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This document transmits the U.S. Fish and Wildlife Service's (Service) biological opinion (BO) on the effects of Bureau of Ocean Energy Management's (BOEM) and Bureau of Safety and Environmental Enforcement's (BSEE) proposed oil and gas leasing, exploration, development, production, decommissioning, and all related activities in the Gulf of Mexico (GOM) Outer Continental Shelf (OCS) within existing leased areas and those areas proposed for future leasing in the Western Planning Area (WPA), the Central Planning Area (CPA), and the Eastern Planning Area (EPA). This BO will cover the following leases and activities for a ten-year period starting from the date this document is signed.

1. All oil and gas leases issued as a result of sales held during the ten-year period, including associated exploration, development, production, and decommissioning activities authorized by BOEM or BSEE under those leases;
2. Those associated exploration, development, production, and decommissioning activities authorized by BOEM or BSEE during this 10-year period under all other oil and gas leases, regardless of the date of issuance of the lease; and
3. Those geological and geophysical permits issued by BOEM during the ten-year period.

This biological opinion is submitted in accordance with section 7 of the Endangered Species Act

(Act) of 1973, as amended (16 U.S.C. 1531 et seq.).

The following table contains a State-by-State listing of threatened (T) and endangered (E) species and their critical habitat (CH) included in this opinion that may potentially be affected by the proposed actions.

State-by-State listing of threatened (T) and endangered (E) species and their critical habitat (CH) included in this opinion that may potentially be affected by the proposed actions.

COMMON NAME	SCIENTIFIC NAME	FEDERAL STATUS	CRITICAL HABITAT	OCCURRENCE WITHIN ACTION AREA	HABITAT TYPE
Reptiles					
Green sea turtle	*Chelonia mydas*	T	N	FL	Coastal beaches
Hawksbill sea turtle	*Eretmochelys imbricata*	E	Y	TX to FL	Coastal beaches
Kemp's ridley sea turtle	*Lepidochelys kempii*	E	N	TX to FL	Coastal beaches
Leatherback sea turtle	*Dermochelys coriacea*	E	Y	TX to FL	Coastal beaches
Loggerhead sea turtle	*Caretta caretta*	T	Y	TX to FL	Coastal beaches
Fish					
Atlantic (Gulf subspecies) sturgeon	*Acipenser oxyrinchus desotoi*	T	Y	LA to FL	GOM waters and adjacent freshwater streams and rivers
Birds					
Cape Sable seaside sparrow	*Ammodramus maritimus mirabilis*	E	Y	S. FL	Inland and coastal terrestrial: short hydroperiod marl prairie
Mississippi sandhill crane	*Grus canadensis pulla*	E	Y	MS	Inland and coastal freshwater: wet pine savanna, pine plantations, swamps and wetlands edged by pine forests
Piping plover	*Charadrius melodus*	T	Y	TX to FL	Coastal beaches
Roseate tern	*Sterna dougallii dougallii*	T	N	FL	Coastal beaches
Rufa red knot	*Calidris canutus rufa*	T	N	TX to FL	Coastal beaches

Whooping crane	*Grus americana*	E, NEP	Y	E: TX; NEP: LA, FL	Inland freshwater and coastal estuarine: salt flats, grasslands, wetlands, ponds, and bays
Wood stork	*Mycteria americana*	T	N	MS, AL, FL	Freshwater and estuarine wetlands with periods of flooding followed by dry periods
Mammals					
Alabama beach mouse	*Peromyscus polionotus ammobates*	E	Y	AL	Coastal terrestrial: dune systems
Choctawhatchee beach mouse	*Peromyscus polionotus allophrys*	E	Y	FL	Coastal terrestrial: dune systems
Perdido Key beach mouse	*Peromyscus polionotus trissyllepsis*	E	Y	AL, FL	Coastal terrestrial: dune systems
St. Andrew beach mouse	*Peromyscus polionotus peninsularis*	E	Y	FL	Coastal terrestrial: dune systems
West Indian manatee	*Trichechus manatus*	T	Y	TX to FL	Inland freshwater; coastal estuarine: tidal rivers and streams, swamps, springs, salt marshes, lagoons, canals

E = Endangered
T = Threatened
NEP = Nonessential Experimental Population

The National Marine Fisheries Service (NMFS) has jurisdiction for sea turtles in the marine environment. When sea turtles leave the marine environment and come onshore to nest, the Service is responsible for those species. The Kemp's ridley, loggerhead, leatherback, green and hawksbill sea turtle are found in GOM coastal waters.

The leatherback sea turtle regularly nests in the U.S. Caribbean in Puerto Rico and the U.S. Virgin Islands. Along the U.S. Atlantic coast, most nesting occurs in Florida (NMFS and Service 1992). Although uncommon, leatherback nesting has also been reported in Texas (Shaver 2008); on the northwest coast of Florida (LeBuff 1990, Florida Fish and Wildlife Conservation Commission [FWC] 2009a); and in southwest Florida a false crawl (nonnesting emergence) has been observed on Sanibel Island (LeBuff 1990).

Major green sea turtle nesting colonies in the Atlantic occur on Ascension Island, Aves Island, Costa Rica, and Surinam. Within the U.S., green sea turtles nest in small numbers in the U.S. Virgin Islands and Puerto Rico, and in larger numbers along the east coast of Florida, particularly in Brevard, Indian River, St. Lucie, Martin, Palm Beach, and Broward Counties (NMFS and Service 1991). Nests have been documented, in smaller numbers, north of these counties, from Volusia through Nassau Counties in Florida, as well as in Georgia, South Carolina, North Carolina, and as far north as Delaware in 2011. Nesting has also been documented in smaller numbers along the Gulf coast of Florida from Escambia County through Franklin County in northwest Florida and from Pinellas County through Monroe County in southwest Florida (FWC 2016).

Within the continental U.S., hawksbill sea turtle nesting is rare and is restricted to the southeastern coast of Florida (Volusia through Miami-Dade Counties) and the Florida Keys (Monroe County) (Meylan 1992, Meylan 1995, FWC/Florida Fish and Wildlife Research Institute [FWRI] 2010b). In the U.S. Caribbean, hawksbill nesting occurs on beaches throughout Puerto Rico and the U.S. Virgin Islands (NMFS and Service 1993).

