Neurobiology 167* (3): 235–246. DOI:10.1016/j.resp.2009.04.023

Hooker, S. K. and H. Whitehead. (2002). Click characteristics of northern bottlenose whales (*Hyperoodon ampullatus*). *Marine Mammal Science 18* (1): 69–80.

Hopkins-Murphy, S. R., D. W. Owens, and T. M. Murphy. (2003). Ecology of immature loggerheads on foraging grounds and adults in internesting habitat in the eastern United States. In A. B. Bolten & B. E. Witherington (Eds.), *Loggerhead Sea Turtles* (pp. 79–92). Washington, DC: Smithsonian Institution Press.

Horn, C., M. Karnauskas, J. C. Doerr, M. H. Miller, M. Neuman, R. Hill, and K. J. McCarthy. (2022). *Endangered Species Act Status Review Report: Queen conch (Aliger gigas)*. Miami, FL: National Marine Fisheries Service, Southeast Fisheries Science Center.

Horne, A. N. and C. P. Stence. (2016). *Assessment of Critical Habitats for Recovering the Chesapeake Bay Atlantic Sturgeon Distinct Population Segment* (Final Report to the Maryland Department of Natural Resources, Stevensville, MD). Stevensville, MD: Maryland Department of Natural Resources, Fishing and Boating Services.

Horrocks, J. A. (1987). Leatherbacks in Barbados. *Marine Turtle Newsletter 41* 7.

Horrocks, J. A., S. Stapleton, H. Guada, C. Lloyd, E. Harris, M. Fastigi, J. Berkel, K. Stewart, J. Gumbs, and K. L. Eckert. (2016). International movements of adult female leatherback turtles in the

Caribbean: results from tag recovery data (2002–2013). *Endangered Species Research 29* (3): 279–287.

Horwood, J. (2009). Sei whale, *Balaenoptera borealis*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1001–1003). Cambridge, MA: Academic Press.

Hostetler, J. A., H. H. Edwards, J. Martin, and P. Schueller. (2018). *Updated statewide abundance estimates for the Florida manatee*. St. Petersburg, FL: Florida Fish and Wildlife Conservation Commission, Fish and Wildlife Research Institute.

Hostetler, J. A., J. Martin, M. Kosempa, H. H. Edwards, K. A. Rood, S. L. Barton, and M. C. Runge. (2021). Reconstructing population dynamics of a threatened marine mammal using multiple data sets. *Scientific Reports 11* (1): 2702. DOI:10.1038/s41598-021-81478-z

Hotchkin, C. and S. Parks. (2013). The Lombard effect and other noise-induced vocal modifications: Insight from mammalian communication systems. *Biological Reviews of the Cambridge Philosophical Society 88* (4): 809–824. DOI:10.1111/brv.12026

Houghton, J. D. R., M. J. Callow, and G. C. Hays. (2003). Habitat utilization by juvenile hawksbill turtles (*Eretmochelys imbricata*, Linnaeus, 1766) around a shallow water coral reef. *Journal of Natural History 37* 1269–1280. DOI:10.1080/00222930110104276

Hourigan, T. F., J. Reed, S. Pomponi, S. W. Ross, A. W. David, and S. Harter. (2017). State of the Deep-Sea Coral and Sponge Ecosystems off the Southeast United States. In *The State of Deep-Sea Coral and Sponge Ecosystems in the United States* (pp. 13-11–13-61). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Habitat Conservation.

Houston, J. (1990). Status of Hubb's beaked whale, *Mesoplodon carlhubbsi*, in Canada. *Canadian Field-Naturalist 104* 121–124.

Howell, L. N. and D. J. Shaver. (2021). Foraging habits of green sea turtles (*Chelonia mydas*) in the northwestern Gulf of Mexico. *Frontiers in Marine Science 8*. DOI:10.3389/fmars.2021.658368

Hughes, T. P., A. H. Baird, D. R. Bellwood, M. Card, S. R. Connolly, C. Folke, R. Grosberg, O. Hoegh-Guldberg, J. B. C. Jackson, J. Kleypas, J. M. Lough, P. A. Marshall, M. Nystrom, S. R. Palumbi, J. M. Pandolfi, B. Rosen, and J. Roughgarden. (2003). Climate change, human impacts, and the resilience of coral reefs. *Science 301* (5635): 929–933.

Hughes, T. P. and J. E. Tanner. (2000). Recruitment Failure, Life Histories, and Long-Term Decline of Caribbean Corals. *Ecology 81* (8): 2250–2263.

Humber, F., B. J. Godley, and A. C. Broderick. (2014). So excellent a fishe: A global overview of legal marine turtle fisheries. *Diversity and Distributions 20* (5): 579–590. DOI:10.1111/ddi.12183

Iafrate, J., E. Reyier, B. Ahr, S. Watwood, D. Scheidt, J. Provancha, K. Holloway-Adkins, A. Carroll, A. DiMatteo, J. Greene, and J. Krumholz. (2022). *Behavior, seasonality, and habitat preferences of mobile fishes and sea turtles within a large sand shoal complex: Habitat Connectivity, Ocean Glider Surveys, and Passive Acoustics*. Sterling, VA: U.S. Department of the Interior, Bureau of Ocean Energy Management.

Ingram, E. C., R. M. Cerrato, K. J. Dunton, and M. G. Frisk. (2019). Endangered Atlantic sturgeon in the New York Wind Energy Area: Implications of future development in an offshore wind energy site. *Scientific Reports 9* (1): 12432. DOI:10.1038/s41598-019-48818-6

Ingram, S. N., L. Walshe, D. Johnston, and E. Rogan. (2007). Habitat partitioning and the influence of benthic topography and oceanography on the distribution of fin and minke whales in the Bay of Fundy, Canada. *Journal of the Marine Biological Association of the United Kingdom 87* 149–156.

International Council for the Exploration of the Sea. (2014). *Report of the ICES/NAFO Working Group on Harp and Hooded Seals*. Quebec City, Canada: ICES Advisory Committee.

International Council of the Exploration of the Sea. (1993). *Report of the Study Group on Long-Finned Pilot Whales*. Copenhagen, Denmark: International Council for the Exploration of the Sea.

International Union for Conservation of Nature. (2019). *Malaclemys terrapin, Diamondback Terrapin*. Retrieved February 20, 2024, from http://dx.doi.org/10.2305/IUCN.UK.2019-1.RLTS.T12695A507698.en.

International Union for Conservation of Nature and Natural Resources. (2010a). *Phoebastria albatrus*. *International Union for Conservation of Nature 2010. International Union for Conservation of Nature Red List of Threatened Species. Version 2010.3.* Retrieved from http://www.iucnredlist.org.

International Union for Conservation of Nature and Natural Resources. (2010b). *Pterodroma cahow*. *International Union for Conservation of Nature Red List of Threatened Species. Version 2010.4.* Retrieved from http://www.iucnredlist.org/.

Irigoien, X., T. A. Klevjer, A. Røstad, U. Martinez, G. Boyra, J. L. Acuña, A. Bode, F. Echevarria, J. I. Gonzalez-Gordillo, S. Hernandez-Leon, S. Agusti, D. L. Aksnes, C. M. Duarte, and S. Kaartvedt. (2014). Large mesopelagic fishes biomass and trophic efficiency in the open ocean. *Nature Communications 5* (1): 3271. DOI:10.1038/ncomms4271

Irvine, A. B. (1983). Manatee metabolism and its influence on distribution in Florida. *Biological Conservation 25* (4): 315–334. DOI:10.1016/0006-3207(83)90068-X

Isbell, F., P. Balvanera, A. S. Mori, J.-S. He, J. M. Bullock, G. R. Regmi, E. W. Seabloom, S. Ferrier, O. E. Sala, N. R. Guerrero-Ramírez, J. Tavella, D. J. Larkin, B. Schmid, C. L. Outhwaite, P. Pramual, E. T. Borer, M. Loreau, T. C. Omotoriogun, D. O. Obura, M. Anderson, C. Portales-Reyes, K. Kirkman, P. M. Vergara, A. T. Clark, K. J. Komatsu, O. L. Petchey, S. R. Weiskopf, L. J. Williams, S. L. Collins, N. Eisenhauer, C. H. Trisos, D. Renard, A. J. Wright, P. Tripathi, J. Cowles, J. E. Byrnes, P. B. Reich, A. Purvis, Z. Sharip, M. I. O'Connor, C. E. Kazanski, N. M. Haddad, E. H. Soto, L. E. Dee, S. Díaz, C. R. Zirbel, M. L. Avolio, S. Wang, Z. Ma, J. Liang, H. C. Farah, J. A. Johnson, B. W. Miller, Y. Hautier, M. D. Smith, J. M. Knops, B. J. Myers, Z. V. Harmáčková, J. Cortés, M. B. Harfoot, A. Gonzalez, T. Newbold, J. Oehri, M. Mazón, C. Dobbs, and M. S. Palmer. (2023). Expert perspectives on global biodiversity loss and its drivers and impacts on people. *Frontiers in Ecology and the Environment 21* (2): 94-103. DOI:https://doi.org/10.1002/fee.2536

Iversen, E. S., D. E. Jory, and S. P. Bannerot. (1986). Predation on queen conchs, *Strombus gigas*, in the Bahamas. *Bulletin of Marine Science 39* (1): 61–75.

Iverson, A. R., A. M. Benscoter, I. Fujisaki, M. M. Lamont, and K. M. Hart. (2020). Migration corridors and threats in the Gulf of Mexico and Florida Straits for loggerhead sea turtles. *Frontiers in Marine Science 7* 1–12. DOI:10.3389/fmars.2020.00208

Iverson, A. R., I. Fujisaki, M. M. Lamont, and K. M. Hart. (2019). Loggerhead sea turtle (*Caretta caretta*) diving changes with productivity, behavioral mode, and sea surface temperature. *PLoS ONE 14* (8): 1-19. DOI:https://doi.org/10.1371/journal.pone.0220372

Jaap, W. C. (1984). *The ecology of the South Florida coral reefs: A community profile* (*Southwest Florida Shelf Coastal Ecological Characterization*). Washington, DC: U.S. Fish and Wildlife Service.

Jaap, W. C., J. C. Halas, and R. G. Muller. (1988). *Community dynamics of stony corals (Milleporina and Scleractinia) at Key Largo National Marine Sanctuary, Florida during 1981–1986*. Presented at the 6th International Coral Reef Symposium. Townsville, Australia.

Jaap, W. C., J. W. Porter, J. L. Wheaton, C. Beaver, K. E. Hackett, M. J. Lybolt, M. K. Callahan, J. Kidney, S. Kupfner, C. Torres, and K. Sutherland. (2002). *EPA/NOAA Coral Reef Evaluation and Monitoring Project (CREMP), 2002 Executive Summary*. St. Petersburg, FL: Florida Marine Research Institute.

Jackson, J. B. C., M. K. Donovan, K. L. Cramer, and V. V. Lam. (2014). *Status and Trends of Caribbean Coral Reefs: 1970–2012*. Gland, Switzerland: Global Coral Reef Monitoring Network, International Union for the Conservation of Nature.

Jacobs, S. R. and J. M. Terhune. (2000). Harbor seal (*Phoca vitulina*) numbers along the New Brunswick coast of the Bay of Fundy in autumn in relation to aquaculture. *Northeastern Naturalist 7* (3): 289–296.

James, M. C., S. A. Eckert, and R. A. Myers. (2005a). Migratory and reproductive movements of male leatherback turtles (*Dermochelys coriacea)*. *Marine Biology 147* 845–853. DOI:10.1007/s00227-005-1581-1

James, M. C. and T. B. Herman. (2001). Feeding of *Dermochelys coriacea* on medusae in the northwest Atlantic. *Chelonian Conservation and Biology 4* (1): 202–205.

James, M. C., R. A. Myers, and C. A. Ottensmeyer. (2005b). Behaviour of leatherback sea turtles, *Dermochelys coriacea*, during the migratory cycle. *Proceedings of the Royal Society B: Biological Sciences 272* 1547–1555. DOI:10.1098/rspb.2005.3110

James, M. C., C. A. Ottensmeyer, and R. A. Myers. (2005c). Identification of high-use habitat and threats to leatherback sea turtles in northern waters: New directions for conservations. *Ecology Letters 8* 195–201. DOI:10.1111/j.1461-0248.2004.00710

James, M. C., S. A. Sherrill-Mix, K. Martin, and R. A. Myers. (2006). Canadian waters provide critical foraging habitat for leatherback sea turtles. *Biological Conservation 133* (3): 347–357. DOI:10.1016/j.biocon.2006.06.012

James, R. K., M. J. A. Christianen, M. M. van Katwijk, J. C. de Smit, E. S. Bakker, P. M. J. Herman, and T. J. Bouma. (2020). Seagrass coastal protection services reduced by invasive species expansion and megaherbivore grazing. *Journal of Ecology 108* (5): 2025–2037. DOI:10.1111/1365-2745.13411

Jaquet, N. and D. Gendron. (2009). The social organization of sperm whales in the Gulf of California and comparisons with other populations. *Journal of the Marine Biological Association of the United Kingdom 89* (5): 975–983. DOI:10.1017/S0025315409001507

Jay, C. V., A. S. Fischbach, and A. A. Kochnev. (2012). Walrus areas of use in the Chukchi Sea during sparse sea ice cover. *Marine Ecology Progress Series 468* 1–13. DOI:10.3354/meps10057

Jefferson, T. A. (2009a). Clymene dolphin, *Stenella clymene*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 241–243). Cambridge, MA: Academic Press.

Jefferson, T. A. (2009b). Rough-toothed dolphin, *Steno bredanensis*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 990–992). Cambridge, MA: Academic Press.

Jefferson, T. A. and N. B. Barros. (1997). *Peponocephala electra*. *Mammalian Species 553* 1–6.

Jefferson, T. A., D. Fertl, J. Bolanos Jiminez, and A. N. Zerbini. (2009). Distribution of common dolphins (*Delphinus spp.*) in the western Atlantic Ocean: A critical re-examination. *Marine Biology 156* 1109–1124.

Jefferson, T. A. and S. Leatherwood. (1994). *Lagenodelphis hosei*. *American Society of Mammalogists 470* 1–5.

Jefferson, T. A., M. A. Webber, and R. L. Pitman. (2008). *Marine Mammals of the World: A Comprehensive Guide to Their Identification*. London, United Kingdom: Elsevier.

Jefferson, T. A., M. A. Webber, and R. L. Pitman. (2015). *Marine Mammals of the World: A Comprehensive Guide to Their Identification* (2nd ed.). Cambridge, MA: Academic Press.

Jenkins, C. and T. Wever. (2007). Naval mine impact burial prediction using seafloor database, experiment, and GIS technologies. *Marine Georesources and Geotechnology 25* (3-4): 199-208.

Jensen, A. S. and G. K. Silber. (2003). *Large Whale Ship Strike Database*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

Jensen, M. E., C. F. Moss, and A. Surlykke. (2005). Echolocating bats can use acoustic landmarks for spatial orientation. *The Journal of Experimental Biology 208* (23): 4399–4410. DOI:10.1242/jeb.01901

Jensen, M. P., C. D. Allen, T. Eguchi, I. P. Bell, E. L. LaCasella, W. A. Hilton, C. A. M. Hof, and P. H. Dutton. (2018). Environmental warming and feminization of one of the largest sea turtle populations in the world. *Current Biology 28* (1): 154–159. DOI:https://doi.org/10.1016/j.cub.2017.11.057

Jessup, D. A., M. A. Miller, J. P. Ryan, H. M. Nevins, H. A. Kerkering, A. Mekebri, D. B. Crane, T. A. Johnson, and R. M. Kudela. (2009). Mass stranding of marine birds caused by a surfactant-producing red tide. *PLoS ONE 4* (2): e4550. DOI:10.1371/journal.pone.0004550

Jiménez, S., A. Domingo, M. Abreu, and A. Brazeiro. (2012). Bycatch susceptibility in pelagic longline fisheries: Are albatrosses affected by the diving behaviour of medium-sized petrels? *Aquatic Conservation: Marine and Freshwater Ecosystems 22* (4): 436–445. DOI:10.1002/aqc.2242

Jodice, P. G. R., P. E. Michael, J. S. Gleason, J. C. Haney, and Y. G. Satgé. (2021). Revising the marine range of the endangered black-capped petrel *Pterodroma hasitata*: occurrence in the northern Gulf of Mexico and exposure to conservation threats. *Endangered Species Research 46* 49–65. DOI:10.3354/esr01143

Jodice, P. G. R., R. A. Ronconi, E. Rupp, G. E. Wallace, and Y. Satgé. (2015). First satellite tracks of the Endangered black-capped petrel. *Endangered Species Research 29* (1): 23–33. DOI:10.3354/esr00697

Johansen, S., O. N. Larsen, J. Christensen-Dalsgaard, L. Seidelin, T. Huulvej, K. Jensen, S. G. Lunneryd, M. Bostrom, and M. Wahlberg. (2016). In-air and underwater hearing in the great cormorant (*Phalacrocorax carbo sinensis*). *Advances in Experimental Medicine and Biology 875* 505–512. DOI:10.1007/978-1-4939-2981-8_61

Johnson, J., J. Gates, and N. Zegre. (2011). Monitoring seasonal bat activity on a coastal barrier island in Maryland, USA. *Environmental Monitoring and Assessment 173* 685–699.

Johnson, M. L. (1989). Juvenile leatherback cared for in captivity. *Marine Turtle Newsletter 47* 13–14.

Johnston, D. W., J. Frungillo, A. Smith, K. Moore, B. Sharp, J. Schuh, and A. J. Read. (2015). Trends in stranding and by-catch rates of gray and harbor seals along the northeastern coast of the United States: evidence of divergence in the abundance of two sympatric phocid species? *PLoS ONE 10* (7): e0131660. DOI:10.1371/journal.pone.0131660

Johnston, D. W., L. H. Thorne, and A. J. Read. (2005). Fin whales, *Balaenoptera physalus,* and minke whales, *Balaenoptera acutorostrata,* exploit a tidally driven island wake ecosystem in the Bay of Fundy. *Marine Ecology Progress Series 305* 287–295.

Jones, D. V. and D. R. Rees. (2022). *Haul-out Counts and Photo-Identification of Pinnipeds in Chesapeake Bay and Eastern Shore, Virginia: 2020/2021 Annual Progress Report*. Norfolk, VA: Naval Facilities Engineering Systems Command Atlantic.

Jones, D. V. and D. R. Rees. (2023). *Haul-out Counts and Photo-Identification of Pinnipeds in Chesapeake Bay and Eastern Shore, Virginia: 2021/2022 Annual Progress Report*. Norfolk, VA: Naval Facilities Engineering Systems Command Atlantic.

Jones, E. S., S. W. Ross, C. M. Robertson, and C. M. Young. (2022). Distributions of microplastics and larger anthropogenic debris in Norfolk Canyon, Baltimore Canyon, and the adjacent continental slope (Western North Atlantic Margin, U.S.A.). *Marine Pollution Bulletin 174* 113047. DOI:10.1016/j.marpolbul.2021.113047

Jones, T., J. K. Parrish, J. Lindsey, C. Wright, H. K. Burgess, J. Dolliver, L. Divine, R. Kaler, D. Bradley, G. Sorenson, R. Torrenta, S. Backensto, H. Coletti, J. Harvey, H. M. Nevins, E. Donnelly-Greenan, D. L. Sherer, J. Roletto, and K. Lindquist. (2023). Marine bird mass mortality events as an indicator of the impacts of ocean warming. *Marine Ecology Progress Series* 1–21. DOI:10.3354/meps14330

Jory, D. E. and E. S. Iversen. (1983). *Queen conch predators: not a roadblock to mariculture*. Presented at the Proceedings of the 35th Gulf and Caribbean Fisheries Institute. Nassau, Bahamas.

Jutila, E. and J. Toivonen. (1985). *Food composition of salmon postsmolts (Salmo salar L.) in the northern part of the Gulf of Bothnia*. Helsinki, Finland: ICES Conference and Meeting Documents.

Kadison, E., R. S. Nemeth, J. Blondeau, T. Smith, and J. Calnan. (2010). *Nassau grouper (Epinephelus striates) in St. Thomas, US Virgin Islands, with evidence for a spawning aggregation site recovery*. Presented at the Proceedings of the Gulf and Caribbean Fisheries Institute. San Juan, PR.

Kahn, J. E., C. Hager, J. C. Watterson, N. Mathies, and K. J. Hartman. (2019). Comparing abundance estimates from closed population mark-recapture models of endangered adult Atlantic sturgeon. *Endangered Species Research 39* 63–76. DOI:https://doi.org/10.3354/esr00957

Kahn, J. E., C. Hager, J. C. Watterson, J. Russo, K. Moore, and K. Hartman. (2014). Atlantic Sturgeon Annual Spawning Run Estimate in the Pamunkey River, Virginia. *Transactions of the American Fisheries Society 143* (6): 1508–1514. DOI:10.1080/00028487.2014.945661

Kanda, N., M. Goto, H. Kat, M. V. McPhee, and L. A. Pastene. (2007). Population genetic structure of Bryde's whales (*Balaenoptera brydei*) at the inter-oceanic and trans-equatorial levels. *Conservation Genetics 8* (4): 853–864. DOI:doi:10.1007/s10592-006-9232-8

Kannan, R., B. Chander, J. Jackson, and T. Chander. (2021). Remote Ornithology: Studying Nesting Behavior of Bermuda Petrels via Live Webcam *Journal of the Arkansas Academy of Science 75* 85-89.

Karlsson, R., M. Tivefälth, I. Duranović, S. Martinsson, A. Kjølhamar, and K. M. Murvoll. (2022). Artificial hard-substrate colonisation in the offshore Hywind Scotland Pilot Park. *Wind Energy Science 7* (2): 801–814. DOI:10.5194/wes-7-801-2022

Kato, H. and W. F. Perrin. (2009). Bryde's whales, *Balaenoptera edeni/brydei*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 158–163). Cambridge, MA: Academic Press.

Katona, S. K., J. A. Beard, P. E. Girton, and F. Wenzel. (1988). Killer whales (*Orcinus orca*) from the Bay of Fundy to the Equator, including the Gulf of Mexico. *Rit Fiskideildar (Journal of the Marine Research Institute Reykjavik) 11* 205–224.

Katona, S. K., V. Rough, and D. T. Richardson. (1993). *A Field Guide to Whales, Porpoises, and Seals from Cape Cod to Newfoundland* (Fourth ed.). Washington, DC: Smithsonian Institution Press.

Kaufman, K. (1990). *The Peterson Field Guide Series, A Field Guide to Advanced Birding: Birding Challenges and How to Approach Them*. Boston, MA: Houghton Mifflin Company.

Kealoha, A. K., S. M. Doyle, K. E. F. Shamberger, J. B. Sylvan, R. D. Hetland, and S. F. DiMarco. (2020). Localized hypoxia may have caused coral reef mortality at the Flower Garden Banks. *Coral Reefs 39* 119-132.

Keaster, A. J., J. A. Grundler, W. S. Craig, and M. A. Jackson. (1996). Noctuid Moths and Other Insects Captured in Wing-Style Traps Baited with Black Cutworm (Lepidoptera: Noctuidae) Pheromone on Offshore Oil Platforms in the Gulf of Mexico, 1988-1991. *Journal of the Kansas Entomological Society 69* (1): 17-25.

Keener, V. W., J. J. Marra, M. L. Finucane, D. Spooner, and M. H. Smith. (2012). *Climate Change and Pacific Islands: Indicators and Impacts. Report for the 2012 Pacific Islands Regional Climate Assessment*. Washington, DC: Island Press.

Keinath, J. A., J. A. Musick, and R. A. Byles. (1987). Aspects of the biology of Virginia's sea turtles: 1979–1986. *Virginia Journal of Science 38* (2): 81.

Kelley, J. R., K. L. Kelley, A. E. Savage, and K. L. Mansfield. (2022). Novel disease state model finds most juvenile green turtles develop and recover from fibropapillomatosis. *Ecosphere 13* (3): 1–13. DOI:10.1002/ecs2.4000

Kemp, N. J. (1996). Habitat loss and degradation. In M. P. Simmonds & J. D. Hutchinson (Eds.), *The Conservation of Whales and Dolphins* (pp. 263–280). New York, NY: John Wiley & Sons.

Kendall, M. S., O. P. Jensen, C. Alexander, D. Field, G. McFall, R. Bohne, and M. E. Monaco. (2005). Benthic mapping using sonar, video transects, and an innovative approach to accuracy assessment: a characterization of bottom features in the Georgia Bight. *Journal of Coastal Research 21* (6): 1154-1165.

Kenney, M. K. (1994). *Harbor seal population trends and habitat use in Maine.* (Unpublished master's thesis). University of Maine, Orono, ME.

Kenney, R. D. (1990). Bottlenose Dolphins off the Northeastern United States. In S. Leatherwood & R. R. Reeves (Eds.), *The Bottlenose Dolphin* (pp. 369–386). San Diego, CA: Academic Press.

Kenney, R. D. (2009). Right Whales (*Eubalaena glacialis, E. japonica*, and *E. australis*). In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 962–972). Cambridge, MA: Academic Press.

Kenney, R. D. (2014, April 17, 2014). *Marine Mammals of Rhode Island, Part 5, Harbor Seal*. Retrieved May 26, 2017, from http://rinhs.org/uncategorized/marine-mammals-of-rhode-island-part-5-harbor-seal/.

Kenney, R. D., C. A. Mayo, and H. E. Winn. (2001). Migration and foraging strategies at varying spatial scales in western north Atlantic right whales: A review of hypotheses. *Journal of Cetacean Research and Management Special Issue 2* 237–250.

Kenney, R. D. and H. E. Winn. (1986). Cetacean high-use habitats of the northeast United States continental shelf. *Fishery Bulletin 84* (2): 345–357.

Kerlinger, P. (2009). *How Birds Migrate* (2nd ed.). Mechanicsburg, PA: Stackpole Books.

Ketchum, J. T., A. Hearn, A. P. Klimley, E. Espinoza, C. Penaherrera, and J. L. Largier. (2014a). Seasonal changes in movements and habitat preferences of the scalloped hammerhead shark (*Sphyrna lewini*) while refuging near an oceanic island. *Marine Biology 161* (4): 755–767.

Ketchum, J. T., A. Hearn, A. P. Klimley, C. Penaherrera, E. Espinoza, S. Bessudo, G. Soler, and R. Arauz. (2014b). Inter-island movements of scalloped hammerhead sharks (*Sphyrna lewini*) and seasonal connectivity in a marine protected area of the eastern tropical Pacific. *Marine Biology 161* (4): 939–951.

Khan, C. B., P. Duley, A. H. Glass, J. Gatzke, L. Crowe, and T. V. N. Cole. (2016). *North Atlantic Right Whale Sighting Survey (NARWSS) and Right Whale Sighting Advisory System (RWSAS) 2014 results summary* (Reference Document 16-01). Woods Hole, MA: Northeast Fisheries Science Center.

Khan, C. B., A. Henry, L. Crowe, P. Duley, J. Gatzke, and T. V. N. Cole. (2018). *North Atlantic Right Whale Sighting Survey (NARWSS) and Right Whale Sighting Advisory System (RWSAS) 2016 Results Summary* (Reference Document 18-01). Woods Hole, MA: Northeast Fisheries Science Center.

Kim, J., M. Stekoll, and C. Yarish. (2019). Opportunities, challenges and future directions of open-water seaweed aquaculture in the United States. *Phycologia 58* (5): 446–461. DOI:10.1080/00318884.2019.1625611

King, C. D., E. Chou, M. L. Rekdahl, S. G. Trabue, and H. C. Rosenbaum. (2021). Baleen whale distribution, behaviour and overlap with anthropogenic activity in coastal regions of the New York Bight. *Marine Biology Research 17* (4): 380–400. DOI:10.1080/17451000.2021.1967993

King, L. (2011, March 28). *Sei whale found stranded in Virginia Beach*. *The Virginian-Pilot.* Retrieved May 23, 2017, from http://pilotonline.com/news/sei-whale-found-stranded-in-virginia-beach/article_2a66b71b-ff9c-506d-a621-9291983a3c7c.html.

Kinze, C. C. (2009). White-beaked dolphin, *Lagenorhynchus albirostris*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1255–1258). Cambridge, MA: Academic Press.

Kirkham, I. R. and D. N. Nettleship. (1987). Status of the roseate tern in Canada. *Journal of Field Ornithology 58* (4): 505–515.

Kiszka, J. J., M. Caputo, J. Vollenweider, M. R. Heithaus, L. Aichinger Dias, and L. P. Garrison. (2023). Critically endangered Rice's whales (*Balaenoptera ricei*) selectively feed on high-quality prey in the Gulf of Mexico. *Scientific Reports 13* (1): 6710. DOI:10.1038/s41598-023-33905-6

Kleypas, J. A., J. W. McManus, and L. A. Meñez. (1999). Environmental limits to coral reef development: Where do we draw the line? *American Zoologist 39* (1): 146-159.

Knopf, F. L. and R. M. Evans. (2004). American White Pelican (*Pelecanus erythrorhynchos*). *The Birds of North America Online 57* 6. DOI:10.2173/bna.57

Knowlton, A. R., P. K. Hamilton, M. K. Marx, H. M. Pettis, and S. D. Kraus. (2012). Monitoring North Atlantic right whale *Eubalaena glacialis* entanglement rates: A 30 yr retrospective. *Marine Ecology Progress Series 466* 293–302. DOI:10.3354/meps09923

Knowlton, A. R., F. T. Korsmeyer, J. E. Kerwin, H. Y. Wu, and B. Hynes. (1995). *The Hydrodynamic Effects of Large Vessels on Right Whales*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Knowlton, A. R. and S. D. Kraus. (2001). Mortality and serious injury of northern right whales (*Eubalaena glacialis*) in the western North Atlantic Ocean. *Journal of Cetacean Resource Management Special Issue 2* 193–208.

Koay, G., H. E. Heffner, and R. S. Heffner. (1997). Audiogram of the big brown bat (*Eptesicus fuscus*). *Hearing Research 105* 202–210.

Koen-Alonso, M., S. N. Pedraza, A. C. M. Schiavini, R. N. P. Goodall, and E. A. Crespo. (1999). Stomach contents of false killer whales (*Pseudorca crassidens*) stranded on the coasts of the Strait of Magellan, Tierra Del Fuego. *Marine Mammal Science 15* (3): 712–724.

Kohler, N. E. and P. A. Turner. (2001). Shark tagging: A review of conventional methods and studies. *Environmental Biology of Fishes 60* (1–3): 191–223. DOI:10.1023/a:1007679303082

Kovacs, K. M. (2009a). Bearded seal, *Erignathus barbatus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 97–101). Cambridge, MA: Academic Press.

Kovacs, K. M. (2009b). Hooded seal, *Cystophora cristata*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 569–573). Cambridge, MA: Academic Press.

Kraus, S. D., J. H. Prescott, and G. S. Stone. (1983). *Harbor Porpoise, Phocoena phocoena, in the U.S. Coastal Waters off the Gulf of Maine: A survey to Determine Seasonal Distribution and Abundance*. Boston, MA: National Marine Fisheries Service.

Krkosek, M., C. W. Revie, P. G. Gargan, O. T. Skilbrei, B. Finstad, and C. D. Todd. (2013). Impact of parasites on salmon recruitment in the Northeast Atlantic Ocean. *Proceedings of the Royal Society B: Biological Sciences 280* (1750): 20122359. DOI:10.1098/rspb.2012.2359

Kroglund, F., B. Finstad, S. O. Stefansson, T. O. Nilsen, T. Kristensen, B. O. Rosseland, H. C. Teien, and B. Salbu. (2007). Exposure to moderate acid water and aluminum reduces Atlantic salmon post-smolt survival. *Aquaculture 273* (2–3): 360–373. DOI:10.1016/j.aquaculture.2007.10.018

Kruse, S., D. K. Caldwell, and M. C. Caldwell. (1999). Risso's dolphin *Grampus griseus* (G. Cuvier, 1812). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 183–212). San Diego, CA: Academic Press.