The Service concurs that the proposed action is not likely to adversely affect nesting leatherback, green, and hawksbill sea turtles and their nests due to the low nesting numbers and the low probability of an oil spill occurring when those species and their nests could be present. Potential impacts to Kemp's ridley and loggerhead sea turtles are further discussed below.

Roseate terns are considered as two distinct population segments (DPSs); the Northeastern population which is listed as endangered, and the Caribbean population (including breeding birds in Florida, Puerto Rico, and the Virgin Islands) which is listed as threatened. Until the early 1970s, the Dry Tortugas were the primary roseate tern breeding area in Florida (Robertson 1978). Predators and nesting failure due to storm surges from tropical storms probably led to the gradual shifting of this colony to the Florida Keys, with much of the activity occurring on spoil or otherwise denuded islands in the Key West area (Robertson 1978). Potential impacts from the proposed project are not expected to extend into roseate tern habitat (i.e., less than 0.5 percent chance). In addition, there was no direct roseate tern mortality reported after the Deepwater Horizon (DWH) oil spill and response. Accordingly, the Service concurs that the proposed project is not likely to adversely affect this species.

The wood stork is primarily associated with freshwater and estuarine habitats that are used for nesting, roosting, and foraging. Wood storks typically nest colonially in medium to tall trees that occur in stands located either in swamps or on islands surrounded by relatively broad expanses of open water (Ogden 1991; Rodgers et al. 1996). Typical foraging sites include a mosaic of shallow water wetlands. Several factors affect the suitability of potential foraging habitat for wood storks. Foraging habitats must provide both a sufficient density and biomass of forage fish and other prey and have vegetation characteristics that allow storks to locate and capture prey. Calm water, about 2 to 16 inches in depth, and free of dense aquatic vegetation, is preferred. Potential impacts from the proposed project are not expected to extend into suitable wood stork habitats. Accordingly, the Service concurs that the proposed project is not likely to adversely affect this species.

NMFS and FWS share jurisdiction for the threatened Atlantic (Gulf subspecies) sturgeon *(Acipenser oxyrinchus desotoi*), which occurs in the Gulf of Mexico waters and migrates to adjacent freshwater (streams and rivers) to spawn. The March 19, 2003, final rule designating Atlantic (Gulf subspecies) sturgeon critical habitat, indicates that NMFS is responsible for all consultations regarding Atlantic (Gulf subspecies) sturgeon and critical habitat in marine units and in estuarine units with BOEM/BSEE acting as the lead federal agency. The two agencies have, therefore, agreed that NMFS will conduct endangered species consultation with BOEM regarding the effects of the proposed action on the Atlantic (Gulf subspecies) sturgeon and this species will not be considered further in this BO.

To date, four nonessential experimental populations (NEPs) of whooping cranes have been established in North America: the non-migratory Florida population (58 FR 5647-5658, January 22, 1993); the migratory Rocky Mountain population, with a range covering 5 western states (CO, ID, NM, UT, WY, 62 FR 38932-38939, July 1997); the migratory Eastern population, breeding in Wisconsin and wintering in the southeast, with a recognized range in 20 states (AL, AR, FL, GA, IA, IL, IN, KY, LA, MI, MN, MO, MS, NC, OH, SC, TN, VA, WI, WV, 66 FR 33903-33917, June 26, 2001); and the non-migratory Louisiana population (76 FR 6066-6082). These NEPs have some overlap in their range, but have a strong homing tendency towards establishing their nesting territory near the natal area. Whooping cranes were reintroduced into the Rocky Mountains (1975-1989), Florida (1993-2005), the Eastern U.S. (2001-2010), and Louisiana (2011-present). These reintroduced populations were designated as NEPs under section 10(j) of the Act, as amended. A NEP population is a reintroduced population believed not to be essential for the survival of the species, but important for its full recovery and eventual removal from the endangered and threatened list. These populations are treated as "threatened" species except that the Act's section 7 consultation regulations (requiring consultation with the Service to reduce adverse impacts from federal actions) do not apply (except where the species occurs within National Parks or National Wildlife Refuges) and critical habitat cannot be designated. Because no NEPs of whooping cranes within National Parks or National Wildlife Refuges would be impacted by the proposed action, only the endangered wintering population in Texas will be considered further in this BO.

For law enforcement purposes, the American alligator (*Alligator mississippiensis*) is classified as "threatened due to similarity of appearance". They are biologically neither endangered nor threatened.

This BO is based on information provided in the February 2014 Biological Assessment (BA) and on meetings, telephone conversations and email correspondence with BOEM/BSEE's and the Service's Office of the Solicitor; the Service's Southeast Regional Office; field offices in Panama City and Vero Beach, Florida, Jackson, Mississippi, Daphne, Alabama, Clear Lake and Corpus Christi, Texas; and the Gulf of Mexico Regional Office of BOEM/BSEE. BOEM/BSEE provided the Service with an Oil Spill Risk Analyses (OSRA), a BA and other documents which were used in preparing this BO. A complete administrative record of this consultation is on file at this office.

Consultation History

In the CPA, the original BO for OCS oil and gas activities in the GOM was submitted by the Service on April 10, 1979. Consultation was reinitiated in November 1981; that consultation resulted in a June 30, 1982, BO, which was amended on October 25, 1982. Subsequent Service reviews of that BO and amendment provided on April 9, 1984; January 15, 1985; January 29, 1986; and June 22, 1987, concluded that formal consultation need not be reinitiated because the October 25, 1982, BO was still up-to-date and valid. The above-referenced BOs covered only oil and gas leasing and exploration. Consultation was reinitiated in January 1989, to include all phases of OCS oil and gas activities (i.e., leasing, exploration, development, production, and abandonment). That consultation resulted in a BO dated July 18, 1989 that was revised on August 25, 1989. On July 26, 1990, the Service concluded that formal consultation need not be reinitiated. Consultation was reinitiated in 1991, and a revised BO was completed on March 15, 1991. On June 8, 1992; May 7, 1993; and June 24, 1994, the Service concluded that formal consultation need not be reinitiated and that the revised BO dated March 15, 1991, was valid. Consultation was reinitiated in 1995, and the Service submitted a revised BO dated July 11, 1995. On October 4, 1996, the FWS concluded that formal consultation need not be reinitiated and that the existing BO dated July 11, 1995, was still up-to-date and valid. Formal consultation was re-initiated on April 16, 1997, for CPA Lease Sales 169, 172, 175, 178, and 182. That consultation included all aspects of oil and gas exploration, development, production and abandonment activities. That formal consultation resulted in the December 24, 1997 BO. Incidental take of federally listed species was not anticipated for the above-mentioned actions.

In the WPA, the original BO for OCS oil and gas activities in the Gulf of Mexico was submitted by the FWS on April 10, 1979. Consultation was reinitiated in November 1981; that consultation resulted in a June 30, 1982, BO, which was amended on October 25, 1982. Subsequent FWS reviews of that BO and amendments provided on April 9, 1984; January 15, 1985; January 29, 1986; and June 22, 1987, concluded that formal consultation need not be reinitiated because the October 25, 1982, BO was still up-to-date and valid. The above-referenced BOs covered only oil and gas leasing and exploration. Consultation was reinitiated in January 1989, to include all phases of OCS oil and gas activities (i.e., leasing, exploration, development, production, and abandonment). That consultation resulted in a BO dated June 29, 1989. On June 23, 1990, January 4, 1991, and April 1, 1993, the Service concluded that formal consultation need not be reinitiated and that the reviewed BO dated June 29, 1989, was valid. Formal consultation was again initiated on March 23, 1994, which resulted in a June 16, 1994, BO. On March 29, 1995, and April 5, 1996, a letter was sent that upheld the June 16, 1994, BO as valid and up-to-date. Consultation was reinitiated on September 9, 1997, which resulted in a March 9, 1998, BO. Again consultation was reinitiated on March 11, 2002, which resulted in the June 25, 2002, BO. Incidental take of federally listed species was not anticipated for the above-mentioned actions.

In the EPA, the Service issued a June 8, 2001, BO on Lease Sale 181. Within that BO, the Service concurred that implementation of the proposed lease sale was not likely to jeopardize the continued existence of any federally listed species or adversely modify any designated critical habitat.

Following the most recent BOs to date, additional information became available that enabled a more thorough evaluation of the proposed action and the potential for impacts to listed species. In addition, the OSRA had changed substantially from previous lease sale analyses; therefore, on April 15, 2002, formal consultation was reinitiated for CPA Lease Sales 185, 190, 194, 198, and 201, and WPA Lease Sales 187, 192, 196, and 200. A final BO was issued on January 13, 2003.

An August 6, 2007 letter and BA was provided to the Service regarding the effects of the proposed GOM OCS Oil and Gas Lease Sales: 2007-2012; additional information was provided in an August 17, 2007, letter. The proposed action included 11 oil and gas lease sales in the WPA and CPA. By memorandum dated September 14, 2007, the Service concurred that the subject action was not likely to adversely affect federally listed threatened or endangered species.

On April 20, 2010, the DWH mobile offshore drilling unit exploded and began to burn uncontrollably. The explosion occurred on a lease in the Mississippi Canyon Block 252 located approximately 53 miles southeast of the nearest land at the end of the Mississippi River's birds foot delta. According to a review of flow rate estimates of the DWH oil spill (McNutt et al. 2011), a total of 5 million barrels of oil were released (before accounting for containment). Following that explosion and spill, BOEM/BSEE requested reinitiation of the existing consultation for OCS oil and gas activities, in a letter dated July 30, 2010. In a September 27, 2010, letter, the Service concurred that the DWH spill necessitated reconsideration of the existing consultation for the 2007-2012 Five Year Program GOM lease sales. Per BOEM/BSEE's electronic mail request on March 8, 2012, for a revised/updated list of federally listed species that may be affected by the activities associated with the 2007-2012 lease sales, the Service responded with a letter dated April 6, 2012.

Since the initial request for reinitiation in 2010, this consultation has expanded in scope to include a programmatic approach for oil and gas activities in the GOM. As discussed above, this BO will cover the following leases and activities for a ten-year period starting from the date this document is signed:

1. All oil and gas leases issued as a result of sales held during the ten-year period, including associated exploration, development, production, and decommissioning activities authorized by BOEM or BSEE under those leases;
2. Those associated exploration, development, production, and decommissioning activities authorized by BOEM or BSEE during this 10-year period under all other oil and gas leases, regardless of the date of issuance of the lease; and
3. Those geological and geophysical permits issued by BOEM during the ten-year period.

Because of the need for reinitiation of consultation due to DWH and because of the change in scope of the proposed activities, BOEM/BSEE submitted a BA to the Service, dated February 28, 2014. That BA included an analysis of potential impacts to federally listed species associated with oil and gas lease sales in the GOM, as well as all OCS oil and gas activities in the GOM including exploration, development, and production related to prior and future lease sales. An updated BA, dated August 2015, was provided to the Service in response to our November 7, 2014, request for additional information. Following review of the August 2015 BA, the Service again requested additional information via a memorandum dated October 22, 2015. On March 8, 2016, the Service received all necessary information from BOEM/BSEE to initiate formal consultation. Under mutual agreement, agency coordination has continued since that time.

BIOLOGICAL OPINION

DESCRIPTION OF THE PROPOSED ACTION

As mentioned above, this BO is an expanded programmatic consultation, which includes lease sales in the GOM proposed and expected for a ten-year period starting from the date this document is signed. Because a lease life is typically up to 40 years, this consultation may include lease activities for a 50 year time span (i.e., 40 years out from leases issued at year 10).

Under the Act, the "action area" of the project includes all areas that are directly or indirectly affected by the action, and not merely the immediate area involved in the action. When including indirect effects, the action area includes federal OCS waters as well as the coastal areas, ports, airspace and waterways used by related transport vessels, costal infrastructure, fabrication sites, pipelines connecting to the offshore pipeline system, transportation, and other estuarine and marine areas affected by OCS oil and gas activities.

For purposes of this consultation, "offshore" refers to the OCS portion of the GOM, beginning 10 miles offshore Florida; 3.5 miles offshore Louisiana, Mississippi, and Alabama; and 10.4

miles offshore Texas and extending seaward to the limits of the U.S. jurisdiction over the continental shelf (often referred to as the Exclusive Economic Zone [EEZ]), in water depths up to approximately 10,978 feet.