Kuker, K. J., J. A. Thomson, and U. Tscherter. (2005). Novel surface feeding tactics of minke whales, *Balaenoptera acutorostrata*, in the Saguenay-St. Lawrence National Marine Park. *Canadian Field-Naturalist 119* (2): 214–218.

Kumar, A., J. Nissen, T. Norris, J. Oswald, T. Yack, and E. Ferguson. (2013, June 12–15 2013). *Using Passive Acoustics to Monitor the Presence of Marine Mammals during Naval Exercises* [Poster]. Presented at the 6th International Workshop on Detection, Classification, Localization and

Density Estimation of Marine Mammals using Passive Acoustics in University of St. Andrews, St. Andrews, Scotland. Retrieved from The 6th International Workshop on Detection, Classification, Localization and Density Estimation of Marine Mammals using Passive Acoustics.

Kuschke, S., J. Wyneken, C. Cray, E. Turla, M. Kinsella, and D. Miller. (2023). Fusarium spp. an emerging fungal threat to leatherback (*Dermochelys coriacea*) eggs and neonates. *Frontiers in Marine Science 10* 1–12. DOI:10.3389/fmars.2023.1170376

Kynard, B. (1997). Life history, latitudinal patterns and status of the shortnose sturgeon, *Acipenser brevirostrum*. *Environmental Biology of Fishes 48* (1–4): 319–334.

LaBrecque, E., C. Curtice, J. Harrison, S. M. Van Parijs, and P. N. Halpin. (2015a). Biologically Important Areas for Cetaceans Within U.S. Waters—East Coast Region. *Aquatic Mammals 41* (1): 17–29. DOI:10.1578/am.41.1.2015.54

LaBrecque, E., C. Curtice, J. Harrison, S. M. Van Parijs, and P. N. Halpin. (2015b). Biologically Important Areas for Cetaceans Within U.S. Waters—Gulf of Mexico Region. *Aquatic Mammals 41* (1): 30–38. DOI:10.1578/am.41.1.2015.54

Laidre, K., I. Stirling, L. F. Lowry, O. Wiig, M. P. Heide-Jorgensen, and S. H. Ferguson. (2008). Quantifying the sensitivity of arctic marine mammals to climate-induced habitat change. *Ecological Applications 18* (2): S97–S125.

Laidre, K. L., H. Stern, K. M. Kovacs, L. Lowry, S. E. Moore, E. V. Regehr, S. H. Ferguson, O. Wiig, P. Boveng, R. P. Angliss, E. W. Born, D. Litovka, L. Quakenbush, C. Lydersen, D. Vongraven, and F. Ugarte. (2015). Arctic marine mammal population status, sea ice habitat loss, and conservation recommendations for the 21st century. *Conservation Biology 29* (3): 724–737. DOI:10.1111/cobi.12474

Laist, D. W., A. R. Knowlton, J. G. Mead, A. S. Collet, and M. Podesta. (2001). Collisions between ships and whales. *Marine Mammal Science 17* (1): 35–75.

Laloë, J.-O., N. Esteban, J. Berkel, and G. C. Hays. (2016). Sand temperatures for nesting sea turtles in the Caribbean: Implications for hatchling sex ratios in the face of climate change. *Journal of Experimental Marine Biology and Ecology 474* 92–99.

Laloë, J.-O. and G. C. Hays. (2023). Can a present-day thermal niche be preserved in a warming climate by a shift in phenology? A case study with sea turtles. *Royal Society Open Science 10* (2): 1–10. DOI:10.1098/rsos.221002

Lam, T., Lingxu, S. Takahashi, and E. A. Burgess. (2011). *Market Forces: An Examination of Marine Turtle Trade in China and Japan*. Hong Kong, China: TRAFFIC East Asia.

Lamont, M. M., A. M. Benscoter, and K. M. Hart. (2023). Green turtle movements in the Gulf of Mexico: Tracking reveals new migration corridor and habitat use suggestive of MPA expansion. *Global Ecology and Conservation 43* (e02380): 1-11. DOI:10.1016/j.gecco.2023.e02380

Lamont, M. M., I. Fujisaki, B. S. Stephens, and C. Hackett. (2015). Home range and habitat use of juvenile green turtles (*Chelonia mydas*) in the northern Gulf of Mexico. *Animal Biotelemetry 3* (1): 53.

Lamont, M. M. and A. R. Iverson. (2018). Shared habitat use by juveniles of three sea turtle species. *Marine Ecology Progress Series 606* 187–200. DOI:10.3354/meps12748

Landsberg, J. H. (2002). The effects of harmful algal blooms on aquatic organisms. *Reviews in Fisheries Science 10* (2): 113–390. DOI:10.1080/20026491051695

Laney, R. W., J. E. Hightower, R. Versak, M. F. Mangold, and S. E. Winslow. (2007). Distribution, habitat use, and size of Atlantic sturgeon captured during cooperative winter tagging cruises, 1988-2006. *American Fisheries Society Symposium 56* 167–182.

Langhamer, O., H. Holand, and G. Rosenqvist. (2016). Effects of an Offshore Wind Farm on the Common Shore Crab *Carcinus maenas*: Tagging Pilot Experiments in the Lillgrund Offshore Wind Farm (Sweden). *PLoS ONE 11* (10): 17. DOI:10.1371/journal.pone.0165096

Lapointe, B. E. (1986). Phosphorus-limited photosynthesis and growth of *Sargassum natans* and *Sargassum fluitans* (Phaeophyceae) in the western North Atlantic. *Deep Sea Research Part A. Oceanographic Research Papers 33* (3): 391–399. DOI:https://doi.org/10.1016/0198-0149(86)90099-3

Lavigne, D. M. (2009). Harp seal, *Pagophilus groenlandicus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 560–562). Cambridge, MA: Academic Press.

Law, K. L., S. E. Moret-Ferguson, D. S. Goodwin, E. R. Zettler, E. Deforce, T. Kukulka, and G. Proskurowski. (2014). Distribution of surface plastic debris in the eastern Pacific Ocean from an 11-year data set. *Environmental Science & Technology 48* (9): 4732–4738. DOI:10.1021/es4053076

Lazell, J. D., Jr. (1980). New England waters: Critical habitat for marine turtles. *Copeia 1980* (2): 290–295.

Leatherwood, J. S., W. F. Perrin, R. L. Garvie, and J. C. La Grange. (1973). *Observations of Sharks Attacking Porpoises (Stenella spp. and Delphinus cf. D. delphis)*. San Diego, CA: Naval Undersea Technical.

Leatherwood, S., D. K. Caldwell, and H. E. Winn. (1976). *Whales, Dolphins and Porpoises of the Western North Atlantic: A Guide to their Identification*. Seattle, WA: National Oceanic and Atmospheric Administration.

Leatherwood, S., T. A. Jefferson, J. C. Norris, W. E. Stevens, L. J. Hansen, and K. D. Mullin. (1993). Occurrence and sounds of Fraser's dolphins (*Lagenodelphis hosei*) in the Gulf of Mexico. *Texas Journal of Science 45* (4): 349–354.

Leatherwood, S. and R. R. Reeves. (1983). *The Sierra Club Handbook of Whales and Dolphins*. San Francisco, CA: Sierra Club Books.

Lee, D. S. (1987). December records of seabirds off North Carolina. *The Wilson Bulletin 99* (1): 116–121.

Lee, D. S. and W. A. Mackin. (2008, April 1, 2009). *Bermuda Petrel*. Retrieved 2010, from http://wicbirds.net/index.html.

Lefebvre, K. A., L. Quakenbush, E. Frame, K. B. Huntington, G. Sheffield, R. Stimmelmayr, A. Bryan, P. Kendrick, H. Ziel, T. Goldstein, J. A. Snyder, T. Gelatt, F. Gulland, b. Dickerson, and V. Gill. (2016). Prevalence of algal toxins in Alaskan marine mammals foraging in a changing arctic and subarctic environment. *Harmful Algae 55* (2016): 13–24.

Lefebvre, L. W., M. Marmontel, J. P. Reid, G. B. Rathbun, and D. P. Domning. (2001). Status and biogeography of the West Indian manatee. In C. A. Woods & F. E. Sergile (Eds.), *Biogeography of the West Indies: Patterns and Perspectives* (2nd ed., pp. 425–474). Boca Raton, FL: CRC Press.

Lefebvre, L. W., T. J. O'Shea, G. B. Rathbun, and R. C. Best. (1989). Distribution, status, and biogeography of the West Indian manatee. *Biogeography of the West Indies 1989* 567–610.

Lefebvre, L. W., J. P. Reid, W. J. Kenworthy, and J. A. Powell. (2000). Characterizing manatee habitat use and seagrass grazing in Florida and Puerto Rico: Implications for conservation and management. *Pacific Conservation Biology 5* 289–298.

Legault, C. M. (2005). Population viability analysis of Atlantic salmon in Maine, U.S.A. *Transactions of the American Fisheries Society 134* (3): 549–562. DOI:10.1577/t04-017.1

Lehmköster, J. (2015). Climate change impacts on marine ecosystems. In T. Schröder (Ed.), *World Ocean Review* (pp. 102–117). Hamburg, Germany: Mareverlag Publishing House.

Leiter, S. M., K. M. Stone, J. L. Thompson, C. M. Accardo, B. C. Wikgren, M. A. Zani, T. V. N. Cole, R. D. Kenney, C. A. Mayo, and S. D. Kraus. (2017). North Atlantic right whale *Eubalaena glacialis* occurrence in offshore wind energy areas near Massachusetts and Rhode Island, USA. *Endangered Species Research 34* 45–59.

Lennox, R. J., O. H. Diserud, S. J. Cooke, E. B. Thorstad, F. G. Whoriskey, Ø. Solem, T. B. Havn, and I. Uglem. (2016). Influence of gear switching on recapture of Atlantic salmon (*Salmo salar*) in catch-and-release fisheries. *Ecology of Freshwater Fish 25* (3): 422–428. DOI:10.1111/eff.12223

Leon, Y. M. and K. A. Bjorndal. (2002). Selective feeding in the hawksbill turtle, an important predator in coral reef ecosystems. *Marine Ecology Progress Series 245* 249–258.

Leopold, M. F., L. Begeman, J. D. L. van Bleijswijk, L. L. IJsseldijk, H. J. Witte, and A. Grone. (2015). Exposing the grey seal as a major predator of harbor porpoises. *Proceedings of the Royal Society B 282* (1798). DOI:1098/rspb.2014.2429

Lesage, V., K. Gavrilchuk, R. D. Andrews, and R. Sears. (2016). *Wintering areas, fall movements and foraging sites of blue whales satellite-tracked in the western North Atlantic.* Ottawa, Canada: Fisheries and Oceans Canadas.

Lesage, V. and M. O. Hammill. (2001). The status of the grey seal, *Halichoerus grypus*, in the Northwest Atlantic. *Canadian Field-Naturalist 115* (4): 653–662.

Lesser, M. P., J. C. Bythell, R. D. Gates, R. W. Johnstone, and O. Hoegh-Guldberg. (2007). Are infectious diseases really killing corals? Alternative interpretations of the experimental and ecological data. *Journal of Experimental Marine Biology and Ecology 346* (1-2): 36–44. DOI:10.1016/j.jembe.2007.02.015

Lester, L. A. (2012). *Direct and Indirect Effects of Recreational Boats on Diamondback Terrapins (Malaclemys terrapin).* (Unpublished doctoral dissertation). Drexel University, Philadelphia, PA. Retrieved from http://hdl.handle.net/1860/3982.

Lettrich, M. D., M. J. Asaro, D. L. Borggaard, D. M. Dick, R. B. Griffis, J. A. Litz, C. D. Orphanides, D. L. Palka, M. S. Soldevilla, B. Balmer, S. Chavez, D. Cholewiak, D. Claridge, R. Y. Ewing, K. L. Fazioli, D. Fertl, E. M. Fougeres, D. Gannon, L. Garrison, J. Gilbert, A. Gorgone, A. Hohn, S. Horstman, B. Josephson, R. D. Kenney, J. J. Kiszka, K. Maze-Foley, W. McFee, K. D. Mullin, K. Murray, D. E. Pendleton, J. Robbins, J. J. Roberts, G. Rodriguez- Ferrer, E. I. Ronje, P. E. Rosel, T. Speakman, J. E. Stanistreet, T. Stevens, M. Stolen, R. T. Moore, N. L. Vollmer, R. Wells, H. R. Whitehead, and A. Whitt. (2023). Vulnerability to climate change of United States marine mammal stocks in the western North Atlantic, Gulf of Mexico, and Caribbean. *PLoS ONE 18* (9): e0290643. DOI:10.1371/journal.pone.0290643

Levinton, J. S. (2013a). *Marine Biology: Function, Biodiversity, Ecology* (4th ed.). New York, NY: Oxford University Press.

Levinton, J. S. (2013b). Plankton. In *Marine Biology: Function, Biodiversity, Ecology* (4th ed., pp. 141–162). New York, NY: Oxford University Press.

Lewison, R. L., L. B. Crowder, B. P. Wallace, J. E. Moore, T. Cox, R. Zydelis, S. McDonald, A. DiMatteo, D. C. Dunn, C. Y. Kot, R. Bjorkland, S. Kelez, C. Soykan, K. R. Stewart, M. Sims, A. Boustany, A. J. Read, P. Halpin, W. J. Nichols, and C. Safina. (2014). Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. *Proceedings of the National Academy of Sciences of the United States of America 111* (14): 5271–5276.

Lien, J., D. Nelson, and D. J. Hai. (2001). Status of the white-beaked dolphin, *Lagenorhynchus albirostris*, in Canada. *Canadian Field-Naturalist 115* (1): 118–126.

Lindstrom, U., A. Harbitz, T. Haug, and K. T. Nilssen. (1998). Do harp seals, *Phoca groenlandica,* exhibit particular prey preferences? *ICES Journal of Marine Science 55* 941–953.

Lindstrom, U. and T. Haug. (2001). Feeding strategy and prey selectivity in common minke whales (*Balaenoptera acutorostrata*) foraging in the southern Barents Sea during early summer. *Journal of Cetacean Research and Management 3* (3): 239–250.

Linnehan, B. K., F. M. Gomez, S. M. Huston, A. Hsu, R. Takeshita, K. M. Colegrove, C. A. Harms, A. Barratclough, A. C. Deming, T. K. Rowles, W. B. Musser, E. S. Zolman, R. S. Wells, E. D. Jensen, L. H. Schwacke, and C. R. Smith. (2021). Cardiac assessments of bottlenose dolphins (*Tursiops truncatus*) in the Northern Gulf of Mexico following exposure to *Deepwater Horizon* oil. *PLoS ONE 16* (12): e0261112. DOI:10.1371/journal.pone.0261112

Lirman, D. (2000). Fragmentation in the branching coral *Acropora palmata* (Lamarck): Growth, survivorship, and reproduction of colonies and fragments. *Journal of Experimental Marine Biology and Ecology 251* 41–57.

Lirman, D., S. Schopmeyer, D. Manzello, L. J. Gramer, W. F. Precht, F. Muller-Karger, K. Banks, B. Barnes, E. Bartels, A. Bourque, J. Byrne, S. Donahue, J. Duquesnel, L. Fisher, D. Gilliam, J. Hendee, M. Johnson, K. Maxwell, E. McDevitt, J. Monty, D. Rueda, R. Ruzicka, and S. Thanner. (2011). Severe 2010 cold-water event caused unprecedented mortality to corals of the Florida reef tract and reversed previous survivorship patterns. *PLoS ONE 6* (8): e23047. DOI:10.1371/journal.pone.0023047

Litz, J. A. (2007). *Social structure, genetic structure, and persistent organohalogen pollutants in bottlenose dolphins (Tursiops truncatus) in Biscayne Bay, Florida.* (Unpublished doctoral dissertation). University of Miami, Miami, FL.

Liu, Z., P. Chen, Y.-Y. Li, M.-W. Li, Q. Liu, W.-L. Pan, D.-M. Xu, J. Bai, L.-B. Zhang, J. Tang, and P. Shi. (2021). Cochlear hair cells of echolocating bats are immune to intense noise. *Journal of Genetics and Genomics 48* (11): 984-993. DOI:https://doi.org/10.1016/j.jgg.2021.06.007

Lloyd, J. (2015, February 19, 2015). *Seals' Appearance is a Puzzle*. Retrieved May 26, 2017, from http://www.coastalreview.org/2015/02/seals-appearance-in-nc-a-puzzle/.

Lockley, E. C. and C. Eizaguirre. (2021). Effects of global warming on species with temperature-dependent sex determination: Bridging the gap between empirical research and management. *Evolutionary Applications 14* 2361-2377. DOI:10.1111/eva.13226

Loeb, S. C. and E. A. Winters. (2013). Indiana bat summer maternity distribution: effects of current and future climates. *Ecology and Evolution 3* (1): 103-114. DOI:https://doi.org/10.1002/ece3.440

Loring, P. H., P. W. C. Paton, J. D. McLaren, H. Bai, R. Janaswamy, H. F. Goyert, C. R. Griffin, and P. R. Sievert. (2019). *Tracking Offshore Occurrence of Common Terns, Endangered Roseate Terns, and Threatened Piping Plovers*

with VHF Arrays. Hadley, MA.

Lott, C. A. (2006). A new raptor migration monitoring site in the Florida Keys: Counts from 1999—2004. *Journal of Raptor Research 40* (3): 200–209. DOI:10.3356/0892-1016(2006)40[200:anrmms]2.0.co;2

Lough, J. M. and M. J. H. van Oppen. (2009). Coral bleaching: Patterns, processes, causes and consequences. In M. J. H. van Oppen & J. M. Lough (Eds.), *Coral Bleaching* (Vol. Ecological Studies 205). Berlin, Germany: Springer-Verlag.

Luning, K. (1990). *Seaweeds: Their Environment, Biogeography, and Ecophysiology*. New York, NY: John Wiley & Sons, Inc.

Luo, J. and L. Wiegrebe. (2016). Biomechanical control of vocal plasticity in an echolocating bat. *The Journal of Experimental Biology 219* (6): 878–886. DOI:10.1242/jeb.134957

Lutcavage, M. and J. A. Musick. (1985). Aspects of the biology of sea turtles in Virginia. *Copeia 1985* (2): 449–456.

Lydersen, C. and K. M. Kovacs. (1993). Diving behaviour of lactating harp seal, *Phoca groenlandica*, females from the Gulf of St. Lawrence, Canada. *Animal Behaviour 46* 1213–1221.

MacLeod, C., W. F. Perrin, R. Pitman, J. Barlow, L. Ballance, A. D'Amico, T. Gerrodette, G. Joyce, K. D. Mullin, D. L. Palka, and G. T. Waring. (2006). Known and inferred distributions of beaked whale species (family Ziphiidae; Order Cetacea). *Journal of Cetacean Research and Management 7* (3): 271–286.

MacLeod, C. D. (2000). Review of the distribution of *Mesoplodon* species (order Cetacea, family Ziphiidae) in the North Atlantic. *Mammal Review 30* (1): 1–8.

MacLeod, C. D. (2006). How big is a beaked whale? A review of body length and sexual size dimporphism in the family Ziphiidae. *Journal of Cetacean Research and Management 7* (3): 301–308.

MacLeod, C. D., N. Hauser, and H. Peckham. (2004). Diversity, relative density and structure of the cetacean community in summer months east of Great Abaco, Bahamas. *Journal of the Marine Biological Association of the United Kingdom 84* 469–474.

MacLeod, C. D. and G. Mitchell. (2006). Key areas for beaked whales worldwide. *Journal of Cetacean Research and Management 7* (3): 309–322.

MacLeod, C. D., M. B. Santos, and G. J. Pierce. (2003). Review of data on diets of beaked whales: Evidence of niche separation and geographic segregation. *Journal of the Marine Biological Association of the United Kingdom 83* 651–665.

Macreadie, P. I., A. M. Fowler, and D. J. Booth. (2011). Rigs-to-reefs: Will the deep sea benefit from artificial habitat? *Frontiers in Ecology and the Environment 9* (8): 455–461. DOI:10.1890/100112

Madeiros, J. (2009). Cahow update. *Bermuda Audubon Society Newsletter 20* (2): 2.

Madeiros, J., N. Carlile, and D. Priddel. (2012). Breeding biology and population increase of the endangered Bermuda petrel, *Pterodroma cahow*. *Bird Conservation International 22* (1): 35–45. DOI:10.1017/s0959270911000396

Magor, D. M. (1979). *Survey of the Caribbean manatee (Trichechus manatus, L.) on Vieques Island, Puerto Rico*. Washington, DC: U.S. Department of the Navy.

Maine Department of Marine Resources. (2014). *Fishery Management Plan for Rockweed (Ascophyllum nodosum)*. Augusta, ME: Maine Department of Marine Resources.

Maison, K. A., I. K. Kelly, and K. P. Frutchey. (2010). *Green Turtle Nesting Sites and Sea Turtle Legislation throughout Oceania* (National Oceanic and Atmospheric Administration Technical Memorandum NMFS-F/SPO-110). Silver Spring, MD: Scientific Publications Office.

Maldini, D., L. Mazzuca, and S. Atkinson. (2005). Odontocete stranding patterns in the main Hawaiian islands (1937–2002): How do they compare with live animal surveys? *Pacific Science 59* (1): 55–67.

Mansfield, K. L. (2006). *Sources of mortality, movements and behavior of sea turtles in Virginia.* (PhD dissertation). College of William and Mary, Williamsburg, VA.

Mansfield, K. L., M. L. Mendilaharsu, N. F. Putman, M. A. G. dei Marcovaldi, A. E. Sacco, G. Lopez, T. Pires, and Y. Swimmer. (2017). First satellite tracks of South Atlantic sea turtle 'lost years': seasonal variation in trans-equatorial movement. *Proceeding of the Royal Society B 284* 1–9. DOI:10.1098/rspb.2017.1730

Mansfield, K. L., J. Wyneken, and J. Luo. (2021). First Atlantic satellite tracks of 'lost years' green turtles support the importance of the Sargasso Sea as a sea turtle nursery. *Proceedings of the Royal Society B 288* 1-10. DOI:10.1098/rspb.2021.0057

Marcoux, M., H. Whitehead, and L. Rendell. (2007). Sperm whale feeding variations by location, year, social group and clan: Evidence from stable isotopes. *Marine Ecology Progress Series 333* 309–314.

Marhaver, K. L., M. J. A. Vermeij, and M. M. Medina. (2015). Reproductive natural history and successful juvenile propagation of the threatened Caribbean Pillar Coral *Dendrogyra cylindrus*. *BioMed Central Ecology 15* (9): 1–11. DOI:10.1186/s12898-015-0039-7

Marquez, M. R. (1994). *Synopsis of biological data on the Kemp's ridley turtle, Lepidochelys kempii (Garman, 1880)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Márquez, M. R. (1990). *FAO Species Catalogue: Sea Turtles of the World; An Annotated and Illustrated Catalogue of Sea Turtle Species Known to Date*. Rome, Italy: United Nations Environment Programme, Food and Agriculture Organization of the United Nations.

Marsh, H. E. (1989). Mass stranding of dugongs by a tropical cyclone in northern Australia. *Marine Mammal Science 5* (1): 78–84.

Martin, J., Q. Sabatier, T. A. Gowan, C. Giraud, E. Gurarie, C. S. Calleson, J. G. Ortega-Ortiz, C. J. Deutsch, A. Rycyk, and S. M. Koslovsky. (2015). A quantitative framework for investigating risk of deadly collisions between marine wildlife and boats. *Methods in Ecology and Evolution 7* (1): 42–50. DOI:10.1111/2041-210X.12447

Masaki, Y. (1976). Biological studies on the North Pacific sei whale. *Bulletin of the Far Seas Fisheries Research Laboratory 14* 1–104.

Masaki, Y. (1977). The separation of the stock units of sei whales in the North Pacific. *Reports of the International Whaling Commission Special Issue 1* 71–79.

Maslo, B., J. Burger, and S. N. Handel. (2012). Modeling foraging behavior of piping plovers to evaluate habitat restoration success. *The Journal of Wildlife Management 76* (1): 181–188. DOI:10.1002/jwmg.210

Mass.gov. (2024). *The Northern Long-eared Bat*. Retrieved May 30, 2024, from https://www.mass.gov/info-details/the-northern-long-eared-bat.

Mate, B. R., S. L. Nieukirk, and S. D. Kraus. (1997). Satellite-monitored movements of the northern right whale. *The Journal of Wildlife Management 61* (4): 1393–1405.

Mate, B. R., K. M. Stafford, R. Nawojchik, and J. L. Dunn. (1994). Movements and dive behavior of a satellite-monitored Atlantic white-sided dolphin (*Lagenorhynchus acutus*) in the Gulf of Maine. *Marine Mammal Science 10* 116–121.

May, N., R. Parsons, and T. Toft. (2012). *Dissolved Oxygen - Frequency of Hypoxia - 2001 to 2011*. Retrieved from https://www.ncei.noaa.gov/maps/gulf-data-atlas/atlas.htm?plate=Hypoxia.

Mayo, C. A., L. Ganley, C. A. Hudak, S. Brault, M. K. Marx, E. Burke, and M. W. Brown. (2018). Distribution, demography, and behavior of North Atlantic right whales (*Eubalaena glacialis*) in Cape Cod Bay, Massachusetts, 1998–2013. *Marine Mammal Science 34* (4): 979–996. DOI:10.1111/mms.12511

Maze-Foley, K. and K. D. Mullin. (2006). Cetaceans of the oceanic northern Gulf of Mexico: Distributions, group sizes and interspecific associations. *Journal of Cetacean Research and Management 8* (2): 203–213.

Mazzoil, M., D. R. McCulloch, and R. H. Defran. (2005). Observations on the site fidelity of bottlenose dolphins (*Tursiops truncatus*) in the Indian River Lagoon, Florida. *Florida Scientist 68* (4): 217–226.

Mazzotti, F. J. (1983). *The Ecology of Crocodylus acutus in Florida: A Thesis in Ecology.* (Unpublished doctoral dissertation). Pennsylvania State University, University Park, PA.

Mazzotti, F. J. (2014). *American Crocodile Surveys on the Naval Air Station Key West*. Fort Lauderdale, FL: Department of Wildlife Ecology and Conservation Fort Lauderdale Research and Education Center University of Florida.

Mazzotti, F. J., L. A. Brandt, P. Moler, and M. S. Cherkiss. (2007). American crocodile (*Crocodylus acutus*) in Florida: Recommendations for endangered species recovery and ecosystem restoration. *Journal of Herpetology 41* (1): 122–132.

Mazzotti, F. J., M. S. Cherkiss, M. Parry, J. Beauchamp, M. Rochford, B. Smith, K. Hart, and L. A. Brandt. (2016). Large reptiles and cold temperatures: Do extreme cold spells set distributional limits for tropical reptiles in Florida? *Ecosphere 7* (8): 1–9 e01439. DOI:10.1002/ecs2.1439

Mazzotti, F. J. and W. A. Dunson. (1989). Osmoregulation in crocodilians. *American Zoologist 29* 903–920.

Mazzotti, F. J., B. J. Smith, M. A. Squires, M. S. Cherkiss, S. C. Farris, C. Hackett, K. M. Hart, V. Briggs-Gonzalez, and L. A. Brandt. (2019). Influence of salinity on relative density of American crocodiles (*Crocodylus acutus*) in Everglades National Park: Implications for restoration of Everglades ecosystems. *Ecological Indicators 102* 608–616. DOI:https://doi.org/10.1016/j.ecolind.2019.03.002

McAlarney, R., E. Cummings, B. McLellan, and A. Pabst. (2014). *Protected Species Monitoring in the Virginia Capes OPAREA, Cape Hatteras, North Carolina: January 2013–December 2013*. Wilmington, NC: University of North Carolina Wilmington.

McAlarney, R., E. Cummings, W. McLellan, and D. A. Pabst. (2016). *Aerial Surveys for Protected Species in the Cape Hatteras and Norfolk Canyon Regions: 2015 Annual Progress Report*. Virginia Beach, VA: Naval Facilities Engineering Command Atlantic.

McAlarney, R., E. Cummings, W. McLellan, and D. A. Pabst. (2017). *Aerial Surveys for Protected Species in the Cape Hatteras and Norfolk Canyon Regions: 2016 Annual Progress Report*. Virginia Beach, VA: Naval Facilities Engineering Command Atlantic.

McAlarney, R., E. Cummings, W. McLellan, and D. A. Pabst. (2018). *Aerial Surveys for Protected Species in the Cape Hatteras and Norfolk Canyon Regions: 2017 Annual Progress Report*. Virginia Beach, VA: Naval Facilities Engineering Command Atlantic.

McAlpine, D. F. (2002). Pygmy and Dwarf Sperm whales. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (pp. 1007–1009). San Diego, CA: Academic Press.

McAlpine, D. F. (2009). Pygmy and dwarf sperm whales *Kogia breviceps* and *K. sima*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 936–938). Cambridge, MA: Academic Press.

McAlpine, D. F., P. T. Stevick, L. D. Murison, and S. D. Turnbull. (1999). Extralimital records of hooded seals (*Cystophora cristata*) from the Bay of Fundy and northern Gulf of Maine. *Northeastern Naturalist 6* 225–230.

McAlpine, D. F. and R. J. Walker. (1999). Additional extralimital records of the harp seal, *Phoca groenlandica*, from the Bay of Fundy, New Brunswick. *Canadian Field-Naturalist 113* 290–292.

McCormick, S. D., L. P. Hansen, T. P. Quinn, and R. L. Saunders. (1998). Movement, migration, and smolting of Atlantic salmon (*Salmo salar*). *Canadian Journal of Fisheries and Aquatic Sciences 55* 77–92.

McCoshum, S. M., E. L. Pratt, K. C. Lent, and E. M. Boisen. (2023). Literature review of tri-colored bat natural history with implications to management. *Frontiers in Conservation Science 4* 1-14. DOI:10.3389/fcosc.2023.1204901

McCracken, G. F., Y.-F. Lee, E. H. Gillam, W. Frick, and J. Krauel. (2021). Bats Flying at High Altitudes. In B. K. Lim, M. B. Fenton, R. M. Brigham, S. Mistry, A. Kurta, E. H. Gillam, A. Russell, & J. Ortega (Eds.), *50 Years of Bat Research: Foundations and New Frontiers* (pp. 189-205). Springer International Publishing.

McDonald, T. L., B. A. Schroeder, B. A. Stacy, B. P. Wallace, L. A. Starcevich, J. Gorham, M. C. Tumlin, D. Cacela, M. Rissing, D. B. McLamb, E. Ruder, and B. E. Witherington. (2017). Density and exposure of surface-pelagic juvenile sea turtles to *Deepwater Horizon* oil. *Endangered Species Research 33* 69–82. DOI:10.3354/esr00771

McDonnell, L. H., T. L. Jackson, G. H. Burgess, L. Phenix, A. J. Gallagher, H. Albertson, N. Hammerschlag, and J. A. Browder. (2020). Saws and the city: Smalltooth sawfish *Pristis pectinata* encounters, recovery potential, and research priorities in urbanized coastal waters off Miami, Florida, USA. *Endangered Species Research 43* 543–553.

McGinty, N., A. D. Barton, N. R. Record, Z. V. Finkel, D. G. Johns, C. A. Stock, and A. J. Irwin. (2021). Anthropogenic climate change impacts on copepod trait biogeography. *Global Change Biology 27* (7): 1431-1442. DOI:https://doi.org/10.1111/gcb.15499

McGrew, K. A., S. E. Crowell, J. L. Fiely, A. M. Berlin, G. H. Olsen, J. James, H. Hopkins, and C. K. Williams. (2022). Underwater hearing in sea ducks with applications for reducing gillnet bycatch through acoustic deterrence. *The Journal of Experimental Biology 225* (20). DOI:10.1242/jeb.243953

McKenna, M. F., J. Calambokidis, E. M. Oleson, D. W. Laist, and J. A. Goldbogen. (2015). Simultaneous tracking of blue whales and large ships demonstrates limited behavioral responses for avoiding collision. *Endangered Species Research 27* (3): 219–232. DOI:10.3354/esr00666

McKown, M. (2014). *Acoustic monitoring of diablotin activity on Hispaniola: Pilot data and potential next steps*. Santa Cruz, CA: Conservation Metrics.