For oil and gas leasing purposes, the GOM is divided into three geographic areas: (1) the WPA, (2) the CPA, and (3) the EPA (Figure 1). The 2017-2022 leasing schedule (https://www.boem.gov/five-year-program-2017-2022/) for the GOM includes 10 total sales; one sale in 2017, two sales each year in 2018, 2019, 2020, and 2021, and one sale in 2022.

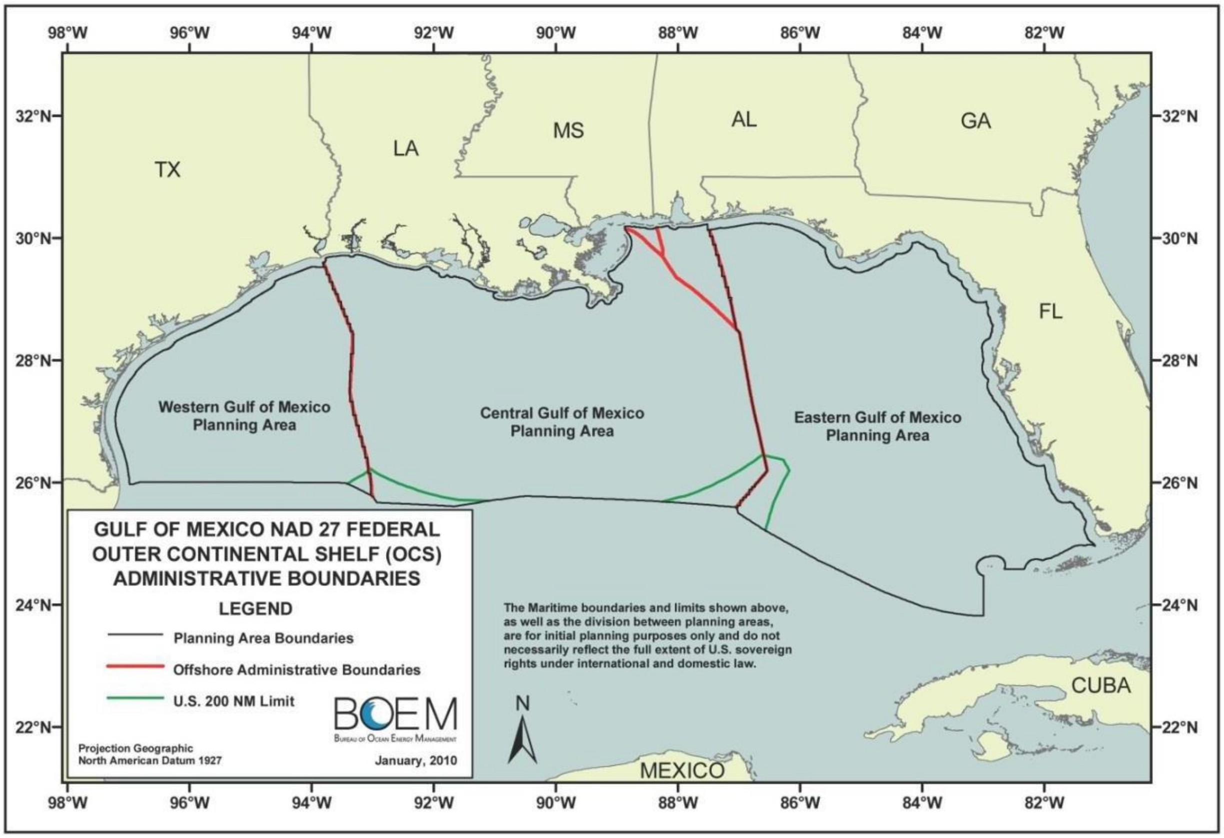

Figure 1. Federal leasing boundaries in the Gulf of Mexico. (https://www.boem.gov/uploadedFiles/BOEM/Oil_and_Gas_Energy_Program/Mapping_and_Data/Administrative_Boundaries/Gulf_Plan.pdf).

The CPA covers approximately 66.45 million acres and as of March 2016, approximately 48.3 million acres are currently unleased. The WPA covers approximately 28.58 million acres and as of March 2016, approximately 23.6 million acres are currently unleased. The EPA covers approximately 657,905 acres (nearest point of land is 125 miles northwest in Louisiana) and as of March 2016, approximately 595,475 acres are currently unleased.

The majority of the EPA is unavailable for leasing consideration through June 30, 2022, under the GOM Energy Security Act of 2006. Leasing information related to all three planning areas is updated monthly and can be found on BOEM's website at https://www.boem.gov/Gulf-of-Mexico-Region-Lease-Map/.

In addition to leasing, this BO covers all phases of OCS oil and gas activity including: (1) exploration and delineation plans and drilling, (2) development and production drilling, (3) infrastructure placement/structure installation and decommissioning activities, (4) associated vessel and helicopter traffic, (5) operational waste discharges, and (6) post-authorization compliance (inspection) activities. The following is a brief description/overview of the proposed activities.

Leasing

BOEM's Five Year Program consists of a schedule of oil and gas lease sales indicating the size, timing, and location of proposed leasing activity determined to best meet national energy needs for the five-year period following its approval. An area must be included in an approved Five Year Program in order to be offered for leasing. After developing the Five Year Program, BOEM conducts further tiered National Environmental Policy Act (NEPA) analysis at the lease sale, exploration plan (i.e., a specific project), and development plan (i.e., specific platform) stages. Leasing in and of itself is a purely administrative action and does not authorize subsequent oil and gas activities. BOEM/BSEE must review those activities before they occur (e.g., review and approval of exploration, development, and production plans; permit approvals for pipeline laying, drilling, and structure removal, etc.).

Exploration and Delineation Plans and Drilling

An Exploration Plan (EP) must be submitted to BOEM for review and approval before any exploration activities can begin on a lease. The EP describes exploration activities, proposed drilling and well-testing operations, environmental monitoring plans, oil-spill response plans, a proposed schedule of the exploration activities, and other relevant information. After an EP is approved by BOEM and before drilling operations begin, the operator is required to submit and obtain approval from BSEE for an Application to Drill (APD). An environmental review would be completed at this plan stage prior to approval/permit issuance for drilling.