McLellan, W. (2017). *USWTR Onslow Bay Aerial Survey Data*. Data from University of North Carolina Wilmington, published online by Duke University's Ocean Biodiversity Information System Spatial Ecological Analysis of Megavertebrate Populations. Retrieved from http://seamap.env.duke.edu/dataset/751.

McLellan, W., H. Foley, R. McAlarney, E. Cummings, Z. Swaim, L. Hodge, J. Stanistreet, K. Urian, D. Waples, C. Paxton, D. Pabst, J. Bell, and A. Read. (2014, 28-30 March 2014). *Patterns of cetacean species occurrence, distribution and density at three sites along the continental shelf break of the U.S. Atlantic coast*. Presented at the Southeast and Mid-Atlantic Marine Mammal Symposium. Wilmington, NC.

McLellan, W., R. McAlarney, E. Cummings, J. Bell, A. Read, and D. A. Pabst. (2015). *Year-round Presence of Beaked Whales off Cape Hatteras, North Carolina*. Presented at the 21st Biennial Conference on the Biology of Marine Mammals. San Francisco, CA.

McLellan, W. A., R. J. McAlarney, E. W. Cummings, A. J. Read, C. G. M. Paxton, J. T. Bell, and D. A. Pabst. (2018). Distribution and abundance of beaked whales (Family Ziphiidae) off Cape Hatteras, North Carolina, U.S.A. *Marine Mammal Science 34* (4): 997–1017. DOI:10.1111/mms.12500

Mead, J. (1977). Records of Sei and Bryde's whales from the Atlantic Coast of the United States, the Gulf of Mexico, and the Caribbean. *Reports of the International Whaling Commission 1* 113–116.

Mead, J. G. (1989a). Beaked whales of the genus *Mesoplodon*. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 4, pp. 349–430). San Diego, CA: Academic Press.

Mead, J. G. (1989b). Bottlenose whales: *Hyperoodon ampullatus* (Forster, 1770) and *Hyperoodon planifrons* Flower, 1882. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 4, pp. 321–348). San Diego, CA: Academic Press.

Mead, J. G. and C. W. Potter. (1995). Recognizing two populations of the bottlenose dolphin (*Tursiops truncatus*) off the Atlantic Coast of North America: Morphologic and ecologic considerations. *IBI Reports 5* 31–44.

Measures, L., B. Roberge, and R. Sears. (2004). Stranding of a Pygmy Sperm Whale, *Kogia breviceps,* in the Northern Gulf of St. Lawrence, Canada. *Canadian Field-Naturalist 118* (4): 495–498.

Mengak, L., A. Dayer, B. Longenecker, and C. Spiegel. (2019). *Guidance and Best Practices for Evaluating and Managing Human Disturbances to Migrating Shorebirds on*

Coastal Lands in the Northeastern United States. U.S. Fish and Wildlife Service.

Merkel, F. R. and K. L. Johansen. (2011). Light-induced bird strikes on vessels in Southwest Greenland. *Marine Pollution Bulletin 62* (11): 2330–2336.

Metz, T. and A. Landry. (2016). Trends in Kemp's ridley sea turtle (*Lepidochelys kempii*) relative abundance, distribution, and size composition in nearshore waters of the northwestern Gulf of Mexico. *Gulf of Mexico Science 33* (2): 179–191. DOI:10.18785/goms.3302.05

Meyer-Gutbrod, E. L., K. T. A. Davies, C. L. Johnson, S. Plourde, K. A. Sorochan, R. D. Kenney, C. Ramp, J.-F. Gosselin, J. W. Lawson, and C. H. Greene. (2023). Redefining North Atlantic right whale habitat-use patterns under climate change. *Limnology and Oceanography 68* (S1): S71–S86. DOI:https://doi.org/10.1002/lno.12242

Meyer-Gutbrod, E. L., C. H. Greene, K. T. A. Davies, and D. G. Johns. (2021). Ocean Regime Shift is Driving Collapse of the North Atlantic Right Whale Population. *Oceanography 34* (3): 22–31.

Meyer, K. D., S. M. McGehee, and M. W. Collopy. (2004). Food deliveries at swallow-tailed kite nests in southern Florida. *Condor 106* (1): 171–176. DOI:10.1650/7339

Meylan, A. (1988). Spongivory in hawksbill turtles: A diet of glass. *Science 239* (4838): 393–395.

Meylan, A. (1995). Sea turtle migration—Evidence from tag returns. In K. A. Bjorndal (Ed.), *Biology and Conservation of Sea Turtles* (Revised ed., pp. 91–100). Washington, DC: Smithsonian Institution Press.

Meylan, A. B., B. E. Witherington, B. Brost, R. Rivero, and P. S. Kubilis. (2006). Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of *Caretta, Chelonia*, and *Dermochelys*. In M. Frick, A. Panagopoulou, A. F. Rees, & K. Williams (Eds.), *Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation: Book of Abstracts* (pp. 306–307). Athens, Greece: National Marine Fisheries Service Southeast Fisheries Science Center, International Sea Turtle Society.

Meylan, P. A., R. F. Hardy, J. A. Gray, and A. B. Meylan. (2022). A half-century of demographic changes in a green turtle (*Chelonia mydas*) foraging aggregation during an era of seagrass decline. *Marine Biology 169* (6): 1–20. DOI:10.1007/s00227-022-04056-5

Meynier, L., C. Pusineri, J. Spitz, M. B. Santos, G. J. Pierce, and V. Ridoux. (2008). Intraspecific dietary variation in the short-beaked common dolphin *Delphinus delphis* in the Bay of Biscay: Importance of fat fish. *Marine Ecology Progress Series 354* 277–287.

Michel, J. M., R. Greer, M. Hoffman, P. McGowan, and R. Wood. (2001). *Acute Mortality of Diamondback Terrapins from the Chalk Point Oil Spill*. Silver Spring, MD: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Damage Assessment, Remediation, and Restoration Program.

Mid-Atlantic Regional Council On The Ocean. (2015). *Assessment of Marine Debris in the Mid-Atlantic*. Williamsburg, VA: Mid-Atlantic Regional Council On The Ocean.

Mignucci-Giannoni, A. A. (1988). A stranded sperm whale, *Physeter catodon*, at Cayo Santiago, Puerto Rico. *Caribbean Journal of Science 24* (3–4): 173–190.

Mignucci-Giannoni, A. A. (1998). Zoogeography of cetaceans off Puerto Rico and the Virgin Islands. *Caribbean Journal of Science 34* (3–4): 173–190.

Mignucci-Giannoni, A. A. and C. A. Beck. (1998). The diet of the manatee (*Trichechus manatus*) in Puerto Rico. *Marine Mammal Science 14* (2): 394–397.

Mignucci-Giannoni, A. A. and D. K. Odell. (2001). Tropical and subtropical records of hooded seals (*Cystophora cristata*) dispel the myth of extant Caribbean monk seals (*Monachus tropicalis*). *Bulletin of Marine Science 68* (1): 47–58.

Mignucci-Giannoni, A. A., B. Pinto-Rodríguez, M. Velasco-Escudero, R. A. Montoya-Ospina, N. M. Jiménez-Marrero, M. A. Rodríguez-López, E. H. Williams, Jr., and D. K. Odell. (1999). Cetacean strandings in Puerto Rico and the Virgin Islands. *Journal of Cetacean Research and Management 1* (2): 191–198.

Mignucci-Giannoni, A. A., S. L. Swartz, A. Martinez, C. M. Burks, and W. A. Watkins. (2003). First records of the pantropical spotted dolphin (*Stenella attenuata*) for the Puerto Rican Bank, with a review of the species for the Caribbean. *Caribbean Journal of Science 39* (3): 381–392.

Miller, L. J., A. D. Mackey, T. Hoffland, and S. A. Kuczaj, II. (2010). Potential effects of a major hurricane on Atlantic bottlenose dolphin (*Tursiops truncatus*) reproduction in the Mississippi Sound. *Marine Mammal Science 26* (3): 707–715. DOI:10.1111/j.1748-7692.2010.00371

Miller, M., A. S. Bourque, and J. A. Bohnsack. (2002). An analysis of the loss of acroporid corals at Looe Key, Florida, U.S.A.: 1983–2000. *Coral Reefs 21* (2): 179–182. DOI:10.1007/s00338-002-0228-7

Miller, M. H., J. Carlson, P. Cooper, D. Kobayashi, M. Nammack, and J. Wilson. (2013). *Status Review Report: Scalloped Hammerhead Shark (Sphyrna lewini)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Miller, M. H., J. Carlson, P. Cooper, D. Kobayashi, M. Nammack, and J. Wilson. (2014). *Status Review Report: Scalloped Hammerhead Shark (Sphyrna lewini)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Miller, M. H. and C. Klimovich. (2016). *Endangered Species Act Status Review Report: Giant Manta Ray (Manta birostris) and Reef Manta Ray (Manta alfredi).* Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Milligan, R. J., K. J. Morris, B. J. Bett, J. M. Durden, D. O. B. Jones, K. Robert, H. A. Ruhl, and D. M. Bailey. (2016). High resolution study of the spatial distributions of abyssal fishes by autonomous underwater vehicle. *Scientific Reports 6* (1): 26095. DOI:10.1038/srep26095

Mitchell, J. E. D. (1991). Winter Records of the Minke Whale (*Balaenoptera acutorostrata acutorostrata* Lacepede 1804) in the Southern North Atlantic. *Report of the International Whaling Commision 41* 455–457.

Mitchelmore, C. L., C. A. Bishop, and T. K. Collier. (2017). Toxicological estimation of mortality of oceanic sea turtles oiled during the Deepwater Horizon oil spill. *Endangered Species Research 33* 39–50.

Mitsch, W. J., J. G. Gosselink, C. J. Anderson, and L. Zhang. (2009). *Wetland Ecosystems*. Hoboken, NJ: John Wiley & Sons, Inc.

Miyazaki, N. and W. F. Perrin. (1994). Rough-toothed dolphin, *Steno bredanensis* (Lesson, 1828). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 1–21). San Diego, CA: Academic Press.

Montero, N., P. S. Tomillo, V. S. Saba, M. A. G. dei Marcovaldi, M. López-Mendilaharsu, A. S. Santos, and M. M. P. B. Fuentes. (2019). Effects of local climate on loggerhead hatchling production in Brazil: Implications from climate change. *Scientific Reports 9* (1): 1–12. DOI:10.1038/s41598-019-45366-x

Mooney, T. A., A. Smith, O. N. Larsen, K. A. Hansen, M. Wahlberg, and M. H. Rasmussen. (2019). Field-based hearing measurements of two seabird species. *Journal of Experimental Biology 222* 1–7. DOI:DOI:10.1242/jeb.190710

Moore, C. J. (2008). Synthetic polymers in the marine environment: A rapidly increasing, long-term threat. *Environmental Research 108* (2): 131–139. DOI:10.1016/j.envres.1008.07.025

Moore, J. C. and E. Clark. (1963). Discovery of right whales in the Gulf of Mexico. *Science 141* (3577): 269.

Mora, C., D. P. Tittensor, S. Adl, A. G. Simpson, and B. Worm. (2011). How many species are there on Earth and in the ocean? *PLoS Biology 9* (8): 1–8. DOI:10.1371/journal.pbio.1001127

Morano, J. L., D. P. Salisbury, A. N. Rice, K. L. Conklin, K. L. Falk, and C. W. Clark. (2012). Seasonal and geographical patterns of fin whale song in the western North Atlantic Ocean. *The Journal of the Acoustical Society of America 132* (2): 1207–1212. DOI:10.1121/1.4730890

Moreno, I. B., A. N. Zerbini, D. Danilewicz, M. C. de Oliveira Santos, P. C. Simoes-Lopes, J. Lailson-Brito, Jr., and A. F. Azevedo. (2005). Distribution and habitat characteristics of dolphins of the genus *Stenella* (Cetacea: Delphinidae) in the southwest Atlantic Ocean. *Marine Ecology Progress Series 300* 229–240.

Morissette, L., M. O. Hammill, and C. Savenkoff. (2006). The trophic level of marine mammals in the northern Gulf of St. Lawrence. *Marine Mammal Science 22* (1): 74–103.

Morreale, S. J. and E. A. Standora. (1998). *Early life stage ecology of sea turtles in northeastern U.S. waters*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Morris, A. V., C. M. Roberts, and J. P. Hawkins. (2000). The threatened status of groupers (Epinephelinae). *Biodiversity & Conservation 9* (7): 919–942. DOI:10.1023/a:1008996002822

Morris, R. L., R. Hale, E. M. A. Strain, S. E. Reeves, A. Vergés, E. M. Marzinelli, C. Layton, V. Shelamoff, T. D. J. Graham, M. Chevalier, and S. E. Swearer. (2020). Key principles for managing recovery of kelp forests through restoration. *BioScience 70* (8): 688–698. DOI:10.1093/biosci/biaa058

Mortimer, J. A. (1995). Feeding ecology of sea turtles. In K. A. Bjorndal (Ed.), *Biology and Conservation of Sea Turtles* (Revised ed., pp. 103–109). Washington, DC: Smithsonian Institution Press.

Mortimer, J. A. and M. Donnelly. (2008). *Hawksbill Turtle (Eretmochelys imbricate): Marine Turtle Specialist Group 2008 International Union for Conservation of Nature Red List Status Assessment*. Retrieved from http://www.iucnredlist.org/details/8005/0.

Moyle, P. B. and J. J. Cech, Jr. (2004). *Fishes: An Introduction to Ichthyology* (5th ed.). London, United Kingdom: Pearson Educational, Inc.

Mrosovsky, N., G. D. Ryan, and M. C. James. (2009). Leatherback turtles: The menace of plastic. *Marine Pollution Bulletin 58* (2): 287–289. DOI:10.1016/j.marpolbul.2008.10.018

Muller, E. M. and R. van Woesik. (2012). Caribbean coral diseases: primary transmission or secondary infection? *Global Change Biology 18* (12): 3529-3535. DOI:https://doi.org/10.1111/gcb.12019

Mullin, K. D. (2007). *Abundance of cetaceans in the oceanic northern Gulf of Mexico from 2003 and 2004 ship surveys*. Pascagoula, MS: Southeast Fisheries Science Center, National Marine Fisheries Service.

Mullin, K. D. and G. L. Fulling. (2003). Abundance of cetaceans in the southern U.S. North Atlantic Ocean during summer 1998. *Fishery Bulletin 101* (3): 603–613.

Mullin, K. D. and G. L. Fulling. (2004). Abundance of cetaceans in the oceanic northern Gulf of Mexico, 1996–2001. *Marine Mammal Science 20* (4): 787–807.

Mullin, K. D., L. V. Higgins, T. A. Jefferson, and L. J. Hansen. (1994a). Sightings of the Clymene dolphin (*Stenella clymene*) in the Gulf of Mexico. *Marine Mammal Science 10* (4): 464–470.

Mullin, K. D. and W. Hoggard. (2000). *Visual surveys of cetaceans and sea turtles from aircraft and ships* (Cetaceans, sea turtles and seabirds in the northern Gulf of Mexico: Distribution, abundance and habitat associations). New Orleans, LA: Minerals Management Service.

Mullin, K. D., W. Hoggard, and L. J. Hansen. (2004). Abundance and seasonal occurrence of cetaceans in outer continental shelf and slope waters of the north-central and northwestern Gulf of Mexico. *Gulf of Mexico Science 22* (1): 62–73.

Mullin, K. D., W. Hoggard, C. L. Roden, R. R. Lohoefener, and C. M. Rogers. (1994b). Cetaceans on the upper continental slope in the north-central Gulf of Mexico. *Fishery Bulletin 92* (4): 773–786.

Mullin, K. D., T. A. Jefferson, L. J. Hansen, and W. Hoggard. (1994c). First sightings of melon-headed whales (*Peponocephala electra*) in the Gulf of Mexico. *Marine Mammal Science 10* (3): 342–348.

Mumby, P. J., A. R. Harborne, and D. R. Brumbaugh. (2011). Grouper as a natural biocontrol of invasive lionfish. *PLoS ONE 6* (6): e21510. DOI:10.1371/journal.pone.0021510

Murphy, R. C. and L. Mowbray. (1951). New light on the Cahow, *Pterodrama cahow*. *The Auk 68* (3): 266–280.

Murray, M. (2011, August 21). Previously believed loggerhead was actually a green turtle. *The News Journal*. Retrieved from http://www.delawareonline.com/.

Musick, J. A. and C. J. Limpus. (1997). Habitat utilization and migration of juvenile sea turtles. In P. L. Lutz & J. A. Musick (Eds.), *The Biology of Sea Turtles* (pp. 137–163). Boca Raton, FL: CRC Press.

Mussi, B., A. Miragliuolo, T. De Pippo, M. C. Gambi, and D. Chiota. (2004). The submarine canyon of Cuma (southern Tyrrhenian Sea, Italy), a cetacean key area to protect. *European Research on Cetaceans 15* 178–179.

Nagaoka, S., A. Martins, R. dos Santos, M. Tognella, E. de Oliveira Filho, and J. Seminoff. (2012). Diet of juvenile green turtles (*Chelonia mydas*) associating with artisanal fishing traps in a subtropical estuary in Brazil. *Marine Biology 159* (3): 573–581. DOI:10.1007/s00227-011-1836-y

Nance, J. M., C. W. Caillouet, Jr., and R. A. Hart. (2012). Size-composition of annual landings in the white shrimp, *Litopenaeus setiferus*, fishery of the northern Gulf of Mexico, 1960–2006: Its trends and relationships with other fishery-dependent variables. *Marine Fisheries Review 72* (2): 1–13.

Narazaki, T., K. Sato, K. J. Abernathy, G. J. Marshall, and N. Miyazaki. (2013). Loggerhead turtles (*Caretta caretta*) use vision to forage on gelatinous prey in mid-water. *PLoS ONE 8* (6): e66043. DOI:10.1371/journal.pone.0066043

National Aquarium. (2024). *Diamondback Terrapin (Malaclemys terrapin)*. Retrieved August 13, 2024, from https://aqua.org/explore/animals/diamondback-terrapin.

National Audubon Society. (2005). *Bermuda Petrel, Pterodroma cahow*. *Bird Conservation, Waterbird Conservation, Waterbird Species.* Retrieved 2010, from http://web1.audubon.org/waterbirds/species.php?speciesCode=berpet.

National Audubon Society. (2015, September). *Important Bird Areas Program: A Global Currency for Bird Conservation*. Retrieved May 15, 2017, from http://web4.audubon.org/bird/iba/.

National Audubon Society. (2017). *Guide to North American Birds: Roseate Tern*. Retrieved May 11, 2017, from http://www.audubon.org/field-guide/bird/roseate-tern.

National Geographic. (2024). *Diamondback terrapin, facts and photos*. Retrieved January 15, 2024, from https://www.nationalgeographic.com/animals/reptiles/facts/diamondback-terrapins.

National Marine Fisheries Service. (1998). *Final Recovery Plan for the Shortnose Sturgeon Acipenser brevirostrum*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2005). *Recovery Plan for the North Atlantic Right Whale (Eubalaena glacialis), Revision*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2007a). *Report to Congress on the Impact of Hurricanes Katrina, Rita, and Wilman on Commercial and Recreational Fisheries Habitat of Alabama, Florida, Louisiana, Mississippi, and Texas*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2007b). *Status Review of Atlantic sturgeon (Acipenser oxyrinchus oxyrinchus)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2009a). *Recovery Plan for the Smalltooth Sawfish (Pristis pectinata)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2009b). *Sperm Whale (Physeter macrocephalus): 5-Year Review: Summary and Evaluation*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2010a). Endangered and Threatened Species; Proposed Listing of Nine Distinct Population Segments of Loggerhead Sea Turtles as Endangered or Threatened. *Federal Register 75* (50): 12598–12656.

National Marine Fisheries Service. (2010b). *Final Recovery Plan for the Fin Whale (Balaenoptera physalus)*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service. (2010c). *Final Recovery Plan for the Sperm Whale (Physeter macrocephalus)*. Silver Spring, MD.

National Marine Fisheries Service. (2010d). *Smalltooth Sawfish (Pristis pectinata)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2011a). *A Decade of Support to Save and Conserve Stranded Marine Mammals*.

National Marine Fisheries Service. (2011b). *Final Recovery Plan for the Sei Whale (Balaenoptera borealis)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2011c). *Sea Turtles and the Gulf of Mexico Oil Spill*. Retrieved from http://www.nmfs.noaa.gov/pr/health/oilspill/turtles.htm.

National Marine Fisheries Service. (2012). *Management Report for 82 Corals Status Review under the Endangered Species Act: Existing Regulatory Mechanisms and Conservation Efforts*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Regional Office.

National Marine Fisheries Service. (2013a). *2013 Annual Report of a Comprehensive Assessment of Marine Mammal, Marine Turtle, and Seabird Abundance and Spatial Distribution in US Waters of the Western North Atlantic Ocean*. Woods Hole, MA and Miami, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center and Southeast Fisheries Science Center.

National Marine Fisheries Service. (2013b). *Biological Opinion and Conference Opinion on Atlantic Fleet Training and Testing Activities (2013–2018)*. Silver Spring, MD: Office of Protected Resources.

National Marine Fisheries Service. (2013c). *Nassau Grouper, Epinephelus striatus (Bloch 1792) Biological Report*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service. (2014). *Sea Turtles, Dolphins, and Whales - 10 years after the Deepwater Horizon Oil Spill*. Retrieved from https://www.fisheries.noaa.gov/national/marine-life-distress/deepwater-horizon-oil-spill-2010-sea-turtles-dolphins-and-whales.

National Marine Fisheries Service. (2015a). *Nassau Grouper, Epinephelus striatus (Bloch 1792) Biological Report*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service. (2015b). *Recovery Plan. Elkhorn Coral (Acropora palmata) and Staghorn Coral (A. cervicornis)*. St. Petersburg, FL: Southeast Regional Office.

National Marine Fisheries Service. (2015c). *Sperm Whale (Physeter macrocephalus) 5-Year Review: Summary and Evaluation, June 2015*. Silver Spring, MD: Office of Protected Resources.

National Marine Fisheries Service. (2016a). *Guidelines for Preparing Stock Assessment Reports Pursuant to Section 117 of the Marine Mammal Protection Act*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2016b). *Species in the Spotlight: Priority Actions: 2016–2020, Atlantic Salmon, Salmo salar* (Atlantic Salmon 5-Year Action Plan). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service. (2017). *FAQs on the 2016–2017 Humpback Whale Atlantic Coast Unusual Mortalit Event*. Retrieved May 30, 2017, from https://www.fisheries.noaa.gov/insight/frequent-questions-2016-2018-humpback-whale-atlantic-coast-unusual-mortality-event.

National Marine Fisheries Service. (2018a). *Recovery Outline, Atlantic Sturgeon, Gulf of Maine, New York Bight, Chesapeake Bay, and South Atlantic Distinct Population Segments*. Silver Spring, MD: NOAA Fisheries Protected Resources.

National Marine Fisheries Service. (2018b). *Smalltooth Sawfish (Pristis pectinata) 5-Year Review: Summary and Evaluation of United States Distinct Population Segment of Smalltooth Sawfish*. St. Petersburg, FL: Southeast Regional Office.

National Marine Fisheries Service. (2020a, 11/10/2020). *Atlantic Sturgeon Population Estimates*. Retrieved 01/05/2023, from https://www.fisheries.noaa.gov/national/endangered-species-conservation/atlantic-sturgeon-population-estimates.

National Marine Fisheries Service. (2020b). *Recovery Plan for the Blue Whale (Balaenoptera musculus) - First Revision*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2020c). *Scalloped Hammerhead Shark (Sphyrna lewini) 5-Year Review: Summary and Evaluation*. Silver Spring, MD: National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service. (2022a). *2018–2020 Pinniped Unusual Mortality Event Along the Northeast Coast*. Silver Spring, MD: National Oceanic and Atmospheric Administration Fisheries.

National Marine Fisheries Service. (2022b). *Atlantic Sturgeon*. Retrieved January 6, 2023, from https://www.fisheries.noaa.gov/species/atlantic-sturgeon.

National Marine Fisheries Service. (2022c). *Chesapeake Bay Distinct Population Segment of Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus) 5-Year Review: Summary and Evaluation*. Gloucester, MA: Greater Atlantic Regional Fisheries Office.

National Marine Fisheries Service. (2022d). *Giant Manta Ray*. Retrieved January 18, 2023, from https://www.fisheries.noaa.gov/species/giant-manta-ray.

National Marine Fisheries Service. (2022e). *Gulf of Maine Distinct Population Segment of Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus) 5-Year Review: Summary and Evaluation*. Gloucester, MA: Greater Atlantic Regional Fisheries Office.

National Marine Fisheries Service. (2022f). *Gulf Sturgeon Acipenser oxyrinchus desotoi*. Retrieved June 6, 2023, from https://www.fisheries.noaa.gov/species/gulf-sturgeon.

National Marine Fisheries Service. (2022g). *Nassau Grouper Epinephelus striatus*. Retrieved January 10, 2023, from https://www.fisheries.noaa.gov/species/nassau-grouper.

National Marine Fisheries Service. (2022h). *New York Bight Distinct Population Segment of Atlantic Sturgeon (Acipenser oxyrinchus oxyrinchus) 5-Year Review: Summary and Evaluation*. Gloucester, MA: Greater Atlantic Regional Fisheries Office.

National Marine Fisheries Service. (2022i). *Oceanic Whitetip Shark Carcharhinus longimanus*. Retrieved January 18, 2023, from https://www.fisheries.noaa.gov/species/oceanic-whitetip-shark.

National Marine Fisheries Service. (2022j). *Staghorn coral (Acropora cervicornis), Elkhorn coral (Acropora palmata), Lobed star coral (Orbicella annularis), Mountainous star coral (Orbicella faveolata), Boulder star coral (Orbicella franksi), Rough cactus coral (Mycetophyllia ferox), Pillar coral (Dendrogyra cylindrus) 5-Year Review: Summary and Evaluation*. St. Petersburg, FL: Southeast Regional Office.

National Marine Fisheries Service. (2023a). *Atlantic Salmon (Protected): In the Spotlight*. Retrieved October 12, 2023, from https://www.fisheries.noaa.gov/species/atlantic-salmon-protected/spotlight.

National Marine Fisheries Service. (2023b). *Elkhorn Coral (Acropora palmata)*. Retrieved July 24, 2023, from https://www.fisheries.noaa.gov/species/elkhorn-coral.

National Marine Fisheries Service. (2023c). *Shortnose Sturgeon*. Retrieved June 7, 2023, from https://www.fisheries.noaa.gov/species/shortnose-sturgeon.

National Marine Fisheries Service. (2024a). *2013–2015 Bottlenose Dolphin Unusual Mortality Event in the Mid-Atlantic (Closed)*. Retrieved August 1, 2024, from https://www.fisheries.noaa.gov/national/marine-life-distress/2013-2015-bottlenose-dolphin-unusual-mortality-event-mid-atlantic.

National Marine Fisheries Service. (2024b). *2016–2024 Humpback Whale Unusual Mortality Event Along the Atlantic Coast*. Retrieved June 20, 2024, from https://www.fisheries.noaa.gov/national/marine-life-distress/2016-2024-humpback-whale-unusual-mortality-event-along-atlantic-coast.

National Marine Fisheries Service. (2024c). *2017–2024 Minke Whale Unusual Mortality Event along the Atlantic Coast*. Retrieved August 3, 2024, from https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2024-minke-whale-unusual-mortality-event-along-atlantic-coast#minke-whale-strandings.

National Marine Fisheries Service. (2024d). *2017–2024 North Atlantic Right Whale Unusual Mortality Event*. Retrieved June 20, 2024, from https://www.fisheries.noaa.gov/national/marine-life-distress/2017-2024-north-atlantic-right-whale-unusual-mortality-event.

National Marine Fisheries Service. (2024e). *2022 Pinniped Unusual Mortality Event in Maine (CLOSED)*. Retrieved August 3, 2024, from https://www.fisheries.noaa.gov/marine-life-distress/2022-pinniped-unusual-mortality-event-maine-closed.

National Marine Fisheries Service. (2024f). *North Atlantic Right Whale Calving Season 2024*. Retrieved August 1, 2024, from https://www.fisheries.noaa.gov/national/endangered-species-conservation/north-atlantic-right-whale-calving-season-2024.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (1991). *Recovery Plan for U.S. Populations of Atlantic Green Turtle (Chelonia mydas)*. Washington, DC: National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (1993). *Recovery Plan for the Hawksbill Turtle (Eretmochelys imbricata) in the U.S. Caribbean, Atlantic and Gulf of Mexico*. Atlanta, GA and Washington, DC: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Region; and U.S. Fish and Wildlife Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (1998). *Recovery Plan for U.S. Pacific Populations of the Hawksbill Turtle (Eretmochelys imbricata)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2005). *Final Recovery Plan for the Gulf of Maine Distinct Population Segment of Atlantic Salmon (Salmo salar)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2007a). *Hawksbill Sea Turtle (Eretmochelys imbricata) 5-year Review: Summary and Evaluation*. Silver Spring, MD: National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2007b). *Loggerhead Sea Turtle (Caretta caretta) 5-year review: Summary and Evaluation*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2008). *Recovery Plan for the Northwest Atlantic Population of the Loggerhead Sea Turtle (Caretta caretta), Second Revision*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2011). *Bi-National Recovery Plan for the Kemp's Ridley Sea Turtle (Lepidochelys kempii)*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2013). *Leatherback Turtle (Dermochelys coriacea) 5-Year Review: Summary and Evaluation*. Silver Spring, MD and Jacksonville, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources; and U.S. Fish and Wildlife Service, Southeast Region, Jacksonville Ecological Services Field Office.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2015). *Kemp's Ridley Sea Turtle (Lepidochelys kempii) 5-Year Review: Summary and Evaluation*. Silver Spring, MD and Albuquerque, NM: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Office of Protected Resources; and U.S. Fish and Wildlife Service, Southwest Region.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2020a). *Atlantic salmon (Salmo salar) 5-Year Review: Summary and Evaluation*. Washington, DC: U.S. Department of Commerce and U.S. Department of the Interior.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2020b). *Endangered Species Act status review of the leatherback turtle (Dermochelys coriacea)* (Report to the National Marine Fisheries Service Office of Protected Resources and U.S. Fish and Wildlife Service). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service; and the U.S. Fish and Wildlife Service.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (2023). *Loggerhead Sea Turtle (Caretta caretta) Northwest Atlantic Ocean DPS 5-Year Review: Summary and Evaluation*. Silver Spring, MD and Jacksonville, FL: National Marine Fisheries Service, Office of Protected Resources, and U.S. Fish and Wildlife Service, Southeast Region, Florida Ecological Services Office.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. (1992). *Recovery Plan for Leatherback Turtles (Dermochelys coriacea) in the U.S. Caribbean, Atlantic and Gulf of Mexico*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

National Ocean Service. (2016, September 2). *Where Are Reef Building Corals Found*. Retrieved March 20, 2017, from http://oceanservice.noaa.gov/education/tutorial_corals/coral05_distribution.html.

National Oceanic and Atmospheric Administration. (2015). *2015 Annual Report to a Comprehensive Assessment of Marine Mammal, Marine Turtle, and Seabird Abundance and Spatial Distribution in U.S. Waters of the Western North Atlantic Ocean–AMAPPS II*. Woods Hole, MA and Miami, FL: Northeast Fisheries Science Center and Southeast Fisheries Science Center.

National Oceanic and Atmospheric Administration. (2016a, September). *Investigation of East Flower Garden Bank coral die-off continues amid new coral bleaching event*. Retrieved March 30, 2017, from http://sanctuaries.noaa.gov/news/sep16/investigation-of-coral-die-off-continues-amid-bleaching-event.html.

National Oceanic and Atmospheric Administration. (2016b, July). *NOAA scientists report mass die-off of invertebrates at East Flower Garden Bank in Gulf of Mexico*. Retrieved March 30, 2017, from http://sanctuaries.noaa.gov/news/jul16/noaa-scientists-report-mass-die-off-of-invertebrates-at-east-flower-garden-bank.html.