Exploration well generally refers to the first well drilled on a prospective geologic structure to confirm that a resource exists and to validate how much of the resource can be expected. If the quantities of the discovered resource appear to be economically viable, one or more follow-up delineation wells help define the amount of the resource or the extent of the reservoir. For all leases issued under the proposed action, BOEM/BSEE estimate the following annual activity levels for exploration and delineation wells:

- WPA: 30-43 exploration and delineation wells annually
- CPA: 143-203 exploration and delineation wells annually
- EPA: 0-1 exploration and delineation wells annually

Development and Production Drilling

A Development and Production Plan (DPP) must be submitted for approval to BOEM before an operator may begin development or production activities in the EPA. Likewise, a DPP or Development Operations and Coordination Document must be submitted for approval in the CPA and WPA. These plans describe the proposed production operations, drilling activities, platforms or other facilities, environmental monitoring plans, a proposed schedule of development and production activities, and other relevant information. A NEPA Review (typically an Environmental Assessment) is prepared at this stage, prior to approval/permit issuance for development drilling.

Development wells are designed to extract resources from a known hydrocarbon reservoir. For all leases issued under the proposed action, BOEM/BSEE estimate the following activity levels for development wells:

- WPA: 37-53 development wells annually
- CPA: 177-251 development wells annually
- EPA: 0-1 development wells annually

Infrastructure Placement/Structure Installation and Decommissioning Activities

OCS exploration, development, and production require certain onshore support facilities including office space, helicopter and fixed-wing aircraft facilities, navigation channels and docks for boating activities, platform and drilling rig construction yards, pipelines, oil and gas processing and separating facilities, refineries, and supply bases. Oil and gas activities in the GOM began more than 45 years ago; therefore, necessary onshore facilities to support those activities are already in place and no major new facilities are anticipated as a result of the proposed lease sales. Due to the uncertain nature of oil and gas supply and demand, companies often prefer to use and expand existing areas and facilities as opposed to building new facilities. No new navigation channels are expected to be dredged and no new onshore infrastructure except for pipeline crossings is expected to result from the proposed activities. However, should new navigation channels or infrastructure be proposed, an environmental analysis will be required before approval.

The range of offshore infrastructures installed for hydrocarbon production includes pipelines, fixed and floating platforms, caissons, well protectors, casing, wellheads, and conductors. For all leases issued under the proposed action, BOEM/BSEE estimate the following activity levels:

- WPA: 7-10 installations of production structures annually
- CPA: 30-41 installations of production structures annually
- EPA: 0-1 installations of production structures annually (no more than 2 structures projected to be installed for the entire BOEM/BSEE 40-year planning period)

For the proposed action, approximately 0-12 new pipeline landfalls are anticipated. Most, if not all, of the OCS pipelines installed are expected to tie into existing infrastructure. New pipelines that go ashore will require an environmental analysis before approval. Where pipeline landfall occurs, the permitting process encourages the use of directional boring to greatly reduce and potentially eliminate impacts to barrier beaches. The low number of new pipeline landfalls and the use of modern nonintrusive installation methods will significantly reduce adverse impacts to sensitive coastal areas.

Some OCS structures have either reached the lifetime of their lease, or are no longer producing hydrocarbon sources. Companies must submit a plan to decommission any structures that will no longer be used for oil and gas activities including the method of removal (e.g. mechanical severance and/or explosive severance). For all leases issued under the proposed action, BSEE estimates the following decommissioning activity levels:

- WPA/CPA: 100-200 structure removals annually
- EPA: None expected

Associated Vessel and Helicopter Traffic

About one percent of the oil produced during the OCS oil and gas activities in both the WPA and CPA is expected to be barged to shore. For the EPA, no tankering of products is anticipated as all EPA production is expected to utilize subsea tieback to the CPA network. Over the 40-year life of the leases, less than one percent of the total oil produced is expected to be barged. Pipelines are the primary method used to transport a variety of liquid and gaseous products between OCS productions sites and onshore facilities around the GOM.

Service vessels are one of the primary modes of transporting personnel and supplies between service bases and offshore platforms, drilling rigs, derrick barges, and pipeline construction barges. BOEM/BSEE anticipates the following vessel traffic activity level:

- WPA: 12,025-18,000 service vessel trips (in round trips) annually
- CPA: 70,075-90,675 service vessel trips (in round trips) annually
- EPA: 12-875 service vessel trip (in round trips) annually

Helicopters are a primary mode of transporting personnel between service bases and offshore platforms, drilling rigs, derrick barges, and pipeline construction barges. For all leases issued under the proposed action, BOEM/BSEE estimate the following annual helicopter activity levels:

- WPA: 130,500-261,250 helicopter trips (in round trips) annually
- CPA: 594,500-1,112,500 helicopter trips (in round trips) annually
- EPA: 0-2 helicopter trips (in round trips) annually

Operational Wastes Discharged Offshore

The U.S. Environmental Protection Agency (USEPA), through National Pollutant Discharge Elimination System (NPDES) general permits issued by the USEPA region that has jurisdictional oversight, regulates waste stream discharges generated from offshore oil and gas activities. The primary operational waste discharges generated during offshore oil and gas exploration and development are drilling fluids, drill cuttings, various waters (e.g., bilge, ballast, fire, and cooling), deck drainage, sanitary wastes, and domestic wastes. During production activities, additional waste streams include produced water, produced sand, and well treatment, workover, and completion fluids. Minor additional discharges occur from numerous sources. These discharges may include desalination slurry, several fluids used in subsea production, and uncontaminated freshwater and saltwater. An OCS lessee or operator, as an individual applicant, submits a notice of intent to the USEPA if they intend to make any discharges covered under the NPDES general permits.

Post-authorization Compliance Activities

BSEE conducts on-site inspections of all oil and gas operations on the OCS to ensure that safety and pollution-prevention requirements of the regulations are met and to oversee industry compliance with all applicable lease terms, approved plans and permits, and Notices To Lessees and Operators (NTLs). In addition to the inspection program, BSEE also conducts field investigations including government unannounced inspection exercises, research, and monitoring that centers on environmental regulations and activity-specific conditions/mitigation imposed to assure environmental compliance.

Oil Spills Related to Outer Continental Shelf Oil and Gas Activities

Within the BA, BOEM/BSEE indicate that a large/catastrophic oil spill associated with OCS-related activities in the GOM is a low-probability event and, therefore, is neither a direct nor an indirect effect of the proposed action since it is not reasonably certain to occur, particularly after the implementation of new safety measures and advances in containment technologies after the DWH event. Beginning in the 1980s, BOEM/BSEE have established comprehensive pollution prevention requirements that include redundant safety systems, as well as inspecting and testing requirements to confirm that those devices are working properly. BOEM/BSEE will continue to consider potential regulatory changes and additional safety standards as appropriate.