National Oceanic and Atmospheric Administration. (2022). *Cetacean Data Availability*. Retrieved from https://cetsound.noaa.gov/cda.

National Oceanic and Atmospheric Administration. (2023). *About Pulley Ridge*. Retrieved May 26, 2023, from https://oceanexplorer.noaa.gov/explorations/13pulleyridge/background/aboutpr/aboutpr.html.

National Park Service. (2012). *Everglades National Park, American Alligator: Species Profile*. Retrieved from http://www.nps.gov/ever/naturescience/alligator.htm.

National Park Service. (2017). *Hibernate or Migrate*. Retrieved February 2, 2017, from https://www.nps.gov/subjects/bats/hibernate-or-migrate.htm.

National Park Service. (2023, June 4). *Satellite Tracking Kemp's Ridley, Loggerhead, and Green Sea Turtles*. Retrieved August 23, 2023, from https://www.nps.gov/pais/learn/nature/tracking.htm.

NatureServe. (2023). *Perimyotis subflavus Tricolored Bat*. Retrieved August 22, 2023, from https://explorer.natureserve.org/Taxon/ELEMENT_GLOBAL.2.102580/Perimyotis_subflavus.

Naval Undersea Warfare Center Division Newport. (2023). *Sea Turtle Distribution and Abundance on the East Coast of the United States*. Newport, RI: U.S. Department of the Navy, Naval Undersea Warfare Center Division Newport.

Neilson, J. L., C. M. Gabriele, A. S. Jensen, K. Jackson, and J. M. Straley. (2012). Summary of Reported Whale-Vessel Collisions in Alaskan Waters. *Journal of Marine Biology 2012* 1–18. DOI:10.1155/2012/106282

Nemoto, T. and A. Kawamura. (1977). Characteristics of food habits and distribution of baleen whales with special reference to the abundance of North Pacific sei and Bryde's whales. *Reports of the International Whaling Commission Special Issue 1* 80–87.

Newton, I. (2007). Weather-related mass-mortality events in migrants. *IBIS 149* (3): 453–467. DOI:10.1111/j.1474-919X.2007.00704

Nifong, J. C. and B. Silliman. (2017). Abiotic factors influence the dynamics of marine habitat use by a highly mobile “freshwater” top predator. *Hydrobiologia 802* (1): 155–174.

Niles, L. J., H. P. Sitters, A. D. Dey, P. W. Atkinson, A. J. Baker, K. A. Bennett, R. Carmona, K. E. Clark, N. A. Clark, C. Espoz, P. M. González, B. A. Harrington, D. E. Hernández, K. S. Kalasz, R. G. Lathrop, R. N. Matus, C. D. T. Minton, R. I. G. Morrison, M. K. Peck, W. Pitts, R. A. Robinson, and I. L. Serrano. (2008). *Status of the Red Knot (Calidris canutus rufa) in the Western Hemisphere* (Studies in Avian Biology). Boise, ID: Cooper Ornithological Society.

Nilsen, F. M., J. A. Bowden, T. R. Rainwater, A. M. Brunell, B. L. Kassim, P. M. Wilkinson, L. J. Guillette Jr., S. E. Long, and T. B. Schock. (2019). Examining toxic trace element exposure in American alligators. *Environment International 128* 324-334. DOI:https://doi.org/10.1016/j.envint.2019.04.035

Nisbet, I. C. T. and J. A. Spendelow. (1999). Contribution of research to management and recovery of the roseate tern: Review of a twelve-year project. *Waterbirds: The International Journal of Waterbird Biology 22* (2): 239–252.

Nordstrom, B., M. C. James, and B. Worm. (2020). Jellyfish distribution in space and time predicts leatherback sea turtle hot spots in the Northwest Atlantic. *PLoS ONE 15* (5): 1–21. DOI:10.1371/journal.pone.0232628

Norris, T. F., J. Oswald, T. Yack, E. Ferguson, C. Hom-Weaver, K. Dunleavy, S. Coates, and T. Dominello. (2012). *An Analysis of Acoustic Data from the Mariana Islands Sea Turtle and Cetacean Survey (MISTCS)*. Encinitas, CA: Bio-Waves, Inc.

North American Bird Conservation Initiative and U.S. Committee. (2010). *The State of the Birds: 2010 Report on Climate Change, United States of America*. Washington, DC: U.S. Department of the Interior.

North American Bird Conservation Initiative U.S. Committee. (2009). *The State of the Birds, United States of America, 2009*. Washington, DC: U.S. Department of Interior.

Northwest Atlantic Leatherback Working Group. (2018). *Northwest Atlantic Leatherback Turtle (Dermochelys coriacea) Status Assessment* (WIDECAST Technical Report No. 16). Basseterre, St. Kitts: Conservation Science Partners and the Wider Caribbean Sea Turtle Conservation Network (WIDECAST).

O'Brien, M., R. Crossley, and K. Karlson. (2006). Piping plover: *Charadrius melodus*. In *The Shorebird Guide* (pp. 54–56, 335–337). New York, NY: Houghton Mifflin Company.

O'Hern, J. E. and D. C. Biggs. (2009). Sperm whale (*Physeter macrocephalus*) habitat in the Gulf of Mexico: Satellite observed ocean color and altimetry applied to small-scale variability in distribution. *Aquatic Mammals 35* (3): 358–366. DOI:10.1578/AM.35.3.2009.358

O’Sullivan, S. and K. D. Mullin. (1997). Killer whales (*Orcinus orca*) in the Northern Gulf of Mexico. *Marine Mammal Science 13* (1): 141–147.

Ocean Biodiversity Information System – Spatial Ecological Analysis of Megavertebrate Populations. (2024). *Datasets contributed through US Navy Marine Species Monitoring Program*. Retrieved from https://seamap.env.duke.edu/partner/NAVY#anchor-north_atlantic.

Odell, D. K. and K. M. McClune. (1999). False killer whale—*Pseudorca crassidens* (Owen, 1846). In S. H. Ridgway & S. R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 213–244). New York, NY: Academic Press.

Ohizumi, H. (2002). Dietary studies of toothed whales: A review of technical issues and new topics. *Fisheries Science 68* (Supplement 1): 264–267.

Okuyama, J., S. R. Benson, P. H. Dutton, and J. A. Seminoff. (2021). Changes in dive patterns of leatherback turtles with sea surface temperature and potential foraging habitats. *Ecosphere 12* (2): e03365. DOI:10.1002/ecs2.3365

Oliver, M. J., M. W. Breece, D. A. Fox, D. E. Haulsee, J. T. Kohut, J. Manderson, and T. Savoy. (2013). Shrinking the haystack: Using an AUV in an integrated ocean observatory to map Atlantic sturgeon in the coastal ocean. *Fisheries 38* (5): 210–216. DOI:10.1080/03632415.2013.782861

Olsen, E., W. P. Budgell, E. Head, L. Kleivane, L. Nøttestad, P. Prieto, M. A. Silva, H. Skov, G. A. Víkingsson, G. Waring, and N. Øien. (2009). First satellite-tracked long-distance movement of a sei whale (*Balaenoptera borealis*) in the North Atlantic. *Aquatic Mammals 35* (3): 313–318. DOI:doi: 10.1578/AM.35.3.2009.313

Olsen, K. M. and H. Larsson. (1995). *Terns of Europe and North America*. Princeton, NJ: Princeton University Press.

Olson, P. A. (2009). Pilot whales, *Globicephala melas* and *G. macrorhynchus*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 898–903). Cambridge, MA: Academic Press.

Onley, D. and P. Scofield. (2007). *Albatrosses, Petrels and Shearwaters of the World*. Princeton, NJ: Princeton University Press.

Oros, J., A. Torrent, P. Calabuig, and S. Deniz. (2005). Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998–2001). *Diseases of Aquatic Organisms 63* 13–24.

Ortmann, A. C., J. Anders, N. Shelton, L. Gong, A. G. Moss, and R. H. Condon. (2012). Dispersed oil disrupts microbial pathways in pelagic food webs. *PLoS ONE 7* (7): e42548. DOI:10.1371/journal.pone.0042548

Oswald, J. N., T. F. Norris, T. M. Yack, E. L. Ferguson, A. Kumar, J. Nissen, and J. Bell. (2016). Patterns of Occurrence and Marine Mammal Acoustic Behavior in Relation to Navy Sonar Activity Off Jacksonville, Florida. *Advances in Experimental Medicine and Biology 875* 791–799. DOI:10.1007/978-1-4939-2981-8_97

Overholtz, W. J. and G. T. Waring. (1991). Diet composition of pilot whales *Globicephala sp.* and common dolphins *Delphinus delphis* in the mid-Atlantic Bight during spring 1989. *Fishery Bulletin 89* (4): 723–728.

Pace III, R. M., P. J. Corkeron, and S. D. Kraus. (2017). State–space mark–recapture estimates reveal a recent decline in abundance of North Atlantic right whales. *Ecology and Evolution 7* (21): 8730–8741. DOI:https://doi.org/10.1002/ece3.3406

Pace III, R. M., R. Williams, S. D. Kraus, A. R. Knowlton, and H. M. Pettis. (2021). Cryptic mortality of North Atlantic right whales. *Conservation Science and Practice 3* (2): e346. DOI:https://doi.org/10.1111/csp2.346

Pace, R. M. (2021). *Revisions and further evaluations of the right whale abundance model: improvements for hypothesis testing* (NOAA Technical Memorandum NMFS-NE-269). Woods Hole, MA: Northeast Fisheries Science Center.

Pace, R. M., III and G. Silber. (2005). Simple analyses of ship and large whale collisions: Does speed kill? Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Pajuelo, M., K. A. Bjorndal, M. D. Arendt, A. M. Foley, B. A. Schroeder, B. E. Witherington, and A. B. Bolten. (2016). Long-term resource use and foraging specialization in male loggerhead turtles. *Marine Biology 163* (11): 235.

Palinkas, C. M., L. P. Sanford, and E. W. Koch. (2018). Influence of shoreline stabilization structures on the nearshore sedimentary environment in mesohaline Chesapeake Bay. *Estuaries and Coasts 41* (4): 952–965. DOI:10.1007/s12237-017-0339-6

Palka, D. (1995a). Influences on spatial patterns of Gulf of Maine harbor porpoises. *Developments in Marine Biology 4* 69–75. DOI:10.1016/S0163-6995(06)80011-9

Palka, D., S. Chavez-Rosales, E. Josephson, D. Cholewiak, H. Haas, L. Garrison, M. Jones, D. Sigourney, G. Waring, and M. Jech. (2017). *Atlantic Marine Assessment Program for Protected Species: 2010–2014* (OCS Study BOEM 2017-071). Washington, DC: Bureau of Ocean Energy Management, Atlantic OCS Region.

Palka, D. and M. Johnson. (2007). *Cooperative Research to Study Dive Patterns of Sperm Whales in the Atlantic Ocean*. New Orleans, LA: U.S. Department of the Interior, Minerals Management Service, Gulf of Mexico OCS Region.

Palka, D., A. Read, and C. Potter. (1997). Summary of knowledge of white-sided dolphins (*Lagenorhynchus acutus*) from U.S. and Canadian Atlantic waters. *Reports of the International Whaling Commission 47* 729–734.

Palka, D. L. (1995b). Abundance estimate of Gulf of Maine harbor porpoise. *Report of the International Whaling Commission 16* 27–50.

Palka, D. L. (1997). *A review of striped dolphins (Stenella coeruleoalba) in U.S. Atlantic waters*. Washington, DC: International Whaling Commission.

Palka, D. L. (2000). *Abundance of the Gulf of Maine/Bay of Fundy Harbor Porpoise Based on Shipboard and Aerial Surveys during 1999*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Palka, D. L. (2006). *Summer Abundance Estimates of Cetaceans in U.S. North Atlantic Navy Operating Areas*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Palka, D. L. (2012). *Cetacean Abundance Estimates in U.S. Northwestern Atlantic Ocean Waters from Summer 2011 Line Transect Survey*. Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Pandolfi, J. M., R. H. Bradbury, E. Sala, T. P. Hughes, K. A. Bjorndal, R. G. Cooke, D. McArdle, L. McClenachan, M. J. H. Newman, G. Paredes, R. R. Warner, and J. B. C. Jackson. (2003). Global trajectories of the long-term decline of coral reef ecosystems. *Science 301* 955–958.

Pandolfi, J. M., J. B. C. Jackson, N. Baron, R. H. Bradbury, H. M. Guzman, T. P. Hughes, C. V. Kappel, F. Micheli, J. C. Ogden, H. P. Possingham, and E. Sala. (2005). Are U.S. coral reefs on the slippery slope to slime? *Science 307* (5716): 1725–1726.

Panigada, S., M. Zanardelli, M. Mackenzie, C. Donovan, F. Melin, and P. S. Hammond. (2008). Modelling habitat preferences for fin whales and striped dolphins in the Pelagos Sanctuary (Western Mediterranean Sea) with physiographic and remote sensing variables. *Remote Sensing of Environment 112* (8): 3400–3412. DOI:10.1016/j.rse.2007.11.017

Papastamatiou, Y. P., R. D. Grubbs, J. L. Imhoff, S. J. B. Gulak, J. K. Carlson, and G. H. Burgess. (2015). A subtropical embayment serves as essential habitat for sub-adults and adults of the critically endangered smalltooth sawfish. *Global Ecology and Conservation 3* 764–775.

Parker, L. G. (1995). Encounter with a juvenile hawksbill turtle offshore Sapelo Island, Georgia. *Marine Turtle Newsletter 71* 19–22.

Parnell, K., S. McDonald, and A. Burke. (2007). Shoreline effects of vessel wakes, Marlborough Sounds, New Zealand. *Journal of Coastal Research* (Special Issue 50): 502–506.

Patino-Martinez, J., A. Marco, L. Quinones, and B. Godley. (2008). Globally significant nesting of the leatherback turtle (*Dermochelys coriacea*) on the Caribbean coast of Colombia and Panama. *Biological Conservation 141* (8): 1982–1988. DOI:10.1016/j.biocon.2008.05.012.

Patrício, A. R., C. E. Diez, R. P. Van Dam, and B. J. Godley. (2016). Novel insights into the dynamics of green turtle fibropapillomatosis. *Marine Ecology Progress Series 547* 247–255.

Patrício, A. R., L. A. Hawkes, J. R. Monsinjon, B. J. Godley, and M. M. P. B. Fuentes. (2021). Climate change and marine turtles: recent advances and future directions. *Endangered Species Research 44* 363–395.

Patrício, R., C. E. Diez, and R. P. Van Dam. (2014). Spatial and temporal variability of immature green turtle abundance and somatic growth in Puerto Rico. *Endangered Species Research 23* (1): 51–62.

Patrick, C. J., D. E. Weller, X. Li, and M. Ryder. (2014). Effects of shoreline alteration and other stressors on submerged aquatic vegetation in subestuaries of Chesapeake Bay and the Mid-Atlantic coastal bays. *Estuaries and Coasts 37* (6): 1516–1531. DOI:10.1007/s12237-014-9768-7

Patterson, K. L., J. W. Porter, K. B. Ritchie, S. W. Polson, E. Mueller, E. C. Peters, D. L. Santavy, and G. J. Smith. (2002). The etiology of white pox, a lethal disease of the Caribbean elkhorn coral, *Acropora palmata*. *Proceedings of the National Academy of Sciences 99* (13): 8725–8730.

Pavlov, D. and A. Kasumyan. (2000). Patterns and mechanisms of schooling behavior in fish: a review. *Journal of Ichthyology 40* (2): S163.

Payne, P. M. and D. W. Heinemann. (1993). The distribution of pilot whales (*Globicephala* spp.) in shelf/shelf edge and slope waters of the northeastern United States, 1978–1988. *Reports of the International Whaling Commission 14* 51–68.

Payne, P. M., D. W. Heinemann, and L. A. Selzer. (1990). *A Distributional Assessment of Cetaceans in Shelf/Shelf-Edge and Adjacent Slope Waters of the Northeastern United States Based on Aerial and Shipboard Surveys, 1978–1988*. Woods Hole, MA: U.S. Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Pelletier, S. K., K. Omland, K. S. Watrous, and T. S. Peterson. (2013). *Information Synthesis on the Potential for Bat Interactions with Offshore Wind Facilities–Final Report* (U.S. Department of the Interior, Bureau of Ocean Energy Management, Headquarters, Herndon, VA. Outer Continental Shelf Study Bureau of Energy Management). Herndon, VA: U.S. Department of the Interior, Bureau of Ocean Energy Management.

Peltier, H., M. Authier, F. Caurant, W. Dabin, P. Daniel, C. Dars, F. Demaret, E. Meheust, O. Van Canneyt, J. Spitz, and V. Ridoux. (2021). In the wrong place at the wrong time: Identifying spatiotemporal co-occurrence of bycaught common dolphins and fisheries in the Bay of Biscay (NE Atlantic) from 2010 to 2019. *Frontiers in Marine Science 8*. DOI:10.3389/fmars.2021.617342

Peña, J. (2006). *Plotting Kemp's Ridleys, plotting the future of sea turtle conservation* (SWOT Report). Washington, DC: The State of the World's Sea Turtles.

Perkins, J. S. and G. W. Miller. (1983). Mass stranding of *Steno bredanensis* in Belize. *Biotropica 15* (3): 235–236.

Perrin, W. F. (2001). *Stenella attenuata*. *American Society of Mammalogists 683* 1–8.

Perrin, W. F. (2002). *Stenella frontalis*. *American Society of Mammalogists 702* 1–6.

Perrin, W. F. (2009a). Atlantic spotted dolphin, *Stenella frontalis*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 54–56). Cambridge, MA: Academic Press.

Perrin, W. F. (2009b). Common dolphins, *Delphinus delphis* and *D. capensis*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 255–259). Cambridge, MA: Academic Press.

Perrin, W. F. (2009c). Pantropical spotted dolphin, *Stenella attenuata*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 819–821). Cambridge, MA: Academic Press.

Perrin, W. F. (2009d). Spinner dolphin, *Stenella longirostris*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 1100–1103). Cambridge, MA: Academic Press.

Perrin, W. F. and J. R. Geraci. (2002). Stranding. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (pp. 1192–1197). San Diego, CA: Academic Press.

Perrin, W. F. and J. W. Gilpatrick, Jr. (1994). Spinner dolphin, *Stenella longirostris* (Gray, 1828). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 99–128). San Diego, CA: Academic Press.

Perrin, W. F. and A. A. Hohn. (1994). Pantropical spotted dolphin, *Stenella attenuata*. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 71–98). San Diego, CA: Academic Press.

Perrin, W. F., S. Leatherwood, and A. Collet. (1994a). Fraser's dolphin, *Lagenodelphis hosei* Fraser, 1956. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 225–240). San Diego, CA: Academic Press.

Perrin, W. F., E. D. Mitchell, J. G. Mead, D. K. Caldwell, M. C. Caldwell, P. J. H. van Bree, and W. H. Dawbin. (1987). Revision of the spotted dolphins, *Stenella* spp. *Marine Mammal Science 3* (2): 99–170.

Perrin, W. F., E. D. Mitchell, J. G. Mead, D. K. Caldwell, and P. J. H. van Bree. (1981). *Stenella clymene*, a rediscovered tropical dolphin of the Atlantic. *Journal of Mammalogy 62* (3): 583–598.

Perrin, W. F. and W. A. Walker. (1975). The rough-toothed porpoise, *Steno bredanensis*, in the eastern tropical Pacific. *Journal of Mammalogy 56* 905–907.

Perrin, W. F., C. E. Wilson, and F. I. Archer, II. (1994b). Striped dolphin—*Stenella coeruleoalba* (Meyen, 1833). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 129–159). San Diego, CA: Academic Press.

Perrin, W. F., B. Würsig, and J. G. M. Thewissen. (2009). *Encyclopedia of Marine Mammals* (2nd ed.). Cambridge, MA: Academic Press.

Perryman, W. L. (2009). Melon-headed whale, *Peponocephala electra*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 719–721). Cambridge, MA: Academic Press.

Perryman, W. L., D. W. K. Au, S. Leatherwood, and T. A. Jefferson. (1994). Melon-headed whale, *Peponocephala electra* (Gray, 1846). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 363–386). San Diego, CA: Academic Press.

Perryman, W. L. and T. C. Foster. (1980). *Preliminary Report on Predation by Small Whales, Mainly the False Killer Whale, Pseudorca crassidens, on Dolphins (Stenella spp. and Delphinus delphis) in the Eastern Tropical Pacific*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Peterson, P. M., K. Romaschenko, Y. H. Arrieta, and J. M. Saarela. (2014). A molecular phylogeny and new subgeneric classification of *Sporobolus* (Poaceae: Chloridoideae: Sporobolinae). *TAXON 63* (6): 1212–1243. DOI:https://doi.org/10.12705/636.19

Pettis, H. M., R. M. Pace, III, and P. K. Hamilton. (2022). *North Atlantic Right Whale Consortium 2021 Annual Report Card*. Boston, MA: North Atlantic Right Whale Consortium.

Pfau, B. and W. M. Roosenburg. (2010). Diamondback terrapins in Maryland: Research and conservation. *Radiata 19* 2–34.

Phorn, B. (2018, October 2). *Dozens of endangered baby sea turtles hatched in New York City and crawled to the sea*. Retrieved August 23, 2023, from https://abcnews.go.com/US/dozens-endangered-baby-sea-turtles-hatched-york-city/story?id=58229791.

Piatt, J. F. and N. L. Naslund. (1995). Abundance, distribution, and population status of marbled murrelets in Alaska. In C. J. Ralph, G. L. Hunt, Jr., M. G. Raphael, & J. F. Piatt (Eds.), *Ecology and Conservation of the Marbled Murrelet* (pp. 285–294). Albany, CA: Pacific Southwest Research Station, Forest Service, U.S. Department of Agriculture.

Piersma, T., R. Hoekstra, A. Dekinga, A. Koolhaas, P. Wolf, P. Battley, and P. Wiersma. (1993). Scale and intensity of intertidal habitat use by knots calidris-canutus in the western Wadden Sea in relation to food, friends and foes. *Netherlands Journal of Sea Research 31* (4): 331–357. DOI:10.1016/0077-7579(93)90052-t

Pike, D. A. (2014). Forecasting the viability of sea turtle eggs in a warming world. *Global Change Biology 20* (1): 7–15.

Pike, D. A., E. A. Roznik, and I. Bell. (2015). Nest inundation from sea-level rise threatens sea turtle population viability. *Royal Society Open Science 2* (7): 150127.

Pitman, R. L., H. Fearnbach, R. LeDuc, J. W. Gilpatrick, Jr., J. K. B. Ford, and L. T. Ballance. (2007). Killer whales preying on a blue whale calf on the Costa Rica Dome: Genetics, morphometrics, vocalisations and composition of the group. *Journal of Cetacean Research and Management 9* (2): 151–157.

Pitman, R. L. and C. Stinchcomb. (2002). Rough-toothed dolphins (*Steno bredanensis*) as predators of mahimahi (*Coryphaena hippurus*). *Pacific Science 56* (4): 447–450.

Plotkin, P. and A. F. Amos. (1998). *Entanglement and ingestion of marine turtles stranded along the south Texas coast*. Presented at the Eighth Annual Workshop on Sea Turtle Conservation and Biology. Fort Fisher, NC.

Poloczanska, E. S., M. T. Burrows, C. J. Brown, J. G. Molinos, B. S. Halpern, O. Hoegh-Guldberg, C. V. Kappel, P. J. Moore, A. J. Richardson, D. S. Schoeman, and W. J. Sydeman. (2016). Responses of marine organisms to climate change across oceans. *Frontiers in Marine Science 3* (62): 1–21. DOI:10.3389/fmars.2016.00062

Ponganis, P. (2015). *Diving Physiology of Marine Mammals and Seabirds*. Cambridge, United Kingdom: Cambridge University Press.

Poole, A. F., R. O. Bierregaard, and M. S. Martell. (2002). *Osprey (Pandiol haliaetus)*. *The Birds of North America Online.* Retrieved 3, 2011, from http://bna.birds.cornell.edu/bna/species/563.

Poot, H., B. J. Ens, H. de Vries, M. A. H. Donners, M. R. Wernand, and J. M. Marquenie. (2008). Green light for nocturnally migrating birds. *Ecology and Society 13* (2): 47. DOI:47

Porter, J. W., P. Dustan, W. C. Jaap, K. L. Patterson, V. Kosmynin, O. W. Meier, M. E. Patterson, and M. Parsons. (2001). Patterns of spread of coral disease in the Florida Keys. *Hydrobiologia 460* 1–24. DOI:doi: 10.1023/A:1013177617800

Poulakis, G. R. and J. C. Seitz. (2004). Recent occurrence of the smalltooth sawfish, *Pristis pectinata* (Elasmobranchiomorphi: Pristidae), in Florida Bay and the Florida Keys, with comments on sawfish ecology. *Florida Scientist 67* 27–35.

Poulakis, G. R., P. W. Stevens, A. A. Timmers, C. J. Stafford, and C. A. Simpfendorfer. (2012). Movements of juvenile endangered smalltooth sawfish, *Pristis pectinata*, in an estuarine river system: Use of non-main-stem river habitats and lagged responses to freshwater inflow-related changes. *Environmental Biology of Fishes 96* (6): 763–778. DOI:10.1007/s10641-012-0070-x

Powell, S. M., R. L. Haedrich, and J. D. McEachran. (2003). The deep-sea fish fauna of the northern Gulf of Mexico. *Journal of Northwest Atlantic Fishery Science 31* 19–33. DOI:10.2960/j.v31.a2

Powers, S. P., J. H. Grabowski, C. H. Peterson, and W. J. Lindberg. (2003). Estimating enhancement of fish production by offshore artificial reefs: Uncertainty exhibited by divergent scenarios. *Marine Ecology Progress Series 264* 265–277.

Pratt, H. D., P. L. Bruner, and D. G. Berrett. (1987). *A Field Guide to the Birds of Hawaii and the Tropical Pacific*. Princeton, New Jersey: Princeton University Press.

Precht, W. F., B. E. Gintert, M. L. Robbart, R. Fura, and R. Woesik. (2016). Unprecedented Disease-Related Coral Mortality in Southeastern Florida. *Scientific Reports 6* (31374): 11. DOI:10.1038/srep31374

Prescott, R. (1982). Harbor seals: Mysterious lords of the winter beach. *Cape Cod Life 3* (4): 24–29.

Pugliares, K. R., T. W. French, G. S. Jones, M. E. Niemeyer, L. A. Wilcox, and B. J. Freeman. (2016). First records of the short-finned pilot whale (*Globicephala macrorhynchus*) in Massachusetts, USA: 1980 and 2011. *Aquatic Mammals 42* (3): 357–362. DOI:10.1578/AM.42.3.2016.357

Pusineri, C., V. Magnin, L. Meynier, J. Spitz, S. Hassani, and V. Ridoux. (2007). Food and feeding ecology of the common dolphin (*Delphinus delphis*) in the oceanic Northeast Atlantic and comparison with its diet in neritic areas. *Marine Mammal Science 23* (1): 30–47.

Putman, N. F., F. A. Abreu-Grobois, I. Iturbe-Darkistade, E. M. Putman, P. M. Richards, and P. Verley. (2015a). Deepwater Horizon oil spill impacts on sea turtles could span the Atlantic. *Biology Letters 11* (12).

Putman, N. F. and K. L. Mansfield. (2015). Direct Evidence of Swimming Demonstrates Active Dispersal in the Sea Turtle "Lost Years.". *Current Biology 25* (9): 1221–1227.

Putman, N. F., P. M. Richards, S. G. Dufault, E. Scott-Dention, K. McCarthy, R. T. Beyea, C. W. Caillouet Jr., W. D. Heyman, E. E. Seney, K. L. Mansfield, and B. J. Gallaway. (2023). Modeling juvenile sea turtle bycatch risk in commercial and recreational fisheries. *iScience* 1-36. DOI:https://doi.org/10.1016/j.isci.2023.105977

Putman, N. F., E. E. Seney, P. Verley, D. J. Shaver, M. C. Lopez-Castro, M. Cook, V. Guzman, B. Brost, S. A. Ceriani, R. J. G. D. Miron, L. J. Pena, M. Tzeek, R. A. Valverde, C. C. G. Canton, J. A. R. Ley, M. C. Tumlin, W. G. Teas, C. W. Caillouet Jr., E. Cuevas, B. J. Gallaway, P. M. Richards, and K. L. Mansfield. (2019). Predicted distributions and abundances of the sea turtle 'lost years' in the western North Atlantic Ocean. *Ecography 43* 506–517. DOI:10.1111/ecog.04929

Putman, N. F., P. Verley, C. S. Endres, and K. J. Lohmann. (2015b). Magnetic navigation behavior and the oceanic ecology of young loggerhead sea turtles. *The Journal of Experimental Biology 218* (7): 1044–1050.

Quattrini, A. M., M. S. Nizinski, J. D. Chaytor, A. W. Demopoulos, E. B. Roark, S. C. France, J. A. Moore, T. Heyl, P. J. Auster, B. Kinlan, C. Ruppel, K. P. Elliott, B. R. Kennedy, E. Lobecker, A. Skarke, and T. M. Shank. (2015). Exploration of the canyon-incised continental margin of the Northeastern United States reveals dynamic habitats and diverse communities. *PLoS ONE 10* (10): 1–32. DOI:10.1371/journal.pone.0139904

Queiroz, N., N. E. Humphries, A. Couto, M. Vedor, I. da Costa, A. M. M. Sequeira, G. Mucientes, A. M. Santos, F. J. Abascal, D. L. Abercrombie, K. Abrantes, D. Acuña-Marrero, A. S. Afonso, P. Afonso, D. Anders, G. Araujo, R. Arauz, P. Bach, A. Barnett, D. Bernal, M. L. Berumen, S. Bessudo Lion, N. P. A. Bezerra, A. V. Blaison, B. A. Block, M. E. Bond, R. Bonfil, R. W. Bradford, C. D. Braun, E. J. Brooks, A. Brooks, J. Brown, B. D. Bruce, M. E. Byrne, S. E. Campana, A. B. Carlisle, D. D. Chapman, T. K. Chapple, J. Chisholm, C. R. Clarke, E. G. Clua, J. E. M. Cochran, E. C. Crochelet, L. Dagorn, R. Daly, D. D. Cortés, T. K. Doyle, M. Drew, C. A. J. Duffy, T. Erikson, E. Espinoza, L. C. Ferreira, F. Ferretti, J. D. Filmalter, G. C. Fischer, R. Fitzpatrick, J. Fontes, F. Forget, M. Fowler, M. P. Francis, A. J. Gallagher, E. Gennari, S. D. Goldsworthy, M. J. Gollock, J. R. Green, J. A. Gustafson, T. L. Guttridge, H. M. Guzman, N. Hammerschlag, L. Harman, F. H. V. Hazin, M. Heard, A. R. Hearn, J. C. Holdsworth, B. J. Holmes, L. A. Howey, M. Hoyos, R. E. Hueter, N. E. Hussey, C. Huveneers, D. T. Irion, D. M. P. Jacoby, O. J. D. Jewell, R. Johnson, L. K. B. Jordan, S. J. Jorgensen, W. Joyce, C. A. Keating Daly, J. T. Ketchum, A. P. Klimley, A. A. Kock, P. Koen, F. Ladino, F. O. Lana, J. S. E. Lea, F. Llewellyn, W. S. Lyon, A. MacDonnell, B. C. L. Macena, H. Marshall, J. D. McAllister, R. McAuley, M. A. Meÿer, J. J. Morris, E. R. Nelson, Y. P. Papastamatiou, T. A. Patterson, C. Peñaherrera-Palma, J. G. Pepperell, S. J. Pierce, F. Poisson, L. M. Quintero, A. J. Richardson, P. J. Rogers, C. A. Rohner, D. R. L. Rowat, M. Samoilys, J. M. Semmens, M. Sheaves, G. Shillinger, M. Shivji, S. Singh, G. B. Skomal, M. J. Smale, L. B. Snyders, G. Soler, M. Soria, K. M. Stehfest, J. D. Stevens, S. R. Thorrold, M. T. Tolotti, A. Towner, P. Travassos, J. P. Tyminski, F. Vandeperre, J. J. Vaudo, Y. Y. Watanabe, S. B. Weber, B. M. Wetherbee, T. D. White, S. Williams, P. M. Zárate, R. Harcourt, G. C. Hays, M. G. Meekan, M. Thums, X. Irigoien, V. M. Eguiluz, C. M. Duarte, L. L. Sousa, S. J. Simpson, E. J. Southall and D. W. Sims. (2019). Global spatial risk assessment of sharks under the footprint of fisheries. *Nature 572* (7770): 461–466. DOI:10.1038/s41586-019-1444-4

Queiroz, N., N. E. Humphries, G. Mucientes, N. Hammerschlag, F. P. Lima, K. L. Scales, P. I. Miller, L. L. Sousa, R. Seabra, and D. W. Sims. (2016). Ocean-wide tracking of pelagic sharks reveals extent of overlap with longline fishing hotspots. *Proceedings of the National Academy of Sciences 113* (6): 1582–1587.

Rabon, D. R., Jr., S. A. Johnson, R. Boettcher, M. Dodd, M. Lyons, S. Murphy, S. Ramsey, S. Roff, and K. Stewart. (2003). Confirmed leatherback turtle (*Dermochelys coriacea*) nests from North Carolina, with a summary of leatherback nesting activities north of Florida. *Marine Turtle Newsletter 101* 4–8.

Rafferty, A. R., E. O. Brazer, Jr., and R. D. Reina. (2012). Depredation by harbor seal and spiny dogfish in a Georges Bank gillnet fishery. *Fisheries Management and Ecology 19* 264–272.

Raine, A. F., C. Gjerdrum, I. Pratte, J. Madeiros, J. J. Felis, and J. Adams. (2021). Marine distribution and foraging habitat highlight potential threats at sea for the Endangered Bermuda petrel *Pterodroma cahow*. *Endangered Species Research 45* 337–356. DOI:10.3354/esr01139

Ramirez-Llodra, E., B. De Mol, J. B. Company, M. Coll, and F. Sardà. (2013). Effects of natural and anthropogenic processes in the distribution of marine litter in the deep Mediterranean Sea. *Progress in Oceanography 118* 273-287.

Ramirez, M. D., L. Avens, L. R. Goshe, M. L. Snover, M. Cook, and S. S. Heppell. (2020). Regional variation in Kemp's Ridley sea turtle diet composition and its potential relationship With somatic growth. *Frontiers in Marine Science 7* 1–18. DOI:10.3389/fmars.2020.00253

Ramp, C., J. Delarue, M. Berube, P. S. Hammond, and R. Sears. (2014). Fin whale survival and abundance in the Gulf of St. Lawrence, Canada. *Endangered Species Research 23* 125–132. DOI:10.3354/esr00571

Ramp, C., J. Delarue, P. J. Palsboll, R. Sears, and P. S. Hammond. (2015). Adapting to a warmer ocean—Seasonal shift of baleen whale movements over three decades. *PLoS ONE 10* (3): e0121374. DOI:10.1371/journal.pone.0121374

Ramp, C., V. Lesage, A. Ollier, M. Auger-Méthé, and R. Sears. (2024). Migratory movements of fin whales from the Gulf of St. Lawrence challenge our understanding of the Northwest Atlantic stock structure. *Scientific Reports 14* (1): 11472. DOI:10.1038/s41598-024-62173-1

Randall, C. J. and R. van Woesik. (2015). Contemporary white-band disease in Caribbean corals driven by climate change. *Nature Climate Change 5* (4): 375–379. DOI:10.1038/nclimate2530

Randall, J. E. (1964). Contributions to the biology of the queen conch, *Strombus gigas*. *Bulletin of Marine Science 14* (2): 246-295.

Randall, M. T. and K. J. Sulak. (2012). Evidence of autumn spawning in Suwannee River Gulf sturgeon, *Acipenser oxyrinchus desotoi* (Vladykov, 1955). *Journal of Applied Ichthyology 28* (4): 489–495. DOI:10.1111/j.1439-0426.2012.01960

Rappucci, G., L. Garrison, M. Soldevilla, J. Ortega-Ortiz, J. Reid, L. Aichinger-Dias, K. Mullin, and J. Litz. (2023). *Gulf of Mexico Marine Assessment Program for Protected Species (GoMMAPPS): Marine Mammals*. New Orleans, LA: U.S. Department of the Interior, Bureau of Ocean Energy Management, New Orleans Office.

Read, A. J. (1999). Harbor porpoise, *Phocoena phocoena* (Linnaeus, 1758). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 323–355). San Diego, CA: Academic Press.

Read, A. J., S. Barco, J. Bell, D. L. Borchers, M. L. Burt, E. W. Cummings, J. Dunn, E. M. Fougeres, L. Hazen, L. E. W. Hodge, A.-M. Laura, R. J. McAlarney, P. Nilsson, D. A. Pabst, C. G. M. Paxton, S. Z. Schneider, K. W. Urian, D. M. Waples, and W. A. McLellan. (2014). Occurrence, distribution, and abundance of cetaceans in Onslow Bay, North Carolina, USA. *Journal of Cetacean Research and Management 14* 23–35.

Record, N. R., J. A. Runge, D. E. Pendleton, W. M. Balch, K. T. A. Davies, A. J. Pershing, C. L. Johnson, K. Stamieszkin, R. Ji, Z. Feng, S. D. Kraus, R. D. Kenney, C. A. Hudak, C. A. Mayo, C. Chen, J. E. Salisbury, and C. R. S. Thompson. (2019). Rapid Climate-Driven Circulation Changes Threaten Conservation of Endangered North Atlantic Right Whales. *Oceanography 32* (2): 162–169.

Reddin, D. G. and P. B. Short. (1991). Postsmolt Atlantic salmon (*Salmo salar)* in the Labrador Sea. *Canadian Journal of Fisheries and Aquatic Sciences 48* (1): 2–6.

Redfern, J. V., T. J. Moore, E. A. Becker, J. Calambokidis, S. P. Hastings, L. M. Irvine, B. R. Mate, D. M. Palacios, and L. Hawkes. (2019). Evaluating stakeholder-derived strategies to reduce the risk of ships striking whales. *Diversity and Distributions 00* 1–11. DOI:10.1111/ddi.12958

Reece, J. S., D. Passeri, L. Ehrhart, S. C. Hagen, A. Hays, C. Long, R. F. Noss, M. Bilskie, C. Sanchez, and M. V. Schwoerer. (2013). Sea level rise, land use, and climate change influence the distribution of loggerhead turtle nests at the largest U.S.A. rookery (Melbourne Beach, Florida). *Marine Ecology Progress Series 493* 259–274.

Reed, J., S. Farrington, H. Moe, S. L. Harter, M. D. Hanisak, A. David, L. Horn, G. Taylor, J. White, J. Voss, S. A. Pomponi, and C. Diaz. (2017). *Characterization of Mesophotic Coral/Sponge Habitats and Fish Assemblages in the Regions of Pulley Ridge and Tortugas from ROV Dives during R/V Walton Smith Cruises of 2012 to 2015*. Silver Spring, MD: National Oceanic and Atmospheric Administration, National Ocean Service.

Reed, J. K., S. Farrington, A. David, S. Harter, S. A. Pomponi, M. C. Diaz, J. D. Voss, K. D. Spring, A. C. Hine, and V. H. Kourafalou. (2019). Pulley Ridge, Gulf of Mexico, USA *Mesophotic Coral Ecosystems* (pp. 57-69). Cham, Switzerland: Springer International Publishing.

Reed, J. K., S. Farrington, M. C. Díaz, S. A. Pomponi, M. D. Hanisak, and J. D. Voss. (2021). *Characterization of the Mesophotic Coral Reefs in the Florida Keys National Marine Sanctuary*. Boca Raton, FL: NOAA CIOERT Report. Harbor Branch Oceanographic Institution Technical Report Number 198.

Reeves, R. R., E. Mitchell, and H. Whitehead. (1993). Status of the northern bottlenose whale, *Hyperoodon ampullatus*. *Canadian Field-Naturalist 107* 490–508.

Reeves, R. R., B. D. Smith, E. A. Crespo, and G. Notarbartolo di Sciara. (2003). *Dolphins, Whales and Porpoises: 2002—2010 Conservation Action Plan for the World's Cetaceans*. Gland, Switzerland and Cambridge, United Kingdom: International Union for Conservation of Nature.

Reeves, R. R., T. D. Smith, E. A. Josephson, P. J. Clapham, and G. Woolmer. (2004). Historical observations of humpback and blue whales in the North Atlantic Ocean: Clues to migratory routes and possible additional feeding grounds. *Marine Mammal Science 20* (4): 774–786. DOI:10.1111/j.1748-7692.2004.tb01192

Reeves, R. R., T. D. Smith, R. L. Webb, J. Robbins, and P. J. Clapham. (2002a). Humpback and fin whaling in the Gulf of Maine from 1800 to 1918. *Marine Fisheries Review 64* (1): 1–12.

Reeves, R. R., B. S. Stewart, P. J. Clapham, and J. A. Powell. (2002b). *National Audubon Society Guide to Marine Mammals of the World*. New York, NY: Alfred A. Knopf.

Reeves, R. R., B. S. Stewart, and S. Leatherwood. (1992). The Sierra Club Handbook of Seals and Sirenians (pp. 359). San Francisco, CA: Sierra Club Books.

Reinert, T. R., A. C. Spellman, and B. L. Bassett. (2017). Entanglement in and ingestion of fishing gear and other marine debris by Florida manatees, 1993 to 2012. *Endangered Species Research 32* 415–427.

Renaud, M. L., J. A. Carpenter, J. A. Williams, and A. M. Landry, Jr. (1996). Kemp's ridley sea turtle (*Lepidochelys kempii*) tracked by satellite telemetry from Louisiana to nesting beach at Rancho Nuevo, Tamaulipas, Mexico. *Chelonian Conservation and Biology 2* (1): 108–109.

Renaud, M. L., J. A. Carpenter, J. A. Williams, and S. A. Manzellatirpak. (1995). Activities of juvenile green turtles, *Chelonia mydas*, at a jettied pass in south Texas. *Fishery Bulletin 93* (3): 586–593.

Reneker, J. L. and S. J. Kamel. (2016). Climate change increases the production of female hatchlings at a northern sea turtle rookery. *Ecology 97* (12): 3257–3264.

Rester, J. and R. Condrey. (1996). The occurrence of the hawksbill turtle, *Eretmochelys imbricata*, along the Louisiana coast. *Gulf of Mexico Science 14* (2): 112–114.

Rex, M. A., R. J. Etter, J. S. Morris, J. Crouse, C. R. McClain, N. A. Johnson, C. T. Stuart, J. W. Deming, R. Thies, and R. Avery. (2006). Global bathymetric patterns of standing stock and body size in the deep-sea benthos. *Marine Ecology Progress Series 317* 1-8.

Reynolds, J. E., III, J. A. Powell, and C. R. Taylor. (2009). Manatees, *Trichechus manatus*, *T. senegalensis*, and *T. inunguis*. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 682–691). Cambridge, MA: Academic Press.

Ribic, C. A., S. B. Sheavly, and D. J. Rugg. (2011). Trends in Marine Debris in the U.S. Caribbean and the Gulf of Mexico 1996-2003. *Journal of Integrated Coastal Zone Management 11* (1): 7-19.

Ribic, C. A., S. B. Sheavly, D. J. Rugg, and E. S. Erdmann. (2010). Trends and drivers of marine debris on the Atlantic coast of the United States 1997–2007. *Marine Pollution Bulletin 60* (8): 1231–1242. DOI:10.1016/j.marpolbul.2010.03.021

Rice, A. N., K. Palmer, J. T. Tielens, C. A. Muirhead, and C. W. Clark. (2014). Potential Bryde's whale (*Balaenoptera edeni*) calls recorded in the northern Gulf of Mexico. *The Journal of the Acoustical Society of America 135* (5): 3066-3076.

Rice, D. W. (1989). Sperm whale *Physeter macrocephalus* Linnaeus, 1758. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 4, pp. 177–234). San Diego, CA: Academic Press.

Rice, D. W. (1998). *Marine Mammals of the World: Systematics and Distribution* (Society for Marine Mammalogy Special Publication). Lawrence, KS: Society for Marine Mammalogy.

Rice, M. R. and G. H. Balazs. (2008). Diving behavior of the Hawaiian green turtle (*Chelonia mydas*) during oceanic migrations. *Journal of Experimental Marine Biology and Ecology 356* (1–2): 121–127. DOI:10.1016/j.jembe.2007.12.010

Richardson and McGillvary. (1991). *Proceedings of the Twenty-fourth Annual Symposium on Sea Turtle Biology and Conservation* (NOAA Technical Memorandum). Miami, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Richardson, W. J., C. R. Greene, Jr., C. I. Malme, and D. H. Thomson. (1995). *Marine Mammals and Noise*. San Diego, CA: Academic Press.

Richmond, A. M. and B. Kynard. (1995). Ontogenetic behavior of shortnose sturgeon, *Acipenser brevirostrum*. *Copeia 1995* 172–182.

Rider, M. J., L. Avens, H. L. Haas, J. M. Hatch, S. H. Patel, and C. R. Sasso. (2024). Where the leatherbacks roam: movement behavior analyses reveal novel foraging locations along the Northwest Atlantic shelf. *Frontiers in Marine Science 11* 1-16. DOI:10.3389/fmars.2024.1325139

Rigby, C., R. Barreto, J. Carlson, D. Fernando, S. Fordham, M. Francis, K. Herman, R. Jabado, K. Liu, A. Marshall, N. Pacoureau, E. Romanov, R. Sherley, and H. Winker. (2019). *Ocean Whitetip Shark*

Carcharhinus longimanus. The IUCN Red List of Threatened Species 2019: e. T39374A2911619. Gland, Switzerland: International Union for Conservation of Nature.

Risch, D., M. Castellote, C. W. Clark, G. E. Davis, P. J. Dugan, L. E. W. Hodge, A. Kumar, K. Lucke, M. D. K., S. L. Nieukirk, C. M. Popescu, C. Ramp, A. J. Read, A. N. Rice, M. A. Silva, U. Siebert, K. M. Stafford, H. Verdaat, and S. M. Van Parijs. (2014). Seasonal migrations of North Atlantic minke whales: Novel insights from large-scale passive acoustic monitoring networks. *Movement Ecology 2* 1–17. DOI:10.1186/s40462-014-0024-3

Risch, D., C. W. Clark, P. J. Dugan, M. Popescu, U. Siebert, and S. M. Van Parijs. (2013). Minke whale acoustic behaviour and multi-year seasonal and diel vocalization patterns in Massachusetts Bay, USA. *Marine Ecology Progress Series 489* 279–295. DOI:10.3354/meps10426

Roberts, J. J., T. M. Yack, E. Fujioka, P. N. Halpin, M. F. Baumgartner, O. Boisseau, S. Chavez-Rosales, T. V. N. Cole, M. P. Cotter, G. E. Davis, R. A. DiGiovanni, Jr., L. C. Ganley, L. P. Garrison, C. P. Good, T. A. Gowan, K. A. Jackson, R. D. Kenney, C. B. Khan, A. R. Knowlton, S. D. Kraus, G. G. Lockhart, K. S. Lomac-MacNair, C. A. Mayo, B. E. McKenna, W. A. McLellan, D. P. Nowacek, O. O' Brien, D. A. Pabst, D. L. Palka, E. M. Patterson, D. E. Pendleton, E. Quintana-Rizzo, N. R. Record, J. V. Redfern, M. E. Rickard, M. White, A. D. Whitt, and A. M. Zoidis. (2024). North Atlantic right whale density surface model for the US Atlantic evaluated with passive acoustic monitoring. *Marine Ecology Progress Series 732* 167–192. DOI:10.3354/meps14547

Roberts, K. E., L. P. Garrison, J. Ortega-Ortiz, C. Hu, Y. Zhang, C. R. Sasso, M. Lamont, and K. M. Hart. (2022). The Influence of Satellite-Derived Environmental and Oceanographic Parameters on Marine Turtle Time at Surface in the Gulf of Mexico. *Remote Sensing 14* (4534): 1-17. DOI:10.3390/rs14184534

Roberts, M. A., C. J. Anderson, B. Stender, A. Segars, J. D. Whittaker, J. M. Grady, and J. M. Quattro. (2005). Estimated contribution of Atlantic coastal loggerhead turtle nesting populations to offshore feeding aggregations. *Conservation Genetics 6* 133–139. DOI:10.1007/s10592-004-7737-6

Roberts, S. M., A. M. Boustany, and P. N. Halpin. (2020). Substrate-dependent fish have shifted less in distribution under climate change. *Communications Biology 3* (1): 586. DOI:10.1038/s42003-020-01325-1

Robertson, G. J. and R. I. Goudie. (1999). Harlequin Duck (*Histrionicus histrionicus*). *The Birds of North America Online* (466): 2. DOI:10.2173/bna.466

Robertson, K. M. and S. J. Chivers. (1997). Prey occurrence in pantropical spotted dolphins, *Stenella attenuata*, from the eastern tropical Pacific. *Fishery Bulletin 95* (2): 334–348.

Robinson, N. J., S. E. Valentine, P. Santidrián Tomillo, V. S. Saba, J. R. Spotila, and F. V. Paladino. (2013). Multidecadal trends in the nesting phenology of Pacific and Atlantic leatherback turtles are associated with population demography. *Endangered Species Research 24* 197–206.

Robydek, A. and J. M. Nunley. (2012). *Determining Marine Migration Patterns and Behavior of Gulf Sturgeon in the Gulf of Mexico off Eglin Air Force Base, Florida* (Department of Defense Legacy Program Technical Notes). Eglin Air Force Base, FL: U.S. Department of Defense.

Roch, S., C. Friedrich, and A. Brinker. (2020). Uptake routes of microplastics in fishes: Practical and theoretical approaches to test existing theories. *Scientific Reports 10* (1): 3896. DOI:10.1038/s41598-020-60630-1

Rocha, L. A., H. T. Pinheiro, B. Shepherd, Y. P. Papastamatiou, O. J. Luiz, R. L. Pyle, and P. Bongaerts. (2018). Mesophotic coral ecosystems are threatened and ecologically distinct from shallow water reefs. *Science 361* (6399): 281-284.

Rochford, M. R., K. L. Krysko, F. J. Mazzotti, M. H. Shirley, M. W. Parry, J. A. Wasilewski, J. S. Beauchamp, C. R. Gillette, E. F. Metzger, III, and M. A. Squires. (2016). Molecular analyses confirming the introduction of Nile crocodiles, *Crocodylus niloticus* Laurenti 1768 (Crocodylidae), in southern Florida, with an assessment of potential for establishment, spread, and impacts. *Herpetological Conservation and Biology 11* (1): 80–89.

Rochman, C. M., M. A. Browne, A. J. Underwood, J. A. van Franeker, Richard C. Thompson, and L. A. Amaral-Zettler. (2016). The ecological impacts of marine debris: unraveling the demonstrated evidence from what is perceived. *Ecology 97* (2): 302–312. DOI:https://doi.org/10.1890/14-2070.1

Rockwood, R. C., J. Calambokidis, and J. Jahncke. (2017). High mortality of blue, humpack and fin whales from modeling of vessel collisions on the U.S. West Coast suggests population impacts and insufficient protection. *PLoS ONE 12* (8): e0183052. DOI:10.1371/journal.pone.0183052

Rode, K. D., E. V. Regehr, D. C. Douglas, G. Durner, A. E. Merocher, G. W. Thiemann, and S. M. Budge. (2014). Variation in the respones of an Arctic top predator experiencing habitat loss: Feeding and reproductive ecology of two polar bear populations. *Global Change Biology 20* 76–88. DOI:10.1111/gcb.12339

Roden, C. L. and K. D. Mullin. (2000). Sightings of cetaceans in the northern Caribbean Sea and adjacent waters, winter 1995. *Caribbean Journal of Science 36* (3–4): 280–288.

Rodríguez, A., N. Holmes, P. Ryan, K. Wilson, L. Faulquier, Y. Murillo, A. Raine, J. Penniman, V. Neves, B. Rodríguez, J. Negro, A. Chiaradia, P. Dann, T. Anderson, B. Metzger, M. Shirai, L. Deppe, J. Wheeler, P. Hodum, C. Gouveia, V. Carmo, G. Carreira, L. Delgado-Alburqueque, C. Guerra-Correa, F. Couzi, M. Travers, and M. Le Corre. (2017). A global review of seabird mortality caused by land-based artificial lights. *Conservation Biology* 1–40. DOI:10.1111/cobi.12900

Roff, G., M. H. Ledlie, J. C. Ortiz, and P. J. Mumby. (2011). Spatial patterns of parrotfish corallivory in the Caribbean: The importance of coral taxa, density and size. *PLoS ONE 6* (12): 1–11. DOI:10.1371/journal.pone.0029133

Rogers, R., K. H. Choate, L. M. Crowe, J. M. Hatch, M. C. James, E. Matzen, S. H. Patel, C. R. Sasso, L. A. Siemann, and H. L. Haas. (2024). Investigating leatherback surface behavior using a novel tag design and machine learning. *Journal of Experimental Marine Biology and Ecology 576* 152012. DOI:https://doi.org/10.1016/j.jembe.2024.152012

Rogillio, H. E., R. T. Ruth, E. H. Behrens, C. N. Doolittle, W. J. Granger, and J. P. Kirk. (2007). Gulf sturgeon movements in the Pearl River drainage and the Mississippi Sound. *North American Journal of Fisheries Management 27* (1): 89–95. DOI:10.1577/m05-170.1

Roman, M. R. and J. J. Pierson. (2022). Interactive Effects of Increasing Temperature and Decreasing Oxygen on Coastal Copepods. *The Biological Bulletin 243* (2): 171–183. DOI:10.1086/722111

Romero, A., I. A. Agudo, S. M. Green, and G. Notarbartolo di Sciara. (2001). *Cetaceans of Venezuela: Their Distribution and Conservation Status* (NOAA Technical Report). Seattle, WA: National Oceanic and Atmospheric Administration.

Rommel, S., A. M. Costidis, T. D. Pitchford, J. D. Lightsey, R. H. Snyder, and E. M. Haubold. (2007). Forensic methods for characterizing watercraft from watercraft-induced wounds on the Florida manatee (*Trichechus manatus latirostris*). *Marine Mammal Science 23* (1): 110–132. DOI:10.1111/j.1748-7692.2006.00095

Ronald, K. and J. L. Dougan. (1982). The ice lover: Biology of the harp seal (*Phoca groenlandica*). *Science 215* 928–933.

Ronald, K. and P. J. Healey. (1981). Harp seal, *Phoca groenlandica* Erxleben, 1777. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 2, pp. 55–87). San Diego, CA: Academic Press.

Ronconi, R. A., P. G. Ryan, and Y. Ropert-Coudert. (2010). Diving of great shearwaters (*Puffinus gravis*) in cold and warm water regions of the South Atlantic ocean. *PLoS ONE 5* (11): e15508. DOI:10.1371/journal.pone.0015508

Roosenburg, W. M., W. Cresko, M. Modesitte, and M. B. Robbins. (1997). Diamondback terrapin (*Malaclemys terrapin*) mortality in crab pots. *Conservation Biology 11* (5): 1166–1172.

Root, B. G., M. R. Ryan, and P. M. Mayer. (1992). Piping plover survival in the great-plains. *Journal of Field Ornithology 63* (1): 10–15.

Rosel, P. E., P. Corkeron, L. Engleby, D. Epperson, K. D. Mullin, M. S. Soldevilla, and B. L. Taylor. (2016). *Status Review of Byrde's Whales (Balaenopter edeni) in the Gulf of Mexico Under the Endangered Species Act* (NOAA Technical Memorandum NMFS-SEFSC-692). Lafayette, LA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Rosel, P. E., S. C. France, J. Y. Wang, and T. D. Kocher. (1999). Genetic structure of harbor porpoise *Phocoena phocoena* populations in the northwest Atlantic based on mitochondrial and nuclear markers. *Molecular Ecology 8* S41–S54.

Rosel, P. E. and H. Watts. (2008). Hurricane impacts on bottlenose dolphins in the northern Gulf of Mexico. *Gulf of Mexico Science 25* (1): 88–94.

Rosel, P. E. and L. A. Wilcox. (2014). Genetic evidence reveals a unique lineage of Bryde's whales in the northern Gulf of Mexico. *Endangered Species Research 25* 19–34.

Rosel, P. E., L. A. Wilcox, T. K. Yamada, and K. D. Mullin. (2021). A new species of baleen whale (*Balaenoptera*) from the Gulf of Mexico, with a review of its geographic distribution. *Marine Mammal Science 37* (2): 577–610.

Rosenfeld, M., M. George, and J. M. Terhune. (1988). Evidence of autumnal harbour seal, *Phoca vitulina*, movement from Canada to the United States. *Canadian Field-Naturalist 102* (3): 527–529.

Ross, G. J. B. and A. J. Bass. (1971). Shark attack on an ailing dolphin *Stenella coeruleoalba* (Meyen). *South African Journal of Science 67* 413–414.

Ross, G. J. B. and S. Leatherwood. (1994). Pygmy killer whale *Feresa attenuata* Gray, 1874. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 5, pp. 387–404). San Diego, CA: Academic Press.

Ross, S. T., W. T. Slack, R. J. Heise, M. A. Dugo, H. Rogillio, B. R. Bowen, P. Mickle, and R. W. Heard. (2009). Estuarine and coastal habitat use of Gulf sturgeon (*Acipenser oxyrinchus desotoi*) in the North-Central Gulf of Mexico. *Estuaries and Coasts 32* (2): 360–374. DOI:10.1007/s12237-008-9122-z

Ross, S. W., S. Brooke, A. M. Quattrini, M. Rhode, and J. C. Watterson. (2015a). A deep-sea community, including *Lophelia pertusa*, at unusually shallow depths in the western North Atlantic Ocean off northeastern Florida. *Marine Biology 162* (3): 635–648. DOI:10.1007/s00227-015-2611-2

Ross, S. W., M. Rhode, and S. Brooke. (2017). Deep-sea coral and hardbottom habitats on the west Florida slope, eastern Gulf of Mexico. *Deep–Sea Research I 120* 14–28.

Ross, S. W., M. Rhode, and A. M. Quattrini. (2015b). Demersal fish distribution and habitat use within and near Baltimore and Norfolk Canyons, U.S. middle Atlantic slope. *Deep Sea Research Part I: Oceanographic Research Papers 103* 137-154.

Ross, S. W., M. Rhode, S. T. Viada, and R. Mather. (2016). Fish species associated with shipwreck and natural hard-bottom habitats from the middle to outer continental shelf of the Middle Atlantic Bight near Norfolk Canyon. *Fishery Bulletin 114* (1): 45–57. DOI:10.7755/FB.114.1.4

Rothenberger, P., J. Blondeau, C. Cox, S. Curtis, W. S. Fisher, V. Garrison, Z. Hillis-Starr, C. F. G. Jeffrey, E. Kadison, I. Lundgren, W. J. Miller, E. Muller, R. Nemeth, S. Paterson, C. Rogers, T. Smith, A. Spitzack, M. Taylor, W. Toller, J. Wright, D. Wusinich-Mendez, and J. Waddell. (2008). The State of Coral Reef Ecosystems of the U.S. Virgin Islands. In J. E. Waddell & A. M. Clarke (Eds.), *The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States* (pp. 29–73). Silver Spring, MD: National Oceanic and Atmospheric Administration.

Rothermel, E. R., M. T. Balazik, J. E. Best, M. W. Breece, D. A. Fox, B. I. Gahagan, D. E. Haulsee, A. L. Higgs, M. H. P. O'Brien, M. J. Oliver, I. A. Park, and D. H. Secor. (2020). Comparative migration ecology of striped bass and Atlantic sturgeon in the US Southern mid-Atlantic bight flyway. *PLoS ONE 15* (6): e0234442. DOI:10.1371/journal.pone.0234442

Rough, V. (1995). *Gray Seals in Nantucket Sound, Massachusetts: Winter and Spring, 1994*. Washington, DC: U.S. Marine Mammal Commission.

Rubega, M. A., D. Schamel, D. M. Tracy, A. Poole, and F. Gill. (2000). Red-necked Phalarope (*Phalaropus lobatus*). *The Birds of North America Online* (538): 5. DOI:10.2173/bna.538

Rudd, M. B., R. N. M. Ahrens, W. E. Pine, S. K. Bolden, and L. Jacobson. (2014). Empirical, spatially explicit natural mortality and movement rate estimates for the threatened Gulf sturgeon (*Acipenser oxyrinchus desotoi*). *Canadian Journal of Fisheries and Aquatic Sciences 71* (9): 1407–1417. DOI:10.1139/cjfas-2014-0010

Runge, M. C., C. A. Langtimm, and W. L. Kendall. (2004). A Stage-Based Model of Manatee Population Dynamics. *Marine Mammal Science 20* (3): 361–385.

Runge, M. C., C. A. Langtimm, J. Martin, and C. J. Fonnesbeck. (2015). *Status and Threats Analysis for the Florida Manatee (Trichechus manatus latirostris), 2012* (Open File Report 2015-1083). Reston, VA: U.S. Geological Survey.

Runge, M. C., C. A. Sanders-Reed, C. A. Langtimm, and C. J. Fonnesbeck. (2007). *A Quantitative Threats Analysis for the Florida Manatee (Trichechus manatus latirostris)*. Laurel, MD: U.S. Geological Survey, Patuxent Wildlife Research Center.

Runge, M. C., C. A. Sanders-Reed, C. A. Langtimm, J. A. Hostetler, J. Martin, C. J. Deutsch, L. I. Ward-Geiger, and G. L. Mahon. (2017). *Status and threats analysis for the Florida manatee (Trichechus manatus latirostris), 2016* (Scientific Investigations Report). Reston, VA: U.S. Department of the Interior, U.S. Geological Survey.

Russell, D. J. and G. H. Balazs. (2015). Increased use of non-native algae species in the diet of the green turtle (*Chelonia mydas*) in a primary pasture ecosystem in Hawaii. *Aquatic Ecosystem Health & Management 18* (3): 342–346.

Russell, I. C., M. W. Aprahamian, J. Barry, I. C. Davidson, P. Fiske, A. T. Ibbotson, R. J. Kennedy, J. C. Maclean, A. Moore, J. Otero, T. Potter, and C. D. Todd. (2012). The influence of the freshwater environment and the biological characteristics of Atlantic salmon smolts on their subsequent marine survival. *ICES Journal of Marine Science: Journal du Conseil*. DOI:doi:10.1093/icesjms/fsr208

Rutter, J. E. (2016). *Bird Friendly Beaches: Evaluating dog and human interactions with Great Lakes piping plovers (Charadrius melodus) and other shorebirds.* (Master of Science). University of Minnesota.

Ryan, C., O. Boisseau, A. Cucknell, M. Romagosa, A. Moscrop, and R. McLanaghan. (2013). *Final Report for the trans-Atlantic Research Passages Between the UK and USA via the Azores and Iceland, Conducted from R/V Song of the Whale 26 March to 28 September 2012*. Essex, United Kingdom: Marine Conservation Research International.

Rycyk, A. M., C. J. Deutsch, M. E. Barlas, S. K. Hardy, K. Frisch, E. H. Leone, and D. P. Nowacek. (2018). Manatee behavioral response to boats. *Marine Mammal Science 34* (4): 924–962. DOI:10.1111/mms.12491

S&P Global Commodity Insights. (2023). *US offshore wind power development expanding beyond the East Coast in 2023*. Retrieved August 16, 2023, from https://www.spglobal.com/commodityinsights/en/market-insights/latest-news/energy-transition/051723-us-offshore-wind-power-development-expandin%E2%80%A6.