When comparing the most recent 15-year data (1996-2010) to the last 15-years data in the previous OCS platform analysis (Anderson and La Belle 2000: 1985 through 1999 data) spill rates increased from 0.13 to 0.25 spills per barrel of oil (BBO) for spills $\geq$1,000 barrels and increased from 0.05 to 0.13 spills per BBO for spills $\geq$10,000 barrels. Those rates include a spill from Hurricane Rita in 2005 and the DWH spill in 2010. Prior to those two spills, the last OCS platform spill $\geq$1,000 barrels occurred in November 1980 (1,456 barrels) and the last OCS platform spill $\geq$10,000 barrels occurred in December 1970 (53,000 barrels). When examining the record over the last 15 years (1996 through 2010) for OCS pipelines, the spill rates dropped from 1.38 to 0.88 spills per BBO for spills $\geq$1,000 and from 0.34 to 0.18 spills per BBO for spills $\geq$10,000 barrels.

Oil Spill Prevention/Mitigation Measures

BOEM/BSEE have regulations to ensure that lessees do not create conditions that will pose an unreasonable risk to public health, life, property, aquatic life, wildlife, recreation, navigation, commercial fishing, or other uses of the ocean during offshore oil and gas operations. In light of the DWH explosion, oil spill, and response, the federal government, along with industry, modified and added rules and safety measures related to oil-spill prevention, containment, and response. In addition, the federal government and industry have advanced their research in response to DWH through government-funded research, industry-funded research, and joint partnerships. Those joint partnerships are often between government agencies, industry, and nongovernmental organizations.

New BSEE regulations are aimed at reducing the probability of an oil spill occurring and new safeguards were put in place to protect the environment. These new safety measures include heightened drilling safety standards to reduce the chances that a loss of well control would occur, as well as a new focus on containment capabilities in the event of an oil spill. The following are some examples of BOEM/BSEE regulations related to safety and reduction in oil spill probability:

- 30 Code of Federal Regulations (CFR) 250.130 indicates that BSEE will inspect OCS facilities and any vessels engaged in drilling or other downhole operations. The purpose of these inspections is to verify that operations are being conducted according to the OCS Lands Act, the regulations, the lease, right-of-way, the BOEM -approved EPs or DPPs, or right-of-use and easement, and other applicable laws and regulations; and to determine whether equipment designed to prevent or ameliorate blowouts, fires, spillages, or other major accidents has been installed and is operating properly.
- 30 CFR 250.168 grants BSEE authority to suspend operations for reasons related to both safety and to compliance issues.
- 30 CFR 250.187 requires incident reporting by operators so that all incidents may be tracked and reviewed with the intent of preventing a repeat of the incident.
- 30 CFR 250.201, 286, and 400 all relate to what information needs to be included in plans submitted to BSEE for review and approval, deep water operations plans and drilling operations requirements such as blowout preventers.
- 30 CFR 250, Subpart H pertains to oil and gas production safety systems and states that production safety equipment shall be designed, installed, used, maintained, and tested in a manner to assure the safety and protection of the human, marine, and coastal environments.
- 30 CFR 250, Subparts I and J regulate platforms, structures, and pipelines.
- 30 CFR 250, Subpart O pertains to well control and production safety training.
- 30 CFR 250, Subpart S governs Safety and Environmental Management Systems (SEMS) programs. All operators are required to have a SEMS program, the goal of which is to promote safety and environmental protection by ensuring all personnel aboard a facility are complying with the policies and procedures identified in the SEMS.

BSEE's responsibilities under the Oil Pollution Act of 1990 (OPA) include spill prevention in federal and state offshore waters, review and approval of oil spill response plans (OSRPs), inspection of oil-spill containment and cleanup equipment, and ensuring oil spill financial responsibility. BSEE regulations (30 CFR 254) require that all owners and operators of oil handling, storage, or transportation facilities located seaward of the coastline submit an OSRP for approval. The regulation at 30 CFR 254.2 requires that an OSRP be submitted and approved before an operator can use a facility, or the operator must certify in writing to BSEE that it is capable of responding to a "worst-case" spill or the substantial threat of such a spill. The facility must be operated in compliance with the approved OSRP or the BSEE-accepted "worst-case" spill certification. Owners or operators of offshore pipelines are required to submit an OSRP for any pipeline that carries oil, condensate, or gas with condensate; pipelines carrying essentially dry gas do not require an OSRP. The OSRP describes how an operator intends to respond to an oil spill. The OSRP may be site-specific or regional. The Emergency Response Action Plan within the OSRP outlines the availability of spill containment and cleanup equipment and trained personnel. It must ensure that full-response capability can be deployed during an oil spill incident. The OSRP includes an inventory of appropriate equipment and materials, their availability, and the time needed for deployment. All BSEE-approved OSRP's must be reviewed at least every 2 years and all resulting modifications must be submitted to BSEE within 15 days whenever:

1. a change occurs that appreciably reduces an owner/operator's response capabilities;
2. a substantial change occurs in the worst-case discharge scenario or in the type of oil being handled, stored, or transported at the facility;
3. there is a change in the name(s) or capabilities of the oil spill removal organizations cited in the OSRP; or
4. there is a change in the applicable Area Contingency Plans.

The responsible party for every covered offshore facility must demonstrate oil spill financial responsibility (OSFR) as required by OPA 90 (30 CFR 253). A covered offshore facility is any structure and all of its components, equipment, pipeline, or device (other than a vessel or a pipeline or deepwater port licensed under the Deepwater Port Act of 1974) used for exploring, drilling, or producing oil, or for transporting oil from such facilities. BOEM/BSEE ensure that each responsible party has sufficient funds for removal costs and damages resulting from the accidental release of liquid hydrocarbons into the environment for which the responsible party is liable.

In the absence of swift and effective action by the responsible party for a spill, the U.S. Coast Guard (USCG) will initiate action pursuant to the OPA 90 to control and clean up a spill offshore under area plans which have been developed for this contingency.