Sadovy, Y. and A.-M. Eklund. (1999). *NOAA Technical Report NMFS 146, Synopsis of Biological Data on the Nassau Grouper, Epinephelus striatus, (Block, 1972), and the Jewfish, E. itajara (Lichtenstein, 1822)* (A Technical Report of the *Fishery Bulletin*, FAO Fisheries Synopsis). Seattle, WA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Sakamoto, W., K. Sato, H. Tanaka, and Y. Naito. (1993). Diving patterns and swimming environment of two loggerhead turtles during internesting. *Nippon Suisan Gakkaishi 59* (7): 1129–1137.

Salisbury, D. P., C. W. Clark, and A. N. Rice. (2015). Right whale occurrence in the coastal waters of Virginia, U.S.A.: Endangered species presence in a rapidly developing energy market. *Marine Mammal Science 32* (2): 508–519. DOI:10.1111/mms.12276

Salmon, M., T. T. Jones, and K. W. Horch. (2004). Ontogeny of diving and feeding behavior in juvenile seaturtles: Leatherback seaturtles (*Dermochelys coriacea* L) and green seaturtles (*Chelonia mydas* L) in the Florida current. *Journal of Herpetology 38* (1): 36–43.

Salvadeo, C. J., D. Lluch-Belda, A. Gómez-Gallardo, J. Urbán-Ramírez, and C. D. MacLeod. (2010). Climate change and a poleward shift in the distribution of the Pacific white-sided dolphin in the northeastern Pacific. *Endangered Species Research 11* 13–19. DOI:10.3354/esr00252

Sampson, L. and A. Giraldo. (2014). Annual abundance of salps and doliolids (*Tunicata*) around Gorgona Island (Colombian Pacific), and their importance as potential food for green sea turtles. *Revista de Biología Tropical 62* 149–159.

Sánchez-Bayo, F. and K. A. G. Wyckhuys. (2019). Worldwide decline of the entomofauna: A review of its drivers. *Biological Conservation 232* 8-27. DOI:https://doi.org/10.1016/j.biocon.2019.01.020

Santos, M. B., V. Martin, M. Arbelo, A. Fernandez, and G. J. Pierce. (2007). Insights into the diet of beaked whales from the atypical mass strandings in the Canary Islands in September 2002. *Journal of the Marine Biological Association of the United Kingdom 87* 243–251. DOI:10.1017/S0025315407054380

Santos, M. B. and G. J. Pierce. (2003). The diet of harbor porpoise (*Phocoena phocoena*) in the northeast Atlantic. *Oceanography and Marine Biology: An Annual Review 41* 355–390.

Sasso, C., P. Richards, S. Benson, M. Judge, N. Putman, D. Snodgrass, and B. Stacy. (2021). Leatherback turtles in the eastern Gulf of Mexico: Foraging and migration behavior during the autumn and winter. *Frontiers in Marine Science 8* (660798): 1–8. DOI:10.3389/fmars.2021.660798

Savannah River Ecology Laboratory. (2023). *American Alligator (Alligator mississippiensis)*. Retrieved August 29, 2023, from https://srelherp.uga.edu/alligators/allmis.htm.

Savannah River Ecology Laboratory and Herpetology Program. (2012). *Species Profile: American Alligator (Alligator mississippiensis)*. Retrieved from http://srelherp.uga.edu/alligators/allmis.htm.

Savoy, T. and D. Pacileo. (2003). Movements and important habitats of subadult Atlantic sturgeon in Connecticut waters. *Transactions of the American Fisheries Society 132* (1): 1–8. DOI:10.1577/1548-8659(2003)132<0001:MAIHOS>2.0.CO;2

Schaefer, J., N. Frazier, and J. Barr. (2016). Dynamics of near-coastal fish assemblages following the Deepwater Horizon oil Spill in the northern Gulf of Mexico. *Transactions of the American Fisheries Society 145* (1): 108–119. DOI:10.1080/00028487.2015.1111253

Scharer, R. M., W. F. Patterson III, J. K. Carlson, and G. R. Poulakis. (2012). Age and growth of endangered smalltooth sawfish (*Pristis pectinata*) verified with LA-ICP-MS analysis of vertebrae. *PLoS ONE 7* (10): e47850. DOI:10.1371/journal.pone.0047850

Schiel, D. R., J. R. Steinbeck, and M. S. Foster. (2004). Ten years of induced ocean warming causes comprehensive changes in marine benthic communities. *Ecology 85* (7): 1833–1839.

Schmidly, D. J. (1981). *Marine Mammals of the Southeastern United States Coast and the Gulf of Mexico*. College Station, TX: Texas A&M University.

Schneider, D. C. and P. M. Payne. (1983). Factors affecting haul-out of harbor seals at a site in southeastern Massachusetts. *Journal of Mammalogy 64* (3): 518–520.

Schnitzler, H. U., C. F. Moss, and A. Denzinger. (2003). From spatial orientation to food acquisition in echolocating bats. *Trends in Ecology and Evolution 18* (8): 386–394. DOI:10.1016/S0169-5347(03)00185-X

Schoeman, R. P., C. Patterson-Abrolat, and S. Plön. (2020). A global review of vessel collisions with marine animals. *Frontiers in Marine Science 7*. DOI:10.3389/fmars.2020.00292

Schopmeyer, S. A., D. Lirman, E. Bertels, J. Byrne, D. S. Gilliam, J. Hunt, M. E. Johnson, E. A. Larson, K. Maxwell, K. Nedimeyer, and C. Walter. (2011). In Situ Coral Nurseries Serve as Genetic Repositories for Coral Reef Restoration after an Extreme Cold-Water Event. *Restoration Ecology 20* (6): 696–703. DOI:10.1111/j.1526-100X.2011.00836

Schramm, K. D., M. J. Marnane, T. S. Elsdon, C. M. Jones, B. J. Saunders, S. J. Newman, and E. S. Harvey. (2021). Fish associations with shallow water subsea pipelines compared to surrounding reef and soft sediment habitats. *Scientific Reports 11* (1): 6238. DOI:10.1038/s41598-021-85396-y

Schreiber, R. and J. Chovan. (1986). Roosting by pelagic seabirds: Energetic, populational, and social considerations. *The Condor 88* 487–492.

Schroeder, B. A. and N. B. Thompson. (1987). *Distribution of the Loggerhead Turtle, Caretta caretta, and the Leatherback Turtle, Dermochelys coriacea, in the Cape Canaveral, Florida Area: Results of Aerial Surveys* (Proceedings of the Cape Canaveral, Florida Sea Turtle Workshop). Miami, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Schubiger, V. (2023, June 27). *Crocodile Lifespan: How Long Do Crocodiles Live?* Retrieved August 23, 2023, from https://a-z-animals.com/blog/crocodile-lifespan-how-long-do-crocodiles-live/.

Schuerch, M., T. Spencer, S. Temmerman, M. L. Kirwan, C. Wolff, D. Lincke, C. J. McOwen, M. D. Pickering, R. Reef, A. T. Vafeidis, J. Hinkel, R. J. Nicholls, and S. Brown. (2018). Future response of global coastal wetlands to sea-level rise. *Nature 561* (7722): 231–234. DOI:10.1038/s41586-018-0476-5

Schuhmacher, H. and H. Zibrowius. (1985). What is hermatypic? A redefinition of ecological groups in corals and other organisms. *Coral Reefs 4* (1): 1–9. DOI:10.1007/BF00302198

Schuyler, Q., B. D. Hardesty, C. Wilcox, and K. Townsend. (2014). Global analysis of anthropogenic debris ingestion by sea turtles. *Conservation Biology 28* (1): 129–139.

Schwacke, L. H., T. A. Marques, L. Thomas, C. G. Booth, B. C. Balmer, A. Barratclough, K. Colegrove, S. De Guise, L. P. Garrison, F. M. Gomez, J. S. Morey, K. D. Mullin, B. M. Quigley, P. E. Rosel, T. K. Rowles, R. Takeshita, F. I. Townsend, T. R. Speakman, R. S. Wells, E. S. Zolman, and C. R. Smith. (2022). Modeling population effects of the Deepwater Horizon oil spill on a long-lived species. *Conservation Biology 36* (4): e13878. DOI:10.1111/cobi.13878

Schwartz, F. J. (1989). *Zoogeography and Ecology of Fishes Inhabiting North Carolina's Marine Waters to Depths of 600 Meters*. Silver Spring, MD: National Oceanic and Atmospheric Administration.

Scott, M. D. and S. J. Chivers. (1990). Distribution and herd structure of bottlenose dolphins in the eastern tropical Pacific Ocean. In S. Leatherwood & R. R. Reeves (Eds.), *The Bottlenose Dolphin* (pp. 387–402). Cambridge, MA: Academic Press.

Sea Turtle Sightings Hotline. (2023). *Sightings by Species Map*. Retrieved August 18, 2023, from https://seaturtlesightings.org/speciesmap.html.

Sears, R. and W. F. Perrin. (2009). Blue whale. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 120–124). Cambridge, MA: Academic Press.

Secor, D. H., E. J. Niklitschek, J. T. Stevenson, T. E. Gunderson, S. P. Minkkinen, B. Richardson, B. Florence, M. Mangold, J. Skjeveland, and A. Henderson-Arzapalo. (2000). Dispersal and growth of yearling Atlantic sturgeon, *Acipenser oxyrinchus*, released into Chesapeake Bay. *Fishery Bulletin 98* (4): 800–810.

Secor, D. H., M. H. P. O’Brien, N. Coleman, A. Horne, I. Park, D. C. Kazyak, D. G. Bruce, and C. Stence. (2022). Atlantic sturgeon status and movement ecology in an extremely small spawning habitat: The Nanticoke River-Marshyhope Creek, Chesapeake Bay. *Reviews in Fisheries Science & Aquaculture 30* (2): 195–214. DOI:10.1080/23308249.2021.1924617

Seitz, J. C. and G. R. Poulakis. (2006). Anthropogenic effects on the smalltooth sawfish (*Pristis pectinata*) in the United States. *Marine Pollution Bulletin 52* (11): 1533–1540.

Selzer, L. A. and P. M. Payne. (1988). The distribution of white-sided (*Lagenorhynchus acutus*) and common dolphins (*Delphinus delphis*) vs. environmental features of the continental shelf of the northeastern United States. *Marine Mammal Science 4* (2): 141–153.

Seminoff, J. A., C. D. Allen, G. H. Balazs, P. H. Dutton, T. Eguchi, H. L. Haas, S. A. Hargrove, M. P. Jensen, D. L. Klemm, A. M. Lauritsen, S. L. MacPherson, P. Opay, E. E. Possardt, S. L. Pultz, E. E. Seney, K. S. Van Houtan, and R. S. Waples. (2015). *Status Review of the Green Turtle (Chelonia mydas) Under the U.S. Endangered Species Act*. La Jolla, CA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southwest Fisheries Science Center.

Seminoff, J. A. and Marine Turtle Specialist Group Green Turtle Task Force. (2004). *Marine Turtle Specialist Group Review: 2004 Global Status Assessment, Green turtle (Chelonia mydas)*. Gland, Switzerland: The World Conservation Union (IUCN) Species Survival Commission, Red List Programme.

Seminoff, J. A., A. Resendiz, and W. J. Nichols. (2002). Home range of green turtles, *Chelonia mydas,* at a coastal foraging area in the Gulf of California, Mexico. *Marine Ecology Progress Series 242* 253–265.

Semmens, B. X., K. E. Luke, P. G. Bush, C. M. R. McCoy, and B. C. Johnson. (2006). Isopod infestation of postspawning Nassau grouper around Little Cayman Island. *Journal of Fish Biology 69* 933–937.

Seney, E. E. (2016). Diet of Kemp's ridley sea turtles incidentally caught on recreational fishing gear in the northwestern Gulf of Mexico. *Chelonian Conservation and Biology 15* (1): 132–137.

Serviere-Zaragoza, E., A. Mazariegos-Villareal, and D. Aldana Aranda. (2009). *Preliminary observation of natural feed of Queen Conch Strombus gigas*. Presented at the Proceedings of the 61st Gulf and Caribbean Fisheries Institute, November 10-14, 2008. Gosier, Guadeloupe, French West Indies.

Servis, J. A., G. Lovewell, and A. D. Tucker. (2015). Diet analysis of subadult Kemp's ridley (*Lepidochelys kempii*) turtles from west-central Florida. *Chelonian Conservation and Biology 14* (2): 173–181.

Shackelford, C., E. Rozenburg, W. Hunter, and M. Lockwood. (2005). *Migration and the Migratory Birds of Texas: Who They Are and Where They Are Going* (Fourth ed.). Austin, TX: Texas Parks and Wildlife.

Shane, S. H., R. S. Wells, and B. Wursig. (1986). Ecology, behavior and social organization of the bottlenose dolphin: A review. *Marine Mammal Science 2* (1): 34–63.

Sharp, S. M., W. A. McLellan, D. S. Rotstein, A. M. Costidis, S. G. Barco, K. Durham, T. D. Pitchford, K. A. Jackson, P. Y. Daoust, T. Wimmer, E. L. Couture, L. Bourque, T. Frasier, B. Frasier, D. Fauquier, T. K. Rowles, P. K. Hamilton, H. Pettis, and M. J. Moore. (2019). Gross and histopathologic diagnoses from North Atlantic right whale *Eubalaena glacialis* mortalities between 2003 and 2018. *Diseases of Aquatic Organisms 135* (1): 1–31.

Shaver, D. (2018, May 2). Personal communication via email from Donna Shaver (National Park Service) to Taylor Houston (ManTech International) regarding PAIS sea turtle nesting data.

Shaver, D., H. Frandsen, J. Walker, J. George, and C. Gredzens. (2020a). Threats to Kemp’s ridley sea turtle (*Lepidochelys kempii* Garman, 1880) nests incubating in situ on the Texas coast. *Herpetology Notes 13* 907–923.

Shaver, D. J. and C. W. Caillouet, Jr. (1998). More Kemp's Ridley turtles return to South Texas to nest. *Marine Turtle Newsletter 82* 1–5.

Shaver, D. J., H. R. Frandsen, J. A. George, and C. Gredzens. (2020b). Green turtle (*Chelonia mydas*) nesting underscores the importance of protected areas in the northwestern Gulf of Mexico. *Frontiers in Marine Science 7* 1–14. DOI:10.3389/fmars.2020.00673

Shaver, D. J., K. M. Hart, I. Fujisaki, D. Bucklin, A. R. Iverson, C. Rubio, T. F. Backof, P. M. Burchfield, R. J. G. D. Miron, P. H. Dutton, A. Frey, J. Peña, D. Gomez Gamez, H. J. Martinez, and J. Ortiz. (2017). Inter-nesting movements and habitat-use of adult female Kemp's ridley turtles in the Gulf of Mexico. *PLoS ONE 12* (3): 1–27. DOI:10.1371/journal.pone.0174248

Shaver, D. J., K. M. Hart, I. Fujisaki, C. Rubio, A. R. Sartain-Iverson, J. Peña, D. G. Gamez, R. J. G. D. Miron, P. M. Burchfield, and H. J. Martinez. (2016). Migratory corridors of adult female Kemp's ridley turtles in the Gulf of Mexico. *Biological Conservation 194* 158–167.

Sheehan, T. F., D. G. Reddin, G. Chaput, and M. D. Renkawitz. (2012). SALSEA North America: A pelagic ecosystem survey targeting Atlantic salmon in the Northwest Atlantic. *ICES Journal of Marine Science 69* (9): 1580–1588.

Sheridan, C. M., J. R. Spotila, W. F. Bien, and H. W. Avery. (2010). Sex-biased dispersal and natal philopatry in the diamondback terrapin, *Malaclemys terrapin*. *Molecular Ecology 19* (24): 5497–5510.

Shields, M., A. Poole, and F. Gill. (2002). Brown Pelican (Pelecanus occidentalis). *The Birds of North America Online* (609): 5. DOI:10.2173/bna.609

Shirasago-Germán, B., E. L. Pérez-Lezama, E. A. Chávez, and R. García-Morales. (2015). Influence of El Niño-Southern Oscillation on the population structure of a sea lion breeding colony in the Gulf of California. *Estuarine, Coastal and Shelf Science 154* 69–76. DOI:10.1016/j.ecss.2014.12.024

Shoop, C. R. and R. D. Kenney. (1992). Seasonal distributions and abundances of loggerhead and leatherback sea turtles in waters of the northeastern United States. *Herpetological Monographs 6* 43–67.

Sibley, D. (2014). *The Sibley Guide to Birds* (Second ed.). New York, NY: Alfred A. Knopf.

Siegel-Causey, D. and S. Kharitonov. (1990). The evolution of coloniality. *Current Ornithology 7* 285–330.

Silber, G., J. Slutsky, and S. Bettridge. (2010). Hydrodynamics of a ship/whale collision. *Journal of Experimental Marine Biology and Ecology 391* 10–19. DOI:10.1016/j.jembe.2010.05.013

Silber, G. K., M. D. Lettrich, P. O. Thomas, J. D. Baker, M. Baumgartner, E. A. Becker, P. Boveng, D. M. Dick, J. Fiechter, J. Forcada, K. A. Forney, R. B. Griffis, J. A. Hare, A. J. Hobday, D. Howell, K. L. Laidre, N. Mantua, L. Quakenbush, J. A. Santora, K. M. Stafford, P. Spencer, C. Stock, W. Sydeman, K. Van Houtan, and R. S. Waples. (2017). Projecting Marine Mammal Distribution in a Changing Climate. *Frontiers in Marine Science 4* 14. DOI:10.3389/fmars.2017.00413

Simard, Y., N. Roy, S. Giard, and F. Aulanier. (2019). North Atlantic right whale shift to the Gulf of St. Lawrence in 2015, revealed by long-term passive acoustics. *Endangered Species Research 40* 271–284.

Simmonds, M. P. and W. J. Eliott. (2009). Climate change and cetaceans: Concerns and recent developments. *Journal of the Marine Biological Association of the United Kingdom 89* (1): 203–210. DOI:10.1017/s0025315408003196

Simmons, A. M. and J. A. Simmons. (2024). Echolocating Bats Have Evolved Decreased Susceptibility to Noise-Induced Temporary Hearing Losses. *Journal of the Association for Research in Otolaryngology* 1-10. DOI:10.1007/s10162-024-00941-6

Simons, T. R., D. S. Lee, and J. C. Haney. (2013). Diablotin Pterodroma hasitata: A biography of the endangered black-capped petrel. *Marine Ornithology: Journal of Seabird Research and Conservation 41* (Special Issue): 1–43.

Simpfendorfer, C. A. (2006). *Movement and Habitat Use of Smalltooth Sawfish*. Sarasota, FL: Mote Marine Laboratory, Center for Shark Research.

Simpfendorfer, C. A., G. R. Poulakis, P. M. O'Donnell, and T. R. Wiley. (2008). Growth rates of juvenile smalltooth sawfish *Pristis pectinata* Latham in the western Atlantic. *Journal of Fish Biology 72* (3): 711-723. DOI:https://doi.org/10.1111/j.1095-8649.2007.01764.x

Simpfendorfer, C. A. and T. R. Wiley. (2005). *Determination of the Distribution of Florida's Remnant Sawfish Population and Identification of Areas Critical to Their Conservation*. Sarasota, FL: Mote Marine Laboratory, Center for Shark Research.

Simpfendorfer, C. A., B. G. Yeiser, T. R. Wiley, G. R. Poulakis, P. W. Stevens, and M. R. Heupel. (2011). Environmental influences on the spatial ecology of juvenile smalltooth sawfish (*Pristis pectinata*): Results from acoustic monitoring. *PLoS ONE 6* (2): e16918. DOI:10.1371/journal.pone.0016918

Sims, D. W., E. J. Southall, G. A. Tarling, and J. D. Metcalfe. (2005). Habitat-specific normal and reverse diel vertical migration in the plankton-feeding basking shark. *Journal of Animal Ecology 74* (4): 755–761.

Širović, A., H. R. Bassett, S. C. Johnson, S. M. Wiggins, and J. A. Hildebrand. (2014). Bryde's whale calls recorded in the Gulf of Mexico. *Marine Mammal Science 30* (1): 399–409. DOI:10.1111/mms.12036

Širović, A., J. A. Hildebrand, S. M. Wiggins, M. A. McDonald, S. E. Moore, and D. Thiele. (2004). Seasonality of blue and fin whale calls and the influence of sea ice in the Western Antarctic Peninsula. *Deep Sea Research II 51* (17–19): 2327–2344. DOI:10.1016/j.dsr2.2004.08.005

Sloan, K. A., D. S. Addison, A. T. Glinsky, A. M. Benscoter, and K. M. Hart. (2022). Inter-Nesting Movements, Migratory Pathways, and Resident Foraging Areas of Green Sea Turtles (*Chelonia*

mydas) Satellite-Tagged in Southwest Florida. *Frontiers in Marine Science 8* (775367): 1-14. DOI:10.3389/fmars.2021.775367

Slone, D. H., J. P. Reid, R. K. Bonde, S. M. Butler, and B. M. Stith. (2006). *Summary of the West Indian manatee (Trichechus manatus) tracking by U.S. Geological Survey-Florida Integrated Science Center Sirenia Project in Puerto Rico* (Report prepared for the U.S. Fish and Wildlife Service). Gainesville, FL: U.S. Coast Guard Florida Integrated Science Center and ASci Corporation.

Small, C. and R. J. Nicholls. (2003). A global analysis of human settlement in coastal zones. *Journal of Coastal Research 19* (3): 584–599.

Smith, A., T. Stehly, and W. Musial. (2015). *2014 U.S. Offshore Wind Market Report: Industry Trends, Technology Advancement, and Cost Reduction*. Baltimore, MD: National Renewable Energy Laboratory.

Smith, B. D., G. Braulik, S. Strindberg, R. Mansur, M. A. A. Diyan, and B. Ahmed. (2009). Habitat selection of freshwater-dependent cetaceans and the potential effects of declining freshwater flows and sea-level rise in waterways of the Sundarbans mangrove forest, Bangladesh. *Aquatic Conservation: Marine and Freshwater Ecosystems 19* (2): 209–225. DOI:10.1002/aqc.987

Smith, C. E., B. J. Hurley, C. N. Toms, A. D. Mackey, M. Solangi, and S. A. Kuczaj Ii. (2013). Hurricane impacts on the foraging patterns of bottlenose dolphins *Tursiops truncatus* in Mississippi Sound. *Marine Ecology Progress Series 487* 231–244. DOI:10.3354/meps10372

Smith, G. M. and C. W. Coates. (1938). Fibro-epithelial growths of the skin in large marine turtles, *Chelonia mydas* (Linnaeus). *Zoologica New York 23* 93-98.

Smith Jr., W. O. (1981). Photosynthesis and productivity of benthic macroalgae on the North Carolina continental shelf. *Botanica Marina 24* (5): 279–284. DOI:doi:10.1515/botm.1981.24.5.279

Smith, L. M., J. A. Gore, T. J. Doonan, and C. J. Campbell. (2022a). Tricolored bats at a southern range edge exhibit partial migration northward in autumn. *Movement Ecology 10* (56): 1–13. DOI:10.1186/s40462-022-00358-x

Smith, N. S. and I. M. Côté. (2021). Biotic resistance on coral reefs? Direct and indirect effects of native predators and competitors on invasive lionfish. *Coral Reefs 40* (4): 1127-1136. DOI:10.1007/s00338-021-02117-7

Smith, S. E., D. M. Brown, J. R. Oliveras, P. L. Sieswerda, S. Ahearn, and D. Reiss. (2022b). A preliminary study on humpback whales lunge feeding in the New York Bight, United States. *Frontiers in Marine Science 9*. DOI:10.3389/fmars.2022.798250

Smith, T. I. J. and J. P. Clugston. (1997). Status and management of Atlantic sturgeon, *Acipenser oxyrinchus*, in North America. *Environmental Biology of Fishes 48* 335–346.

Smultea, M. A., T. A. Jefferson, and A. M. Zoidis. (2010). Rare sightings of a Bryde's whale (*Balaenoptera edeni*) and Sei whales (*B. borealis*) (Cetacea: Balaenopteridae) northeast of Oahu, Hawaii. *Pacific Science 64* (3): 449–457. DOI:10.2984/64.3.449

Soldevilla, M. S., A. J. Debich, I. Pérez-Carballo, S. Jarriel, K. E. Frasier, L. P. Garrison, A. Gracia, J. A. Hildebrand, P. E. Rosel, and A. Serrano. (2024). Rice's whale occurrence in the western Gulf of Mexico from passive acoustic recordings. *Marine Mammal Science n/a* (n/a). DOI:https://doi.org/10.1111/mms.13109

Soldevilla, M. S., J. A. Hildebrand, K. E. Frasier, L. Aichinger Dias, A. Martinez, K. D. Mullin, P. E. Rosel, and L. P. Garrison. (2017). Spatial distribution and dive behavior of Gulf of Mexico Bryde's whales: Potential risk of vessel strikes and fisheries interactions. *Endangered Species Research 32* 533–550.

Solick, D. and C. Newman. (2021). Oceanic records of North American bats and implications for offshore wind energy development in the United States. *Ecology and Evolution 11* 14433-14447. DOI:10.1002/ece3.8175

Somerfield, P. J., W. C. Jaap, K. R. Clarke, M. Callahan, K. Hackett, J. Porter, M. Lybolt, C. Tsokos, and G. Yanev. (2008). Changes in coral reef communities among the Florida Keys, 1996–2003. *Coral Reefs 27* 951–965. DOI:10.1007/s00338-008-0390-7

Soulen, B. K., K. Cammen, T. F. Schultz, and D. W. Johnston. (2013). Factors Affecting Harp Seal (*Pagophilus groenlandicus*) Strandings in the Northwest Atlantic. *PLoS ONE 8* (7): e68779. DOI:10.1371/journal.pone.0068779

South Atlantic Fishery Management Council. (2002). *Fishery management plan for pelagic Sargassum habitat of the South Atlantic Region.* North Charleston, SC: South Atlantic Fishery Management Council.

South Atlantic Fishery Management Council. (2023). *Deep-water Coral HAPCs*. Retrieved May 6, 2023, from https://safmc.net/managed-areas/deep-water-coral-hapcs/.

South Carolina Department of Natural Resources. (2024). *Bats in South Carolina - Northern Long-eared Bat*. Retrieved May 30, 2024, from https://www.dnr.sc.gov/wildlife/bats/NLEB.html.

Southern California Marine Institute. (2016). *Southern California Offshore Banks, National Marine Sanctuary Proposal*. Los Angeles, CA: Marine Institute and the Vantuna Research Group and Moore Laboratory of Zoology, Occidental College.

Southwood, A. L., R. D. Andrews, M. E. Lutcavage, F. V. Paladino, N. H. West, R. H. George, and D. R. Jones. (1999). Heart rates and diving behavior of leatherback sea turtles in the eastern Pacific Ocean. *The Journal of Experimental Biology 202* 1115–1125.

Sowers, D. (2020). *Mapping and Geomorphic Characterization of Deep Sea Coral Ecosystems in the South Atlantic Region*. Presented at the NOAA Office of Ocean Exploration and Research. Silver Spring, MD.

Spalding, M. D., C. Ravilious, and E. P. Green. (2001). *World Atlas of Coral Reefs*. Berkeley, CA: University of California Press.

Spotila, J. R., A. E. Dunham, A. J. Leslie, A. C. Steyermark, P. T. Plotkin, and F. V. Paladino. (1996). Worldwide population decline of *Dermochelys coriacea*: Are leatherback turtles going extinct? *Chelonian Conservation and Biology 2* (2): 209–222.

Stallings, C. D. (2009). Fishery-independent data reveal negative effect of human population density on Caribbean predatory fish communities. *PLoS ONE 4* (5): 1–9.

Stanistreet, J., L. E. Hodge, and A. Read. (2012). *Passive Acoustic Monitoring for Marine Mammals at Site A in the Cape Hatteras Survey Area, March – April 2012*. Beaufort, NC: Department of the Navy.

Stanistreet, J. E., L. E. W. Hodge, D. P. Nowacek, J. T. Bell, J. A. Hildebrand, S. M. Wiggins, and A. J. Read. (2013). *Passive acoustic monitoring of beaked whales and other cetaceans off Cape Hatteras, North Carolina*. Presented at the 20th Biennial Conference on the Biology of Marine Mammals. Dunedin, New Zealand.

Stantial, M., J. Cohen, A. Darrah, S. Farrell, and B. Maslo. (2021). The effect of top predator removal on the distribution of a mesocarnivore and nest survival of an endangered shorebird. *Avian Conservation and Ecology 16* (1): 1-15. DOI:10.5751/ACE-01806-160108

Starr, R. M., E. Sala, E. Ballesteros, and M. Zabala. (2007). Spatial dynamics of the Nassau grouper, *Epinephelus striatus*, in a Caribbean atoll. *Marine Ecology Progress Series 343* 239–249.

State of the World's Sea Turtles. (2022). *Hawksbill Turtles*. Ross, CA: Oceanic Society.

State University of New York College of Environmental Science and Forestry. (2024). *Northern long-eared bat*. Retrieved May 30, 2024, from https://www.esf.edu/aec/adks/mammals/keens_myotis.php.

Stein, A. B., K. D. Friedland, and M. Sutherland. (2004). Atlantic Sturgeon Marine Distribution and Habitat Use along the Northeastern Coast of the United States. *Transactions of the American Fisheries Society 133* 527–537.

Steneck, R. S., M. H. Graham, B. J. Bourque, D. Corbett, J. M. Erlandson, J. A. Estes, and M. J. Tegner. (2002). Kelp forest ecosystems: Biodiversity, stability, resilience and future. *Environmental Conservation 29* (4): 436–459. DOI:10.1017/S0376892902000322

Stenson, G. B., R. A. Myers, W. G. Warren, and I.-H. Ni. (1996). *Pup Production of Hooded Seals (Cystophora cristata) in the Northwest Atlantic*. Dartmouth, Canada: Northwest Atlantic Fisheries Organization Scientific Council Studies.

Stepanuk, J. E., E. I. Heywood, J. F. Lopez, R. A. DiGiovanni Jr, and L. H. Thorne. (2021). Age-specific behavior and habitat use in humpback whales: Implications for vessel strike. *Marine Ecology Progress Series 663* 209–222.

Stevens, E. and C. Pickett. (1994). Managing the Social Environments of Flamingos for Reproductive Success. *Zoo Biology 13* 501–507.

Stevens, P. W., D. A. Blewett, and J. P. Casey. (2006). Short-term effects of a low dissolved oxygen event on estuarine fish assemblages following passage of Hurricane Charley. *Estuaries and Coasts 29* (6A): 997–1003.

Stevick, P. T., J. Allen, P. J. Clapham, N. Friday, S. K. Katona, F. Larsen, J. Lien, D. K. Mattila, P. J. Palsboll, J. Sigurjonsson, T. D. Smith, N. Oien, and P. S. Hammond. (2003). North Atlantic humpback whale abundance and rate of increase four decades after protection from whaling. *Marine Ecology Progress Series 258* 263–273.

Stevick, P. T., J. Allen, P. J. Clapham, S. K. Katona, F. Larsen, J. Lien, D. K. Mattila, P. J. Palsboll, R. Sears, J. Sigurjonsson, T. D. Smith, G. Vikingsson, N. Oien, and P. S. Hammond. (2006). Population spatial structuring on the feeding grounds in North Atlantic humpback whales (*Megaptera novaeangliae*). *Journal of Zoology 270* 244–255. DOI:10.1111/j.1469-7998.2006.00128.x

Stewart, B. S. and S. Leatherwood. (1985). Minke whale, *Balaenoptera acutorostrata* Lacepede, 1804. In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 3, pp. 91–136). San Diego, CA: Academic Press.

Stewart, K., M. Sims, A. Meylan, B. Witherington, B. Brost, and L. B. Crowder. (2011). Leatherback nests increasing significantly in Florida, USA; Trends assessed over 30 years using multilevel modeling. *Ecological Applications 21* (1): 263–273.

Stewart, K. R., E. L. LaCasella, S. E. Roden, M. P. Jensen, L. W. Stokes, S. P. Epperly, and P. H. Dutton. (2016). Nesting population origins of leatherback turtles caught as bycatch in the US pelagic longline fishery. *Ecosphere 7* (3): 1–18.

Stewart, K. R., K. J. Martin, C. Johnson, N. Desjardin, S. A. Eckert, and L. B. Crowder. (2014). Increased nesting, good survival and variable site fidelity for leatherback turtles in Florida, USA. *Biological Conservation 176* 117–125.

Stoner, A. W. and M. Ray. (1993). Aggregation dynamics in juvenile queen conch (*Strombus gigas*): population structure, mortality, growth, and migration. *Marine Biology 116* (4): 571-582.

Stoner, A. W. and V. J. Sandt. (1992). Population structure, seasonal movements and feeding of queen conch, *Strombus gigas*, in deep-water habitats of the Bahamas. *Bulletin of Marine Science 51* (3): 287-300.

Stoner, A. W. and J. M. Waite. (1991). Trophic biology of *Strombus gigas* in nursery habitats: diets and food sources in seagrass meadows. *Journal of Molluscan Studies 57* (4): 451-460.

Straley, J. M. (1990). Fall and winter occurrence of humpback whales (*Megaptera novaeangliae*) in southeastern Alaska. *Reports of the International Whaling Commission Special Issue 12* 319–323.

Sulak, K. J. and M. Randall. (2002). Understanding sturgeon life history: Enigmas, myths, and insights from scientific studies. *Journal of Applied Ichthyology 18* 519–528.

Sulak, K. J., M. T. Randall, R. E. Edwards, T. M. Summers, K. E. Luke, W. T. Smith, A. D. Norem, W. M. Harden, R. H. Lukens, F. Parauka, S. Bolden, and R. Lehnert. (2009). Defining winter trophic habitat of juvenile Gulf sturgeon in the Suwannee and Apalachicola rivermouth estuaries, acoustic telemetry investigations. *Journal of Applied Ichthyology 25* 505–515.

Surrey-Marsden, C., C. Accardo, M. White, C. George, T. Gowan, P. K. Hamilton, K. Jackson, J. Jakush, T. Pitchford, and C. Taylor. (2018). *North Atlantic Right Whale Calving Area Surveys: 2016/2017 Results* (NOAA Technical Memorandum NMFS-SER, 8). St. Petersburg, FL: National Marine Fisheries Service Southeast Regional Office.

Sutherland, K. P., S. Shaban, J. L. Joyner, J. W. Porter, and E. K. Lipp. (2011). Human pathogen shown to cause disease in the threatened elkhorn coral, *Acropora palmata*. *PLoS ONE 6* (8): e23468. DOI:10.1371/journal.pone.0023468

Swaim, Z., H. Foley, and A. Read. (2015). *Protected Species Monitoring in Navy OPAREAs: Small Vessel Surveys in the Jacksonville Operating Area (January 2014-December 2014)*. Virginia Beach, VA: U.S. Fleet forces, Naval Facilities Engineering Command, Atlantic.

Swaim, Z., H. Foley, D. Waples, K. Urian, and A. Read. (2014). *Protected Species Monitoring in Navy OPAREAS off the U.S. Atlantic Coast, January 2013 – December 2013*. Norfolk, VA: U.S. Department of the Navy.

Swartz, S. L. and C. Burks. (2000). *Windwards Humpback (Megaptera novaeangliae) Survey*. Miami, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Swartz, S. L., A. Martinez, J. Stamates, C. Burks, and A. A. Mignucci-Giannoni. (2002). *Acoustic and Visual Survey of Cetaceans in the Waters of Puerto Rico and the Virgin Islands: February – March 2001*. Miami, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Swimmer, Y., A. Gutierrez, K. Bigelow, C. Barceló, B. Schroeder, K. Keene, K. Shattenkirk, and D. G. Foster. (2017). Sea turtle bycatch mitigation in U.S. longline fisheries. *Frontiers in Marine Science 4*. DOI:10.3389/fmars.2017.00260

Swingle, W. M., S. G. Barco, E. B. Bates, G. G. Lockhart, K. M. Phillips, K. R. Rodrique, S. A. Rose, and K. M. Williams. (2016). *Virginia Sea Turtle and Marine Mammal Stranding Network 2015 Grant Report*. Virginia Beach, VA: Virginia Aquarium Foundation.

Swingle, W. M., M. C. Lynott, E. B. Bates, L. R. D'Eri, G. G. Lockhart, K. M. Phillips, and M. D. Thomas. (2014). *Virginia Sea Turtle and Marine Mammal Stranding Network 2013 Grant Report* (Final Report to the Virginia Coastal Zone Management Program, National Oceanic and Atmospheric Administration Coastal Zone Management). Virginia Beach, VA: Virginia Aquarium Foundation Stranding Response Program.

Sydeman, W. J., E. S. Poloczanska, T. E. Reed, and S. A. Thompson. (2015). Climate change and marine vertebrates. *Science 350* (6262): 772–777. DOI:10.1126/science.aac9874

Taruski, A. G. and H. E. Winn. (1976). Winter sightings of odontocetes in the West Indies. *Cetology 22* 1–12.

Taylor, G. A. (2008). Maximum dive depths of eight New Zealand procellariiformes, including pterodroma species. *Papers and Proceedings of the Royal Society of Tasmania 142* (1): 89–99.

Tebbett, S. B., S. R. Connolly, and D. R. Bellwood. (2023). Benthic composition changes on coral reefs at global scales. *Nature Ecology & Evolution 7* (1): 71–81. DOI:10.1038/s41559-022-01937-2

Temte, J. L., M. A. Bigg, and O. Wiig. (1991). Clines revisited: The timing of pupping in the harbour seal (*Phoca vitulina*). *Journal of Zoology 224* 617–632.

Tetra Tech Inc. (2014). *Acoustic and Avian Radar Surveys for Birds and Bats NCTAMSLANT DET Cutler, Maine*. Norfolk, VA: Naval Facilities Engineering Command Mid-Atlantic.

Tetra Tech Inc. (2016a). *Bat Baseline Survey Report Joint Expeditionary Base Fort Story Virginia Beach, Virginia*. Norfolk, VA: Naval Facilities Engineering Command Mid-Atlantic.

Tetra Tech Inc. (2016b). *Bat Baseline Survey Report Naval Air Station Oceana Dam Neck Annex Virginia Beach, Virginia*. Norfolk, VA: Naval Facilities Engineering Command Mid-Atlantic.

Tetra Tech Inc. (2016c). *Bat Baseline Survey Report Naval Weapons Station Earle Monmouth County, New Jersey*. Norfolk, VA: Naval Facilities Engineering Command Mid-Atlantic,.

Tetra Tech Inc. (2016d). *Northern Long-Eared Bat Survey Report Naval Air Station Oceana Virginia Beach, Virginia*. Norfolk, VA: Naval Facilities Engineering Command Mid-Atlantic.

Tetra Tech Inc. (2017a). *Northern Long-Eared Bat Survey Report Naval Station Norfolk and Naval Supply Center Craney Island Fuel Terminal Norfolk and Portsmouth, Virginia*. Norfolk, VA: Naval Facilities Engineering Command Mid-Atlantic.

Tetra Tech Inc. (2017b). *Northern Long-Eared Bat Survey Report Naval Weapons Station Yorktown and Naval Supply Center Cheatham Annex Williamsburg, Virginia*. Norfolk, VA: Naval Facilities Engineering Command Mid-Atlantic.

Teuten, E. L., S. J. Rowland, T. S. Galloway, and R. C. Thompson. (2007). Potential for plastics to transport hydrophobic contaminants. *Environmental Science and Technology 41* (22): 7759–7764. DOI:10.1021/es071737s

Theile, S. (2001). *Queen Conch Fisheries and Their Management in the Caribbean*. Cambridge, United Kingdom: TRAFFIC Europe Brussels.

Theroux, R. B. and R. L. Wigley. (1998). *Quantitative composition and distribution of the macrobenthic invertebrate fauna of the Continental Shelf ecosystems of the Northeastern United States*. Seattle, WA: U.S Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Thiessen, G. J. (1958). Threshold of hearing of a ring-billed gull. *The Journal of the Acoustical Society of America 30* (11): 1047.

Thompson, R., A. Thompson, and R. Brigham. (2015). A flock of *Myotis* bats at sea. *Northeastern Naturalist 22* (4): N27–N30.

Thorbjarnarson, J., F. Mazzotti, E. Sanderson, F. Buitrago, M. Lazcano, K. Minkowski, M. Muniz, P. Ponce, L. Sigler, R. Soberon, A. M. Trelancia, and A. Velasco. (2006). Regional habitat conservation priorities for the American crocodile. *Biological Conservation 128* (1): 25–36. DOI:10.1016/j.biocon.2005.09.013

Thorn, C. D. and M. L. Falgiani. (2013). Jumping sturgeon in the Suwannee River. *Wilderness & Environmental Medicine 24* (3): 298-299. DOI:https://doi.org/10.1016/j.wem.2013.01.006

Thorne, L. H., L. W. Hodge, and A. J. Read. (2012, 20-24 February). *Combining passive acoustics and satellite oceanography to evaluate cetacean habitat use in the South Atlantic Bight*. Presented at the 2012 Ocean Sciences Meeting. Salt Lake City, UT.

Thorstad, E. B., C. D. Todd, I. Uglem, P. A. Bjørn, P. G. Gargan, K. W. Vollset, E. Halttunen, S. Kålås, M. Berg, and B. Finstad. (2015). Effects of salmon lice *Lepeophtheirus salmonis* on wild sea trout *Salmo trutta*—a literature review. *Aquaculture Environment Interactions 7* (2): 91–113. DOI:10.3354/aei00142

Tickler, D. M., T. B. Letessier, H. J. Koldewey, and J. J. Meeuwig. (2017). Drivers of abundance and spatial distribution of reef-associated sharks in an isolated atoll reef system. *PLoS ONE 12* (5): e0177374. DOI:10.1371/journal.pone.0177374

Tittensor, D. P., A. R. Baco, P. E. Brewin, M. R. Clark, M. Consalvey, J. Hall-Spencer, A. A. Rowden, T. Schlacher, K. I. Stocks, and A. D. Rogers. (2009). Predicting global habitat suitability for stony corals on seamounts. *Journal of Biogeography 36* 1111–1128. DOI:10.1111/j.1365-2699.2008.02062

Titus, J. G., D. E. Hudgens, D. L. Trescott, M. Craghan, W. H. Nuckols, C. H. Hershner, J. M. Kassakian, C. J. Linn, P. G. Merritt, T. M. McCue, J. F. O'Connell, J. Tanski, and J. Wang. (2009). State and local governments plan for development of most land vulnerable to rising sea level along the US Atlantic coast. *Environmental Research Letters 4* (4): 044008. DOI:10.1088/1748-9326/4/4/044008

Tolotti, M., R. Bauer, F. Forget, P. Bach, L. Dagorn, and P. Travassos. (2017). Fine-scale vertical movements of oceanic whitetip sharks (*Carcharhinus longimanus*). *Fishery Bulletin 115* (3): 380–395. DOI:10.7755/Fb.115.3.8

Townhill, B. L., J. Tinker, M. Jones, S. Pitois, V. Creach, S. D. Simpson, S. Dye, E. Bear, and J. K. Pinnegar. (2018). Harmful algal blooms and climate change: exploring future distribution changes. *ICES Journal of Marine Science 75* (6): 1882–1893. DOI:10.1093/icesjms/fsy113

Tracy, A. M., M. L. Pielmeier, R. M. Yoshioka, S. F. Heron, and C. D. Harvell. (2019a). Increases and decreases in marine disease reports in an era of global change. *Proceedings of the Royal Society B 286* (1912): 20191718. DOI:https://doi.org/10.1098/rspb.2019.1718

Tracy, J. L., T. Kantola, K. A. Baum, and R. N. Coulson. (2019b). Modeling fall migration pathways and spatially identifying potential migratory hazards for the eastern monarch butterfly. *Landscape Ecology 34* (2): 443-458. DOI:10.1007/s10980-019-00776-0

Traylor-Knowles, N., A. C. Baker, K. M. Beavers, N. Garg, J. R. Guyon, A. Hawthorn, N. J. MacKnight, M. Medina, L. D. Mydlarz, E. C. Peters, J. M. Stewart, M. S. Studivan, and J. D. Voss. (2022). Advances in coral immunity 'omics in response to disease outbreaks. *Frontiers in Marine Science 9*. DOI:10.3389/fmars.2022.952199

True, M., K. Gorman, H. Taylor, R. Reynolds, and W. Ford. (2023). Fall migration, oceanic movement, and site residency patterns of eastern red bats (Lasiurus borealis) on the mid-Atlantic Coast. *Movement Ecology 11* 1-16. DOI:10.1186/s40462-023-00398-x

Tsiamis, K., M. Salomidi, V. Gerakaris, A. O. M. Mogg, E. S. Porter, M. D. J. Sayer, and F. C. Küpper. (2020). Macroalgal vegetation on a north European artificial reef (Loch Linnhe, Scotland): biodiversity, community types and role of abiotic factors. *Journal of Applied Phycology 32* (2): 1353–1363. DOI:10.1007/s10811-019-01918-2

Tsipoura, N. and J. Burger. (1999). Shorebird diet during spring migration stopover on Delaware Bay. *The Condor 101* 635–644.

Tucker, A. D., B. D. MacDonald, and J. A. Seminoff. (2014). Foraging site fidelity and stable isotope values of loggerhead turtles tracked in the Gulf of Mexico and Northwest Caribbean. *Marine Ecology Progress Series 502* 267–279.

Tucker, S., W. Bowen, S. Iversen, W. Blanchard, and G. Stenson. (2009). Sources of variation in diets of harp and hooded seals estimated from quantitative fatty acid signature analysis (QFASA). *Marine Ecology Progress Series 384* 287–302.

Turtle Expert Working Group. (2007). *An assessment of the leatherback turtle population in the Atlantic Ocean*. Miami, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Tuuri, E. M. and S. C. Leterme. (2023). How plastic debris and associated chemicals impact the marine food web: A review. *Environmental Pollution 321* (121156): 1–12. DOI:10.1016/j.envpol.2023.121156

Twiss, J. R., Jr. and R. R. Reeves. (1999). *Conservation and Management of Marine Mammals*. Washington, DC: Smithsonian Institution Press.

U.S. Department of Energy. (2016). *Long-term Bat Monitoring on Islands, Offshore Structures, and Coastal Sites in the Gulf of Maine, Mid-Atlantic, and Great Lakes—Final Report*. Topsham, ME: Stantec.

U.S. Department of Energy. (2022). *Offshore Wind Market Report: 2022 Edition*. Washington, DC: U.S. Department of Energy, Office of Energy Efficiency & Renewable Energy.

U.S. Department of the Interior and Fish and Wildlife Service. (2022). Endangered and Threatened Wildlife and Plants; Endangered Species Status for Tricolored Bat. *Federal Register 87* (177): 56381–56393.

U.S. Department of the Navy. (2008). *Final Environmental Assessment for the Homeporting of Six Zumwalt Class Destroyers at East and West Coast Installations (including Hawaii)*. Norfolk, VA: Naval Facilities Engineering Command Atlantic.

U.S. Department of the Navy. (2010). *Jacksonville Range Complex Operating Area Undersea Warfare Training Range Bottom Mapping and Habitat Characterization, Florida. Final Cruise Report*. Jacksonville, FL: Naval Facilities Engineering Command Atlantic.

U.S. Department of the Navy. (2011). *Marine Species Monitoring for the U.S. Navy's Virginia Capes, Cherry Point and Jacksonville Range Complexes; Annual Report for 2010*. Norfolk, VA: United States Fleet Forces Command.

U.S. Department of the Navy. (2013a). *Comprehensive Exercise and Marine Species Monitoring Report for the U.S. Navy's Atlantic Fleet Active Sonar Training (AFAST) and Virginia Capes, Cherry Point, Jacksonville, and Gulf of Mexico Range Complexes 2009–2012*. Norfolk, VA: United States Fleet Forces Command.

U.S. Department of the Navy. (2013b). *Water Range Sustainability Environmental Program Assessment: Potomac River Test Range*. Dahlgren, VA: Naval Surface Warfare Center.

U.S. Department of the Navy. (2017). *Dive Distribution and Group Size Parameters for Marine Species Occurring in the U.S. Navy's Atlantic and Hawaii-Southern California Training and Testing Study Areas*. Newport, RI: Naval Undersea Warfare Center Division.

U.S. Fish and Wildlife Service. (1993). *Caribbean Roseate Tern Recovery Plan*. Atlanta, GA: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (1995). *Gulf Sturgeon Recovery/Management Plan*. Atlanta, GA: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (1996). *Piping Plover (Charadrius melodus) Atlantic Coast Population Revised Recovery Plan*. Hadley, MA: U.S. Fish and Wildlife Service, Region Five.

U.S. Fish and Wildlife Service. (1998). *Roseate Tern (Sterna dougallii) Northeastern Population Recovery Plan*. Hadley, MA.

U.S. Fish and Wildlife Service. (2005). *Regional Seabird Conservation Plan, Pacific Region*. Portland, OR: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2007). *Indiana Bat (Myotis sodalis) Draft Recovery Plan: First Revision*. Fort Snelling, MN: U.S. Fish and Wildlife Service, Great Lakes-Big Rivers Region - Region 3.

U.S. Fish and Wildlife Service. (2009a). *Abundance and Productivity Estimates Atlantic Coast Piping Plover Population, 1986–2009*. Washington, DC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2009b). *Indiana Bat (Myotis sodalis) 5-Year Review: Summary and Evaluation*. Bloomington, IN: U.S. Fish and Wildlife Service Midwest Region – Region 3, Bloomington Ecological Services Field Office.

U.S. Fish and Wildlife Service. (2009c). *Piping Plover (Charadrius melodus) 5-Year Review: Summary and Evaluation*. Hadley, MA: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2010a). *Caribbean Roseate Tern and North Atlantic Roseate Tern (Sterna dougallii dougallii) 5-Year Review: Summary and Evaluation*. Atlanta, GA: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2010b). *Endangered Species Program: Species Information*.

U.S. Fish and Wildlife Service. (2010c). *Red Knot (Calidris canutus rufa) Spotlight Species Action Plan*. Pleasantville, NJ: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2010d, May 28, 2010). *Species Profile: American crocodile Crocodylus acutus*. Retrieved from https://ecos.fws.gov/ecp0/profile/speciesProfile?spcode=C02J.

U.S. Fish and Wildlife Service. (2010e). *Species Profile: Roseate Tern (Sterna dougallii dougallii)* (Environmental Conservation Online System).

U.S. Fish and Wildlife Service. (2013). *Cahow or Bermuda Petrel (Pterodroma cahow) 5-Year Review: Summary and Evaluation*. Raleigh, NC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2014a). *Stock Assessment Report West Indian Manatee (Trichechus manatus) Puerto Rico Stock (Antillean subspecies, Trichechus manatus manatus)*. Boqueron, Puerto Rico: Caribbean Ecological Services Field Office.

U.S. Fish and Wildlife Service. (2014b). *West Indian Manatee (Trichechus manatus) Florida Stock (Florida subspecies, Trichechus manatus latirostris)*. Jacksonville, FL: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2015). *2015 Population Estimates for the Indiana Bat (Myotis sodalis) by U.S. Fish and Wildlife Service Region*. Washington, DC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2016a). *Endangered and Threatened Wildlife and Plants; 4(d) Rule for the Northern Long-Eared Bat*. Washington, DC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2016b). *Endangered and Threatened Wildlife and Plants; Determination That Designation of Critical Habitat Is Not Prudent for the Northern Long-Eared Bat*. Washington, DC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2016c). *Northern Long-eared Bats: A Federally Threatened Species*. Retrieved from https://forest.umaine.edu/wp-content/uploads/sites/231/2016/03/Northern-Long-eared-Bats-_A-Federally-Threatened-Species-Wende-Mahaney.pdf.

U.S. Fish and Wildlife Service. (2016d). *Programmatic Biological Opinion on Final 4(d) Rule for the Northern Long-Eared Bat and Activities Excepted from Take Prohibitions*. Bloomington, MN: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2017a). *Environmental Conservation Online System Species Profile for Northern long-eared Bat (Myotis septentrionalis)*. Retrieved February 2, 2017, from https://ecos.fws.gov/ecp0/profile/speciesProfile.action?spcode=A0JE.

U.S. Fish and Wildlife Service. (2017b). *Northern Long-Eared Bat Final 4(d) Rule White-Nose Syndrome Zone Around WNS/Pd Positive Counties/Districts (Figure)*. Washington, DC: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2018). *Species Status Assessment Report for the Black-capped petrel (Pterodroma hasitata)*. Atlanta, GA: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service. (2019). *Cahow or Bermuda Petrel (Pterodroma cahow) 5-Year Review: Summary and Evaluation*. Raleigh, NC: U.S. Fish and Wildlife Service, Southeast Region, Raleigh Ecological Services Field Office.

U.S. Fish and Wildlife Service. (2020a). *Northern Long-Eared Bat Final 4(d) Rule White-Nose Syndrome Zone Around WNS/Pd Positive Counties/Districts*. Retrieved September 19, 2023, from https://www.fws.gov/sites/default/files/documents/WNSZone.pdf.

U.S. Fish and Wildlife Service. (2020b). *Piping Plover (Charadrius melodus) 5-Year Review: Summary and Evaluation*. Retrieved from https://ecos.fws.gov/docs/tess/species_nonpublish/3383.pdf.

U.S. Fish and Wildlife Service. (2020c). *Roseate Tern Northeastern North American Population (Sterna dougallii dougallii) 5-Year Review: Summary and Evaluation*. Concord, NH.

U.S. Fish and Wildlife Service. (2020d). *Species Status Assessment Report for the Rufa Red Knot (Calidris canutus rufa)*. Galloway, NJ: U.S. Fish and Wildlife Service, Ecological Services New Jersey Field Office.

U.S. Fish and Wildlife Service. (2021a). *Birds of Conservation Concern 2021*. Washington, DC: U.S. Fish and Wildlife Service, Migratory Birds Program.

U.S. Fish and Wildlife Service. (2021b, November 19). *Northern Long-eared bat distribution in North Carolina*. Retrieved May 30, 2024, from https://www.fws.gov/media/northern-long-eared-bat-distribution-north-carolina.

U.S. Fish and Wildlife Service. (2022a). *American crocodile (Crocodylus acutus) 5-Year Review: Summary and Evaluation*. Vero Beach, FL: U.S. Fish and Wildlife Service, South Atlantic–Gulf and Mississippi Basin Regions, Florida Ecological Services Office.

U.S. Fish and Wildlife Service. (2022b). *Endangered Species Status for Tricolored Bat*. Retrieved August 22, 2023, from https://www.fws.gov/species-publication-action/endangered-species-status-tricolored-bat-0.

U.S. Fish and Wildlife Service. (2022c). *Proposal to List the tricolored bat as endangered*. Retrieved August 22, 2023, from https://www.fws.gov/press-release/2022-09/proposal-list-tricolored-bat-endangered.

U.S. Fish and Wildlife Service. (2022d). *Roseate Tern Caribbean Population (Sterna dougallii dougalii) Status Review: Summary and Evaluation*.

U.S. Fish and Wildlife Service. (2022e). *Saving the tricolored bat*. Retrieved August 22, 2023, from https://fws.gov/story/2022-09/saving-tricolored-bat.

U.S. Fish and Wildlife Service. (2022f). *Species Status Assessment Report for the Northern long-eared bat (Myotis septentrionalis)*. Bloomington, MN.

U.S. Fish and Wildlife Service. (2022g, September 13). *Tricolored Bat Frequently Asked Questions*. Retrieved August 22, 2023, from https://www.fws.gov/story/2022-09/tricolored-bat-frequently-asked-questions#:~:text=Tricolored.

U.S. Fish and Wildlife Service. (2023a, February 14). *New Jersey Municipalities with Hibernation or Maternity Occurrence of Indiana Bat or Northern Long-eared Bat*. Retrieved May 29, 2024, from https://www.fws.gov/media/new-jersey-municipalities-hibernation-or-maternity-occurrence-indiana-bat-or-northern-long.

U.S. Fish and Wildlife Service. (2023b). *Recovery Plan for the Rufa Red Knot (Calidris canutus rufa)*.

U.S. Fish and Wildlife Service. (2023c, December 27). *Service Finalizes Listing of Black-Capped Petrel as an Endangered Species*. Retrieved February 6, 2024, from https://www.fws.gov/press-release/2023-12/service-finalizes-listing-black-capped-petrel-endangered-species.

U.S. Fish and Wildlife Service. (2023d). *Species Profile for Tricolored bat (Perimyotis subflavus)*. Retrieved August 22, 2023, from https://ecos.fws.gov/ecp/species/10515.

U.S. Fish and Wildlife Service. (2023e). *Tricolored Bat (Perimyotis subflavus)*. Retrieved August 22, 2023, from https://fws.gov/species/tricolored-bat-perimyotis-subflavus.

U.S. Fish and Wildlife Service. (2023f). *West Indian Manatee (Trichechus manatus) Florida Stock (Florida subspecies, Trichechus manatus latirostris)* (Stock Assessment Report). Jacksonville, FL: U.S. Fish and Wildlife Service, Florida Ecological Services Office.

U.S. Fish and Wildlife Service. (2023g). *West Indian Manatee (Trichechus manatus) Puerto Rico Stock (Antillean subspecies, Trichechus manatus manatus)* (Stock Assessment Report). Boquerón, PR: U.S. Fish and Wildlife Service, Caribbean Ecological Services Field Office.

U.S. Fish and Wildlife Service. (2024a). *Environmental Conservation Online System: Northern Long-Eared Bat (Myotis septentrionalis)*. Retrieved May 30, 2024, from https://ecos.fws.gov/ecp/species/9045.

U.S. Fish and Wildlife Service. (2024b). *Indiana Bat*. Retrieved May 29, 2024, from https://www.fws.gov/species/indiana-bat-myotis-sodalis.

U.S. Fish and Wildlife Service. (2024c). *Indiana Bat Determination Key Standing Analysis*. Frankfort, KY: U.S. Fish and Wildlife Service, Kentucky Field Office. Retrieved from https://www.fws.gov/carp/sites/default/files/documents/indiana-bat-determination-key-standing-analysis.pdf.

U.S. Fish and Wildlife Service. (2024d). *Northern Long-eared Bat*. Retrieved May 30, 2024, from https://www.fws.gov/species/northern-long-eared-bat-myotis-septentrionalis.

U.S. Fish and Wildlife Service. (2024e). *Tricolored Bat*. Retrieved May 30, 2024, from https://www.fws.gov/species/tricolored-bat-perimyotis-subflavus.

U.S. Fish and Wildlife Service. (2024f). *White-nose Syndrome Response Team*. Retrieved May 29, 2024, from https://www.whitenosesyndrome.org/.

U.S. Fish and Wildlife Service and National Marine Fisheries Service. (2009). *Gulf Sturgeon (Acipenser oxyrinchus desotoi) 5-Year Review: Summary and Evaluation*. Panama City, FL: U.S. Fish and Wildlife Service.

U.S. Fish and Wildlife Service and National Marine Fisheries Service. (2022). *Gulf Sturgeon (Acipenser oxyrinchus desotoi) 5-Year Review: Summary and Evaluation*. USFWS South Atlantic-Gulf and Mississippi Basin Regions, Panama City Fish and Wildlife Conservation Office, and NMFS Southeast Region Protected Resources Division.

U.S. Fish and Wildlife Service Caribbean Field Office. (2009). *Stock Assessment West Indian Manatee (Trichechus manatus)*. Boqueron, Puerto Rico: U.S. Fish and Wildlife Service, Caribbean Ecological Services Field Office.

U.S. Geological Survey. (2007). *Data from the 2006 International Piping Plover Census*. Corvallis, OR: Corvallis Work Group.

Ulanovsky, N. and C. F. Moss. (2008). What the bat's voice tells the bat's brain. *Proceedings of the National Academy of Sciences 105* (25): 8491–8498. DOI:10.1073/pnas.0703550105

Ulanovsky, N. and C. F. Moss. (2011). Dynamics of hippocampal spatial representation in echolocating bats. *Hippocampus 21* (2): 150–161. DOI:10.1002/hipo.20731

University of Georgia. (2017). *Diamondback Terrapin (Malaclemys terrapin).* Retrieved December 12, 2017, from https://srelherp.uga.edu/turtles/malter.htm.

University of Georgia. (2023). *Diamondback Terrapin (Malaclemys terrapin)*. Retrieved January 27, 2023, from https://srelherp.uga.edu/turtles/malter.htm.

Urmy, S. S. and K. J. Benoit-Bird. (2021). Fear dynamically structures the ocean's pelagic zone. *Current Biology 31* (22): 5086-5092.e5083. DOI:10.1016/j.cub.2021.09.003

Vadas, R. L. and R. S. Steneck. (1988). Zonation of deep water benthic algae in the Gulf of Maine. *Journal of Phycology 24* (3): 338–346. DOI:https://doi.org/10.1111/j.1529-8817.1988.tb04476.x

Valverde, R. A. and K. R. Holzwart. (2017). Sea Turtles in the Gulf of Mexico. In C. H. Ward (Ed.), *Habitats and Biota of the Gulf of Mexico: Before the Deepwater Horizon Oil Spill* (pp. 1189–1351). New York, NY: Springer Nature.

van Denderen, P. D., S. G. Bolam, J. G. Hiddink, S. Jennings, A. Kenny, A. D. Rijnsdorp, and T. Van Kooten. (2015). Similar effects of bottom trawling and natural disturbance on composition and function of benthic communities across habitats. *Marine Ecology Progress Series 541* 31-43.

van Denderen, P. D., A. Törnroos, M. Sciberras, H. Hinz, R. Friedland, R. Lasota, M. C. Mangano, C. Robertson, S. Valanko, and J. G. Hiddink. (2022). Effects of bottom trawling and hypoxia on benthic invertebrate communities. *Marine Ecology Progress Series 694* 13-27.

Van der Hoop, J. M., M. J. Moore, S. G. Barco, T. V. Cole, P. Y. Daoust, A. G. Henry, D. F. McAlpine, W. A. McLellan, T. Wimmer, and A. R. Solow. (2013). Assessment of management to mitigate anthropogenic effects on large whales. *Conservation Biology: The Journal of the Society for Conservation Biology 27* (1): 121–133. DOI:10.1111/j.1523-1739.2012.01934

Van der Hoop, J. M., A. S. M. Vanderlaan, T. V. N. Cole, A. G. Henry, L. Hall, B. Mase-Guthrie, T. Wimmer, and M. J. Moore. (2015). Vessel strikes to large whales before and after the 2008 ship strike rule. *Conservation Letters 8* (1): 24–32. DOI:10.1111/conl.12105

van Hooidonk, R. J., D. P. Manzello, J. Moye, M. E. Brandt, J. C. Hendee, C. McCoy, and C. Manfrino. (2012). Coral bleaching at Little Cayman, Cayman Islands 2009. *Estuarine, Coastal and Shelf Science 106* 1–5. DOI:10.1016/j.ecss.2012.04.021

Van Houtan, K. S., D. L. Francke, S. Alessi, T. T. Jones, S. L. Martin, L. Kurpita, C. S. King, and R. W. Baird. (2016). The developmental biogeography of hawksbill sea turtles in the North Pacific. *Ecology and Evolution 6* (8): 2378–2389. DOI:10.1002/ece3.2034

van Neer, A., L. F. Jensen, and U. Siebert. (2015). Grey seal (*Halichoerus grypus*) predation on harbour seals (*Phoca vitulina*) on the island of Helgoland, Germany. *Journal of Sea Research 97* (2015): 1–4.

Van Parijs, S. M. (2015). Letter of introduction to the Biologically Important Areas Issue. *Aquatic Mammals 41* (1): 1–128. DOI:10.1578/AM.41.1.2015.1

van Weelden, C., J. R. Towers, and T. Bosker. (2021). Impacts of climate change on cetacean distribution, habitat and migration. *Climate Change Ecology 1* 100009. DOI:https://doi.org/10.1016/j.ecochg.2021.100009

Vandenbosch, R. (2000). Effects of ENSO and PDO events on seabird populations as revealed by Christmas bird count data. *Waterbirds 23* (3): 416–422.