BOEM indicates that each oil spill event is unique and that its outcome depends on several factors, including time of year and location of the release relative to winds, currents, land, and sensitive resources as well as specifics of the well and response effort. BOEM also indicates that the severity of impacts from an oil spill cannot be predicated on volume alone. Under NEPA, BOEM analyzes a low probability catastrophic event in conjunction with its analysis of potential effects, as requested by the Council on Environmental Quality (CEQ) pursuant to its regulation at 40 C.F.R. 1502.22. The CEQ (2010) recommended that BOEM should "ensure that NEPA documents provide decision makers with a robust analysis of reasonably foreseeable impacts, including an analysis of reasonably foreseeable impacts associated with low probability catastrophic spills for oil and gas activities on the Outer Continental Shelf." For purposes of the Act, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action.

Other Mitigation Measures

According to the BA, BOEM/BSEE and other federal and state agencies that manage coastal natural resources will continue to require or oversee the implementation of conservation and mitigation measures that should prevent or minimize impacts of the proposed project not associated with oil spills to listed species.

Federal Aviation Administration (FAA) and corporate helicopter policy regarding flight altitudes should remove or minimize noise impacts and the number of aircraft strikes to listed coastal birds. The FAA and corporate helicopter policy advise helicopters to maintain a minimum altitude of 700 feet while in transit offshore and 500 feet while working between platforms. When flying over land, the specified minimum altitude is 1,000 feet over unpopulated areas or across coastlines and 2,000 feet over populated areas and biologically sensitive areas such as wildlife refuges, national parks, and national seashores.

Federal environmental laws and regulations require coastal development (e.g., pipelines, navigation channels, docks, and gas-processing plants) associated with the proposed project with a federal nexus to avoid or minimize project impacts to listed species and their critical habitats. During a separate review process the Service and other agencies have the opportunity to provide conditions and recommendations to project plans. In addition, environmental regulations require replanting and restoration of wetlands destroyed by pipe-laying barges and associated onshore pipeline installation.

There are numerous existing laws, regulations, and enforcement guidelines that prohibit and discourage the disposal of solid debris in Gulf waters that can impact listed species and their critical habitats. For example, BSEE prohibits the disposal of equipment, containers, and other materials into coastal and offshore waters by lessees (30 CFR 250.300). Also, BSEE NTL No. 2015-G03 requires annual awareness training to minimize the unintentional loss of debris from industry structures or vessels. BSEE inspectors routinely conduct site visits and issue citations for noncompliance. In addition, MARPOL, Annex V. Public Law 100-220 (101 Statute 1458), which prohibits the disposal of any plastics, garbage, and other solid wastes at sea or in coastal waters, went into effect January 1, 1989, and is enforced by the USCG.

The Marine Debris Research, Prevention, and Reduction Act (MDRPRA [P.L 109-449]) was enacted in December 2006 and amended in 2012. The purposes of MDRPRA are to help identify, determine sources of, assess, prevent, reduce, and remove marine debris and address the adverse impacts of marine debris on the economy of the U.S., marine environment, and navigation safety; to reactivate the Interagency Marine Debris Coordinating Committee; and to develop a federal marine debris information clearinghouse. The MDRPRA established, within the National Oceanic and Atmospheric Administration (NOAA), a Marine Debris Prevention and Removal Program to reduce and prevent the occurrence and adverse impacts of marine debris on the marine environment and navigation safety. Greatly improved handling of waste and trash by industry, along with the annual awareness training required by the marine debris mitigations, is decreasing OCS-related debris in the ocean and reducing the devastating effects on listed species.

BOEM/BSEE have determined that emissions of pollutants into the atmosphere from the activities associated with OCS oil and gas activities should result in minimal effects on offshore and onshore air quality because of the prevailing atmospheric conditions, emission heights and rates, and pollutant concentrations. Subsequently, impacts to listed species are expected to be negligible because air quality impacts from oil and gas activities are not likely to impact ambient air quality.

Operational discharges such as produced water, drilling muds and cuttings are regulated by the EPA through the NPDES program.

Spill Risk

The risk of contact to listed species from offshore spills related to the proposed action operations is dependent upon the likelihood that a spill occurs, the likelihood that the spilled oil reaches the areas inhabited or used by these species, and the likelihood that oil spill contact occurs during the period that a particular listed species is present in the area. Because oil spills may occur from activities associated with OCS activities, the BOEM conducted a formal OSRA for offshore oil spills.

Estimates from spill data show that federal offshore waters may be subjected to many frequent small spills ($\leq$1 barrel); few, infrequent, moderately-sized spills (>1 barrel and <1,000 barrels); and/or rare large spills (>1,000 barrels and $\leq$10,000 barrels) as a result of OCS oil and gas activities. The number of small spills ($\leq$1 barrel) estimated to occur in the WPA ranges from 234-404, in the CPA from 929-1,806 and in the EPA <1-143. As the spill size increases the number of spills estimated decreases and the median spill size increases.

As stated above, BOEM/BSEE anticipate that the most frequent spills associated with OCS oil and gas activities are generally less than 1 barrel in size. These spills are so small and of short duration that impacts to individuals of a species are expected to be extremely limited. In addition, spills less than 1,000 barrels are not expected to persist as a slick on the water surface of the water beyond a few days. Because spills in the OCS would occur at least 3 miles from shore, it is unlikely that any spills would make landfall prior to breaking up. For an offshore spill < 1,000 barrels to make landfall, the spill would have to occur proximate to state waters (defined as 3 to 12 miles from shore). If a spill were to occur proximate to state waters, only a spill greater than 50 barrels would be expected to have a chance of persisting long enough to reach land. Spills greater than 50 barrels and less than 1,000 barrels are very infrequent. Should such a spill occur, the volume that would make landfall would be expected to be extremely small (i.e., a few barrels).

The probabilities of offshore spills equal to or greater than 1,000 barrels occurring and contacting within 30 days any of the species and habitats modeled range between <0.5 and 25 percent. The risk probabilities to listed species generated by the OSRA model do not include the many spill response mitigations put in place and regulated by BOEM/BSEE and other federal and state agencies intended to prevent or minimize the chance of spilled oil from reaching land and to reduce impacts if oil does reach land. Such mitigations should further reduce the probabilities of offshore spills affecting listed species.