Vanderlaan, M. S. A. and T. C. Taggart. (2007). Vessel collisions with whales: The probability of lethal injury based on vessel speed. *Marine Mammal Science 23* (1): 144–156. DOI:10.1111/j.1748-7692.2006.00098

Verkaik, K., J. Hamel, and A. Mercier. (2016). Carry-over effects of ocean acidification in a cold-water lecithotrophic holothuroid. *Marine Ecology Progress Series 557* 189–206. DOI:doi: 10.3354/meps11868

Virginia Department of Wildlife Resources. (2023, April 18). *Northern Long-eared Bat (Myotis septentrionalis)*. Retrieved May 30, 2024, from https://dof.virginia.gov/wp-content/uploads/VA-DWR-Northern-Long-Eared-Bats-Virginia_2023-04-18.pdf.

Virginia Department of Wildlife Resources. (2024). *Virginia Sea Turtle Nests and Crawls 1970-2021*.

Visser, I. N. (1999). A summary of interactions between orca (*Orcinus orca*) and other cetaceans in New Zealand waters. *New Zealand Natural Sciences 24* 101–112.

Vliet, K. A. (2001). Courtship behaviour of American alligators, *Alligator mississippiensis*. In G. C. Grigg, F. Seebacher, & C. E. Franklin (Eds.), *Crocodilian Biology and Evolution* (pp. 383–408). Baulkham Hills, Australia: Surrey Beatty & Sons.

Vollmer, N. L., E. Ashe, R. L. Brownell Jr., F. Cipriano, J. G. Mead, R. R. Reeves, M. S. Soldevilla, and R. Williams. (2019). Taxonomic revision of the dolphin genus *Lagenorhynchus*. *Marine Mammal Science 35* (3): 957–1057. DOI:https://doi.org/10.1111/mms.12573

Votier, S. C., K. Archibald, G. Morgan, and L. Morgan. (2011). The use of plastic debris as nesting material by a colonial seabird and associated entanglement mortality. *Marine Pollution Bulletin 62* (1): 168–172. DOI:https://doi.org/10.1016/j.marpolbul.2010.11.009

Wade, P. R., J. M. Ver Hoef, and D. P. DeMaster. (2009). Mammal-eating killer whales and their prey: Trend data for pinnipeds and sea otters in the North Pacific Ocean do not support the sequential megafaunal collapse hypothesis. *Marine Mammal Science 25* (3): 737–747. DOI:10.1111/j.1748-7692.2009.00282

Wakeford, A. (2001). *State of Florida Conservation Plan for Gulf Sturgeon (Acipenser oxyrinchus desotoi)* (Florida Marine Research Institute Technical Report). St. Petersburg, FL: Florida Marine Research Institute.

Waldman, J. R. and I. Wirgin. (1998). Status and Restoration Options for Atlantic Sturgeon in North America. *Conservation Biology 12* (3): 631–638.

Wallace, B. P., A. D. DiMatteo, B. J. Hurley, E. M. Finkbeiner, A. B. Bolten, M. Y. Chaloupka, B. J. Hutchinson, F. A. Abreu-Grobois, D. Amorocho, K. A. Bjorndal, J. Bourjea, B. W. Bowen, R. B. Duenas, P. Casale, B. C. Choudhury, A. S. Costa, P. H. Dutton, A. Fallabrino, A. Girard, M. Girondot, M. H. Godfrey, M. Hamann, M. Lopez-Mendilaharsu, M. A. Marchovaldi, J. A. Mortimer, J. A. Musick, R. Nel, N. J. Pitcher, J. A. Seminoff, S. Troeng, B. Witherington, and R. B. Mast. (2010a). Regional Management Units for Marine Turtles: A Novel Framework for Prioritizing Conservation and Research across Multiple Scales. *PLoS ONE 5* (12): e15465. DOI:10.1371/journal.pone.--15465

Wallace, B. P., R. L. Lewison, S. L. McDonald, R. K. McDonald, C. Y. Kot, S. Kelez, R. K. Bjorkland, E. M. Finkbeiner, S. Helmbrecht, and L. B. Crowder. (2010b). Global patterns of marine turtle bycatch. *Conservation Letters 3* (3): 131–142. DOI:10.1111/j.1755-236x.2010.00105

Wallace, B. P., B. A. Stacy, E. Cuevas, C. Holyoake, P. H. Lara, A. C. J. Marcondes, J. D. Miller, H. Nijkamp, N. J. Pilcher, I. Robinson, N. Rutherford, and G. Shigenaka. (2020). Oil spills and sea turtles: documented effects and considerations for response and assessment efforts. *Endangered Species Research 41* 17–37. DOI:10.3354/esr01009

Wallace, B. P., M. Zolkewitz, and M. C. James. (2015). Fine-scale foraging ecology of leatherback turtles. *Frontiers in Ecology and Evolution 3* 15.

Wallace, C. (1999). *Staghorn corals of the world: A revision of the coral genus Acropora (Scleractinia; Astrocoeniina; Acroporidae) worldwide, with emphasis on morphology, phylogeny and biogeography*. Collingsworth, Australia: Commonwealth Scientific and Industrial Research Organisation.

Walters, T. M., F. J. Mazzotti, and H. C. Fitz. (2016). Habitat selection by the invasive species Burmese python in southern Florida. *Journal of Herpetology 50* (1): 50–56.

Wang, D., T. C. Gouhier, B. A. Menge, and A. R. Ganguly. (2015). Intensification and spatial homogenization of coastal upwelling under climate change. *Nature 518* 390–407.

Wang, M., C. Hu, B. B. Barnes, G. Mitchum, B. Lapointe, and J. P. Montoya. (2019). The great Atlantic *Sargassum* belt. *Science 365* (6448): 83–87. DOI:10.1126/science.aaw7912

Wang, Y., Y. Pan, S. Parsons, M. Walker, and S. Zhang. (2007). Bats respond to polarity of a magnetic field. *Proceedings of the Royal Society of London B: Biological Sciences 274* (1627): 2901–2905. DOI:10.1098/rspb.2007.0904

Ward-Geiger, L., A. Knowlton, A. Amos, T. Pitchford, B. Mase-Guthrie, and B. Zoodsma. (2011). Recent sightings of the north Atlantic right whale in the Gulf of Mexico. *Gulf of Mexico Science 29* (1): 74–78.

Ward-Paige, C. A., D. M. Keith, B. Worm, and H. K. Lotze. (2012). Recovery potential and conservation options for elasmobranchs. *Journal of Fish Biology 80* (5): 1844–1869. DOI:10.1111/j.1095-8649.2012.03246

Ward, J. R. and K. D. Lafferty. (2004). The elusive baseline of marine disease: Are diseases in ocean ecosystems increasing? *PLoS Biology 2* (4): e120.

Wareham, V. E. and E. N. Edinger. (2007). Distribution of deep-sea corals in the Newfoundland and Labrador region, Northwest Atlantic Ocean. *Bulletin of Marine Science 89* 289–313.

Waring, G. T., C. P. Fairfield, C. M. Ruhsam, and M. Sano. (1992). *Cetaceans associated with Gulf Stream features off the northeastern United States*. Copenhagen, Denmark: International Council for Exploration of the Sea.

Waring, G. T., C. P. Fairfield, C. M. Ruhsam, and M. Sano. (1993). Sperm whales associated with Gulf Stream features off the northeastern U.S.A. shelf. *Fisheries Oceanography 2* 101–105.

Waring, G. T., T. Hamazaki, D. Sheehan, G. Wood, and S. Baker. (2001). Characterization of beaked whale (Ziphiidae) and sperm whale (*Physeter macrocephalus*) summer habitat in shelf-edge and deeper waters off the northeast U.S. *Marine Mammal Science 17* (4): 703–717.

Waring, G. T., E. Josephson, C. P. Fairfield-Walsh, and K. Maze-Foley. (2007). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2007* (NOAA Technical Memorandum NMFS-NE-205). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Waring, G. T., E. Josephson, K. Maze-Foley, and P. E. Rosel. (2010). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2010* (NOAA Technical Memorandum NMFS-NE-219). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Waring, G. T., E. Josephson, K. Maze-Foley, and P. E. Rosel. (2012). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2011* (NOAA Technical Memorandum NMFS-NE-221). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Waring, G. T., E. Josephson, K. Maze-Foley, and P. E. Rosel. (2013). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2012* (NOAA Technical Memorandum NMFS-NE-219). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Waring, G. T., E. Josephson, K. Maze-Foley, and P. E. Rosel. (2014). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2013* (NOAA Technical Memorandum NMFS-NE-228). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Waring, G. T., E. Josephson, K. Maze-Foley, P. E. Rosel, B. Byrd, T. V. N. Cole, L. Engleby, L. P. Garrison, J. Hatch, A. Henry, S. C. Horstman, J. Litz, M. C. Lyssikatos, K. D. Mullin, C. Orphanides, R. M. Pace, D. L. Palka, M. Soldevilla, and F. W. Wenzel. (2016). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2015* (NOAA Technical Memorandum NMFS-NE-238). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Waring, G. T., K. Maze-Foley, and P. E. Rosel, (Eds.). (2015). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2014* (NOAA Technical Memorandum NMFS-NE-231). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Waring, G. T., L. Nottestad, E. Olsen, H. Skov, and G. Vikingsson. (2008). Distribution and density estimates of cetaceans along the mid-Atlantic Ridge during summer 2004. *Journal of Cetacean Research and Management 10* (2): 137–146.

Waring, G. T., R. M. Pace, J. M. Quintal, C. P. Fairfield, and K. Maze-Foley. (2004). *U.S. Atlantic and Gulf of Mexico Marine Mammal Stock Assessments–2003* (NOAA Technical Memorandum NMFS-NE-182). Woods Hole, MA: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northeast Fisheries Science Center.

Warren, J. D. (2009). *Fine-scale Survey of Right and Humpback Whale Prey Abundance and Distribution* (Marine Mammals & Biological Oceanography Annual Reports: FY09). Washington, DC: Office of Naval Research.

Watanabe, Y., Q. Wei, H. Du, L. Li, and N. Miyazaki. (2012). Swimming behavior of Chinese sturgeon in natural habitat as compared to that in a deep reservoir: Preliminary evidence for anthropogenic impacts. *Environmental Biology of Fishes Online* 1–8. DOI:10.1007/s10641-012-0019-0

Watanabe, Y., Q. Wei, D. Yang, X. Chen, H. Du, J. Yang, K. Sato, Y. Naito, and N. Miyazaki. (2008). Swimming behavior in relation to buoyancy in an open swimbladder fish, the Chinese sturgeon. *Journal of Zoology 275* (4): 381-390. DOI:https://doi.org/10.1111/j.1469-7998.2008.00451.x

Waterhouse, L., S. A. Heppell, C. V. Pattengill-Semmens, C. McCoy, P. Bush, B. C. Johnson, and B. X. Semmens. (2020). Recovery of critically endangered Nassau grouper (*Epinephelus striatus*) in the Cayman Islands following targeted conservation actions. *Proceedings of the National Academy of Sciences 117* (3): 1587–1595. DOI:doi:10.1073/pnas.1917132117

Watkins, W. A., K. E. Moore, and P. Tyack. (1985). Sperm whale acoustic behavior in the southeast Caribbean. *Cetology 49* 1–15.

Watson, T. K., K. Hoomanawanui, R. Thurman, B. Thao, and K. Boyne. (2017). *Na Ikena I Kai (Seaward Viewsheds): Inventory of Terrestrial Properties for Assessment of Marine Viewsheds on the Main Eight Hawaiian Islands*. Camarillo, CA: US Department of the Interior, Bureau of Ocean Energy Management, Pacific OCS Region.

Watts, B. D., G. D. Therres, and M. A. Byrd. (2007). Status, distribution, and the future of bald eagles in the Chesapeake Bay area. *Waterbirds 30* 25–38. DOI:10.1675/1524-4695(2007)030[0025:sdatfo]2.0.co;2

Watts, P. and D. E. Gaskin. (1985). Habitat index analysis of the harbor porpoise (*Phocoena phocoena*) in the southern coastal Bay of Fundy, Canada. *Journal of Mammalogy 66* (4): 733–744.

Waugh, S. M., D. P. Filippi, D. S. Kirby, E. Abraham, and N. Walker. (2012). Ecological Risk Assessment for seabird interactions in Western and Central Pacific longline fisheries. *Marine Policy 36* (4): 933–946. DOI:10.1016/j.marpol.2011.11.005

Waycott, M., C. M. Duarte, T. J. B. Carruthers, R. J. Orth, W. C. Dennison, S. Olyarnik, A. Calladine, J. W. Fourqurean, K. L. Heck, Jr., A. R. Hughes, G. A. Kendrick, W. J. Kenworthy, F. T. Short, and S. L. Williams. (2009). Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the National Academy of Sciences 106* (30): 12377–12381.

Weaver, D. C., D. F. Naar, and B. T. Donahue. (2006). Deepwater reef fishes and multibeam bathymetry of the Tortugas South Ecological Reserve, Florida Keys National Marine Sanctuary, Florida *Emerging Technologies for Reef Fisheries Research and Management* (pp. 48-68). Seattle, WA: U.S. Department of Commerce.

Weaver, J. P., D. Rodriguez, M. Venegas-Anaya, J. R. Cedeño-Vázquez, M. R. J. Forstner, and L. D. Densmore. (2008). Genetic characterization of captive Cuban crocodiles (*Crocodylus rhombifer*) and evidence of hybridization with the American crocodile (*Crocodylus acutus*). *Journal of Experimental Zoology Part A: Ecological Genetics and Physiology 309* (10): 649–660.

Weber, M. (1995). *Kemp's Ridley Sea Turtle, Lepidochelys kempii* (National Marine Fisheries Service and U.S. Fish and Wildlife Service Status Reviews of Sea Turtles Listed under the Endangered Species Act of 1973). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Wei, C.-L., G. T. Rowe, E. Escobar-Briones, A. Boetius, T. Soltwedel, M. J. Caley, Y. Soliman, F. Huettmann, F. Qu, Z. Yu, C. R. Pitcher, R. L. Haedrich, M. K. Wicksten, M. A. Rex, J. G. Baguley, J. Sharma, R. Danovaro, I. R. MacDonald, C. C. Nunnally, J. W. Deming, P. Montagna, M. Lévesque, J. M. Weslawski, M. Wlodarska-Kowalczuk, B. S. Ingole, B. J. Bett, D. S. M. Billett, A. Yool, B. A. Bluhm, K. Iken, and B. E. Narayanaswamy. (2011). Global patterns and predictions of seafloor biomass using random forests. *PLoS ONE 5* (12): e15323. DOI:10.1371/journal.pone.0015323

Weimerskirch, H. (2004). Diseases threaten Southern Ocean albatrosses. *Polar Biology 27* 374–379. DOI:10.1007/s00300-004-0600-x

Weinrich, M., M. Martin, R. Griffiths, J. Bove, and M. Schilling. (1997). A shift in distribution of humpback whales, *Megaptera novaeangliae*, in response to prey in the southern Gulf of Maine. *Fishery Bulletin 95* (4): 826–836.

Weishampel, J. F., D. A. Bagley, and L. M. Ehrhart. (2006). Intra-annual loggerhead and green turtle spatial nesting patterns. *Southeastern Naturalist 5* (3): 453–462.

Weller, D. W. (2009). Predation on marine mammals. In W. F. Perrin, B. Wursig, & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 923–931). Cambridge, MA: Academic Press.

Weller, D. W., B. Würsig, H. Whitehead, J. C. Norris, S. K. Lynn, R. W. Davis, N. Clauss, and P. Brown. (1996). Observations of an interaction between sperm whales and short-finned pilot whales in the Gulf of Mexico. *Marine Mammal Science 12* (4): 588–593.

Wells, R. J. D., T. C. TinHan, M. A. Dance, J. M. Drymon, B. Falterman, M. J. Ajemian, G. W. Stunz, J. A. Mohan, E. R. Hoffmayer, W. B. Driggers, and J. A. McKinney. (2018). Movement, behavior, and habitat use of a marine apex predator, the scalloped hammerhead. *Frontiers in Marine Science 5*. DOI:10.3389/fmars.2018.00321

Wells, R. S., G. A. Early, J. G. Gannon, R. G. Lingenfelser, and P. Sweeney. (2008). *Tagging and tracking of rough-toothed dolphins (Steno bredanensis) from the March 2005 mass stranding in the Florida Keys* (NOAA Technical Memorandum NMFS-SEFSC-574). Key Biscayne, FL: National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Southeast Fisheries Science Center.

Wells, R. S., C. A. Manire, L. Byrd, D. R. Smith, J. G. Gannon, D. Fauqiuer, and K. D. Mullin. (2009). Movements and dive patterns of a rehabilitated Risso's dolphin, *Grampus griseus*, in the Gulf of

Mexico and Atlantic Ocean. *Marine Mammal Science 25* (2): 420–429. DOI:10.1111/j.1748-7692.2008.00251

Wells, R. S., H. L. Rhinehart, P. Cunningham, J. Whaley, M. Baran, C. Koberna, and D. P. Costa. (1999). Long distance offshore movements of bottlenose dolphins. *Marine Mammal Science 15* (4): 1098–1114.

Wells, R. S. and M. D. Scott. (1999). Bottlenose dolphin, *Tursiops truncatus* (Montagu, 1821). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 6, pp. 137–182). San Diego, CA: Academic Press.

Wells, R. S. and M. D. Scott. (2009). Common bottlenose dolphin, *Tursiops truncatus*. In W. F. Perrin, W. B., & J. G. M. Thewissen (Eds.), *Encyclopedia of Marine Mammals* (2nd ed., pp. 249–255). Cambridge, MA: Academic Press.

Welsh, S. A., M. F. Mangold, J. E. Skjeveland, and A. J. Spells. (2002). Distribution and movement of shortnose sturgeon (*Acipenser brevirostrum*) in the Chesapeake Bay. *Estuaries 25* (1): 101–104.

Werth, A. J. (2006). Odontocete suction feeding: Experimental analysis of water flow and head shape. *Journal of Morphology 267* (12): 1415–1428. DOI:10.1002/jmor.10486

West, K. L., W. A. Walker, R. W. Baird, W. White, G. Levine, E. Brown, and D. Schofield. (2009). Diet of pygmy sperm whales (*Kogia breviceps*) in the Hawaiian Archipelago. *Marine Mammal Science 25* (4): 931–943. DOI:10.1111/j.1748-7692.2009.00295

Wever, E. G., P. N. Herman, J. A. Simmons, and D. R. Hertzler. (1969). Hearing in the blackfooted penguin (*Spheniscus demersus*), as represented by the cochlear potentials. *Proceedings of the National Academy of Sciences 63* 676–680.

Wheatley-Techmer, W. (2023, June 7). Crocodile Reproduction at NAS Key West. E-mail sent from Wendy Wheatley-Techmer to Christopher E. Petersen.

Wheatley, P. V., H. Peckham, S. D. Newsome, and P. L. Koch. (2012). Estimating marine resource use by the American crocodile, *Crocodylus acutus*, in southern Florida, U.S.A. *Marine Ecology Progress Series 447* 211–229. DOI:10.3354/meps09503

Wheeler, J., Y. Satgé, A. Brown, J. Goetz, B. Keitt, H. Nevins, and E. Rupp. (2021). *Black-capped Petrel (Pterodroma hasitata) Conservation Update and Action Plan*. Natick, MA: International Black-capped Petrel Conservation Group.

White, A. W. (2004). Seabirds in the Bahamian Archipelago and adjacent waters: Transient, wintering, and rare nesting species. *North American Birds 57* 436–451.

Whitehead, H. (1982). Populations of humpback whales in the northwest Atlantic. *Reports of the International Whaling Commission 32* 345–353.

Whitehead, H. (2002). Estimates of the current global population size and historical trajectory for sperm whales. *Marine Ecology Progress Series 242* 295–304.

Whitehead, H. (2003). *Sperm Whales Social Evolution in the Ocean*. Chicago, IL: University of Chicago Press.

Whitehead, H. (2013). Trends in cetacean abundance in the Gully submarine canyon, 1988–2011, highlight a 21% per year increase in Sowerby's beaked whales (*Mesoplodon bidens*). *Canadian Journal of Zoology 91* 141–148.

Whitehead, H. and L. Weilgart. (1991). Patterns of visually observable behaviour and vocalizations in groups of female sperm whales. *Behaviour 118* 276–296.

Whitman, A. A. and P. M. Payne. (1990). Age of harbour seals, *Phoca vitulina concolor*, wintering in southern New England. *Canadian Field-Naturalist 104* (4): 579–582.

Wilcox, C., M. Puckridge, Q. Schuyler, K. Townsend, and B. Hardesty. (2018). A quantitative analysis linking sea turtle mortality and plastic debris ingestion. *Scientific Reports 8* (12536): 1-11. DOI:10.1038/s41598-018-30038-z

WildEarth Guardians. (2009). *A Petition: Requesting the Secretary of Commerce Add the Largetooth Sawfish, Pristis perotteti, to the List of Threatened and Endangered Species Maintained under the Authority of the Endangered Species Act*. Denver, CO: WildEarth Guardians.

Wildermann, N., C. Sasso, C. Gredzens, and M. M. P. B. Fuentes. (2018). Assessing the effect of recreational scallop harvest on the distribution and behaviour of foraging marine turtles. *Fauna & Flora International 54* (3): 307–314. DOI:10.1017/S0030605318000182

Wildermann, N. E., C. R. Sasso, L. W. Stokes, D. Snodgrass, and M. M. P. B. Fuentes. (2019). Habitat use and behavior of multiple species of marine turtles at a foraging area in the northeastern Gulf of Mexico. *Frontiers in Marine Science 6* (155): 1–13. DOI:10.3389/fmars.2019.00155

Wilkin, S. M., T. K. Rowles, E. Stratton, N. Adimey, C. L. Field, S. Wissman, G. Shigenaka, E. Fougeres, B. Mase, Southeast Region Stranding Network, and M. H. Ziccardi. (2017). Marine mammal response operations during the *Deepwater Horizon* oil spill. *Endangered Species Research 33* 107–118. DOI:10.3354/ers00811

William, T. C. and J. M. Williams. (1970). Radio tracking of homing and feeding flights of a neotropical bat, *Phyllostomus hastatus*. *Animal Behaviour 18* 302–309.

Williams-Walls, N., J. O'Hara, R. M. Gallagher, D. F. Worth, B. D. Peery, and J. R. Wilcox. (1983). Spatial and temporal trends of sea turtle nesting on Hutchinson Island, Florida, 1971–1979. *Bulletin of Marine Science 33* (1): 55–66.

Williams, D. and M. Miller. (2011). Attributing mortality among drivers of population decline in *Acropora palmata* in the Florida Keys (U.S.A.). *Coral Reefs 31* (2): 369–382. DOI:10.1007/s00338-011-0847-y

Williams, D. E. and M. W. Miller. (2005). Coral disease outbreak: Pattern, prevalence and transmission in *Acropora cervicornis*. *Marine Ecology Progress Series 301* 119–128.

Williams, H. (2023, June 15). *Hundreds of Kemp's ridley nests, most endangered sea turtle in world, have been found in Texas this year*. Retrieved August 23, 2023, from https://www.kiiitv.com/article/life/animals/sea-turtle-nests-in-texas/503-b7a7ac72-f841-476b-afbe-defb88c70a80.

Williams, K. A., I. J. Stenhouse, E. E. Connelly, and S. M. Johnson. (2015). *Mid-Atlantic Wildlife Studies: Distribution and Abundance of Wildlife along the Eastern Seaboard 2012–2014* (Science Communications Series BRI 2015-19). Portland, ME: Biodiversity Research Institute.

Williams, T. C., J. M. Williams, and D. R. Griffin. (1966). The homing ability of the neotropical bat *Phyllostomus Hastatus*, with evidence for visual orientation. *Animal Behaviour 14* (4): 468–473.

Willmott, J. R., G. Forcey, M. Vukovich, S. McGovern, J. Clerc, and J. Carter. (2021). *Ecological Baseline Studies of the US Outer Continental Shelf. OCS Study BOEM 2021-079*. Washington, DC: U.S.

Department of the Interior, Bureau of Ocean Energy Management, Office of Renewable Energy Programs.

Wilson, S. C. (1978). *Social Organization and Behavior of Harbor Seals, Phoca vitulina concolor, in Maine*. Washington, DC: Smithsonian Institution Press.

Wimmer, T. and H. Whitehead. (2004). Movements and distribution of northern bottlenose whales, *Hyperoodon ampullatus*, on the Scotian Slope and in adjacent waters. *Canadian Journal of Zoology 82* (11): 1782–1794. DOI:10.1139/z04-168

Wingate, D. B. (1964). Discovery of breeding black-capped Petrels on hispaniola. *The Auk 81* 147–159.

Wippelhauser, G. S. and T. S. Squiers, Jr. (2015). Shortnose sturgeon and Atlantic sturgeon in the Kennebec River system, Maine: A 1977–2001 retrospective of abundance and important habitat. *Transactions of the American Fisheries Society 144* 591–601.

Wippelhauser, G. S., G. B. Zydlewski, M. Kieffer, J. Sulikowski, and M. T. Kinnison. (2015). Shortnose sturgeon in the Gulf of Maine: Use of spawning habitat in the Kennebec system and response to dam removal. *Transactions of the American Fisheries Society 144* (4): 742–752. DOI:10.1080/00028487.2015.1037931

Witherington, B. and S. Hirama. (2006). Sea turtles of the epi-pelagic *Sargassum* drift community. In M. Frick, A. Panagopoulou, A. F. Rees, & K. Williams (Eds.), *Book of Abstracts. Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation*. Athens, Greece: International Sea Turtle Society.

Witman, J. D. and R. W. Lamb. (2018). Persistent differences between coastal and offshore kelp forest communities in a warming Gulf of Maine. *PLoS ONE 13* (1): e0189388. DOI:10.1371/journal.pone.0189388

Witt, M. J., L. A. Hawkes, M. H. Godfrey, B. J. Godley, and A. C. Broderick. (2010). Predicting the impacts of climate change on a globally distributed species: The case of the loggerhead turtle. *The Journal of Experimental Biology 213* (6): 901–911. DOI:10.1242/jeb.038133

Witzell, W. N. (1983). *Synopsis of biological data on the hawksbill turtle, Eretmochelys imbricata (Linnaeus, 1766)*. Rome, Italy: United Nations Environment Programme, Food and Agriculture Organization of the United Nations.

Wood LaFond, S. A. (2009). *Dynamics of Recolonization: A study of the gray seal (Halichoerus grypus) in the Northeast U.S.* (Unpublished doctoral dissertation). University of Massachusetts, Boston, Boston, MA.

Wood, L. D., B. Brunnick, and S. L. Milton. (2017a). Home range and movement patterns of subadult hawksbill sea turtles in southeast Florida. *Journal of Herpetology 51* (1): 58–67.

Wood, L. D., S. L. Milton, and T. L. Maple. (2017b). Foraging Behavior of Wild Hawksbill Turtles (*Eretmochelys imbricata*) in Palm Beach County, Florida, USA. *Chelonian Conservation and Biology 16* (1): 70–75. DOI:10.2744/CCB-1242.1

Wooley, C. M. and E. J. Crateau. (1985). Movement, microhabitat, exploitation, and management of Gulf of Mexico sturgeon, Apalachicola River, Florida. *North American Journal of Fisheries Management 5* 590–605.

Work, T. M. and G. H. Balazs. (2013). Tumors in sea turtles: The insidious menace of fibropapillomatosis. *The Wildlife Professional Fall 2013* 44–47.

World Ocean Review. (2015). *Sustainable Use of Our Oceans - Making Ideas Work*. Hamburg, Germany: Maribus.

World Register of Introduced Marine Species. (2023). *Effects of fisheries on marine biodiversity*. Retrieved June 2, 2023, from https://www.coastalwiki.org/introduced/Effects_of_fisheries_on_marine_biodiversity.

World Register of Marine Species Editorial Board. (2024). *World Register of Marine Species - Home Page*. Retrieved July 31, 2024, from https://www.marinespecies.org/.

Worm, B., E. B. Barbier, N. Beaumont, J. E. Duffy, C. Folke, B. S. Halpern, J. B. C. Jackson, H. K. Lotze, F. Micheli, S. R. Palumbi, E. Sala, K. A. Selkoe, J. J. Stachowicz, and R. Watson. (2006). Impacts of Biodiversity Loss on Ocean Ecosystem Services. *Science 314* (5800): 787-790. DOI:doi:10.1126/science.1132294

Wright, A. J. and L. A. Kyhn. (2015). Practical management of cumulative anthropogenic impacts with working marine examples. *Conservation Biology 29* (2): 333–340. DOI:10.1111/cobi.12425

Würsig, B., T. A. Jefferson, and D. J. Schmidly. (2000). *The Marine Mammals of the Gulf of Mexico*. College Station, TX: Texas A&M University Press.

Wurster, C. F., Jr. and D. B. Wingate. (1968). DDT residues and declining reproduction in the Bermuda petrel. *Science 159* (3818): 979–981. DOI:10.1126/science.159.3818.979

Wurtsbaugh, W. A., H. W. Paerl, and W. K. Dodds. (2019). Nutrients, eutrophication and harmful algal blooms along the freshwater to marine continuum. *WIREs Water 6* (5): e1373. DOI:https://doi.org/10.1002/wat2.1373

Yates, D. (2015). *Bat Mist Net Surveys at Maine Naval Installations: Cutler, Great Pond and Redington*. Portland, ME: Biodiversity Research Institute.

Yochem, P. and S. Leatherwood. (1985). Blue whale, *Balaenoptera musculus* (Linnaeus, 1758). In S. H. Ridgway & R. Harrison (Eds.), *Handbook of Marine Mammals* (Vol. 3, pp. 193–240). San Diego, CA: Academic Press.

Yoshida, H., J. Compton, S. Punnett, T. Lovell, K. Draper, G. Franklin, N. Norris, P. Phillip, R. Wilkins, and H. Kato. (2010). Cetacean sightings in the eastern Caribbean and adjacent waters, spring 2004. *Aquatic Mammals 36* (2): 154–161. DOI:doi:10.1578/AM.36.2.2010.154

Young, C. N., J. Carlson, C. Hutt, D. Kobayashi, C. T. McCandless, and J. Wraith. (2016). *Status review report: Oceanic whitetip shark (Carcharhinius longimanus)* (Final Report to the National Marine Fisheries Service, Office of Protected Resources). Silver Spring, MD: National Oceanic and Atmospheric Administration, National Marine Fisheries Service.

Young, C. N. and J. K. Carlson. (2020). The biology and conservation status of the oceanic whitetip shark (*Carcharhinus longimanus*) and future directions for recovery. *Reviews in Fish Biology and Fisheries 30* (2): 293–312. DOI:10.1007/s11160-020-09601-3

Zimmerman, R. C. (2021). Scaling up: Predicting the impacts of climate change on seagrass ecosystems. *Estuaries and Coasts 44* (2): 558–576. DOI:10.1007/s12237-020-00837-7

Zoidis, A. M., K. S. Lomac-MacNair, D. S. Ireland, M. E. Rickard, K. A. McKown, and M. D. Schlesinger. (2021). Distribution and density of six large whale species in the New York Bight from monthly aerial surveys 2017 to 2020. *Continental Shelf Research 230* 104572. DOI:https://doi.org/10.1016/j.csr.2021.104572

Zollett, E. A. (2009). Bycatch of protected species and other species of concern in U.S. East Coast commercial fisheries. *Endangered Species Research 9* (1): 49–59. DOI:10.3354/esr00221

Zolman, E. S. (2002). Residence patterns of bottlenose dolphins (*Tursiops truncatus*) in the Stono River estuary, Charleston County, South Carolina, U.S.A. *Marine Mammal Science 18* 879–892.

Zoodsma, B. J., K. Howe, M. White, J. Jakush, C. George, T. Gowan, P. K. Hamilton, K. Jackson, T. Pitchford, C. Taylor, and L. Ward. (2016). *North Atlantic right whale calving area surveys: 2014/2015 results* (NOAA Technical Memorandum NMFS SER-4). St. Petersburg, FL: National Marine Fisheries Service Southeast Regional Office.

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