When comparing the most recent 15 year data (1996 through 2010 data) to the last 15 year data in the previous OCS platform analysis (Anderson and LaBelle 2000: 1985 through 1999 data) spill rates increased from 0.13 to 0.25 spills per billion barrel of oil for spills ≥1,000 barrels, and increased from 0.05 to 0.13 spills per billion barrel of oil for spills ≥10,000 barrels. Although the spill rates have increased, they are still relatively low and include a spill from Hurricane Rita (2005) and the DWH spill in 2010. Prior to these two spills, the last OCS platform spill ≥1,000 barrels occurred in November 1980 (1,456 barrels) and the last OCS platform spill ≥10,000 barrels occurred in December 1970 (53,000 barrels).

As the DWH explosion and oil spill illustrate though, the risk for a catastrophic spill to occur is low, but not zero. Within the subject BA, BOEM/BSEE attempted to conservatively describe this risk. BOEM/BSEE estimated the frequency of OCS crude and condensate spills exceeding a specified spill size on a per well drilled basis based on historical frequency and not considering any new regulatory reforms and safety measures that could further reduce the risk.

BOEM/BSEE indicate in the BA, that there has been a noticeable downward trend in the number of wells drilled, as technological advances have been made allowing for higher rates of production from a fewer number of wells. BOEM/BSEE expect this trend to continue into the foreseeable future. Accordingly, BOEM/BSEE expect the number of wells drilled per year to continue to decline with an estimated 300 to 500 wells drilled per year.

Spill rates were calculated based on the assumption that spills occur in direct proportion to the volume of oil handled and are expressed as the number of spills per BBO handled. Using the OSRA model, the probabilities were calculated of a particular number of offshore spills ≥1,000 barrels resulting from OCS oil and gas activities during the 40-year NEPA analysis period (2012-2017 Multisale EIS). For WPA OCS oil and gas activities, there is an 11-18 percent chance of one spill ≥1,000 barrels occurring, a 1-2 percent chance of two spills ≥1,000 barrels occurring, and a 12-20 percent chance of one or more spills ≥1,000 barrels occurring in the WPA.   For CPA OCS oil and gas activities, there is a 31-37 percent chance of one spill ≥1,000 barrels occurring, an 8-18 percent chance of two spills ≥1,000 barrels occurring, a 1-6 percent chance of three spills ≥1,000 barrels occurring, and a 0-1 percent chance of four spills ≥1,000 barrels occurring. Overall, there is a 41-62 percent chance of one or more spills ≥1,000 barrels occurring in the CPA. For EPA OCS oil and gas activities, there is a 0-7 percent chance of one spill ≥1,000 barrels occurring, a <0.5 percent chance of two spills ≥1,000 barrels occurring, and a <0.5 percent chance of three spills ≥1,000 barrels occurring. Overall, there is a 0-8 percent chance of one or more spills ≥1,000 barrels occurring.

Offshore response and cleanup is preferable to shoreline cleanup; however, if an oil slick reaches the coastline it is expected that the specific shoreline cleanup countermeasures identified and prioritized in the appropriate Area Contingency Plans for various habitat types would be used. The sensitivity of the contaminated shoreline is the most important factor in the development of cleanup recommendations. Shorelines of low productivity and biomass can withstand more intrusive cleanup methods such as pressure washing. Shorelines of high productivity and biomass are very sensitive to intrusive cleanup methods, and in many cases, the cleanup is more damaging than allowing natural recovery.

Oil spill cleanup operations can affect barrier beach stability. If large quantities of sand were to be removed during spill-cleanup operations, a new beach profile and sand configuration would be established in response to the reduced sand supply and volume. The net result of those changes could be accelerated rates of shoreline erosion, especially in a sand-starved, eroding-barrier setting such as found along the Louisiana Gulf Coast. To address those possible impacts, the Gulf Coast States have established policies to limit sand removal by cleanup operations. Oil on the beach may be cleaned up manually, mechanically, or by using both methods. Removal of sand during cleanup is expected to be minimized to avoid significantly reducing sand volumes. Some oil will likely remain on the beach at varying depths and may persist for several years as it slowly biodegrades and volatilizes.

BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to federally listed species associated with a spill of this magnitude are not addressed in this BO.

STATUS OF THE SPECIES/CRITICAL HABITAT

Sea turtles

Loggerhead sea turtle (*Caretta caretta*)
Kemp's ridley sea turtle (*Lepidochelys kempii*)

General information (Loggerhead and Kemp's ridley sea turtle)

The Service has responsibility for conserving sea turtles when they come ashore to nest. NMFS has jurisdiction over sea turtles in the marine environment. In applying the jeopardy standard under the Act, the Service has determined that sea turtle species occurring in the U.S. represent populations that qualify for separate consideration under section 7. Even though sea turtles are wide-ranging and have distributions outside the U.S., a jeopardy finding could be made when a proposed action, along with cumulative effects, is likely to jeopardize a sea turtle species' U.S. population.

The reproductive strategy of sea turtles involves producing large numbers of offspring to compensate for the high natural mortality during their first several years of life; however, increased unnatural mortality is occurring due to increased human-caused impacts on sea turtle populations. Therefore, activities that affect the behavior and/or survivability of turtles on their remaining nesting beaches, particularly the few remaining high-density nesting beaches, could seriously reduce the ability to conserve sea turtle populations.

Loggerhead sea turtle

Species/critical habitat description

The loggerhead sea turtle, which occurs throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans, was federally listed worldwide as a threatened species on July 28, 1978 (43 Federal Register [FR] 32800). On September 22, 2011, the loggerhead sea turtle's listing under the Act was revised from a single threatened species to nine DPSs listed as either threatened or endangered. The nine DPSs and their statuses are:

Northwest Atlantic Ocean DPS – threatened
Northeast Atlantic Ocean – endangered
Mediterranean Sea DPS – endangered
South Atlantic Ocean DPS – threatened
North Pacific Ocean DPS – endangered
South Pacific Ocean DPS – endangered
North Indian Ocean DPS – endangered
Southwest Indian Ocean – threatened
Southeast Indo-Pacific Ocean DPS – threatened

The loggerhead sea turtle grows to an average weight of about 200 pounds and is characterized by a large head with blunt jaws. Adults and subadults have a reddish-brown carapace. Scales on the top of the head and top of the flippers are also reddish-brown with yellow on the borders. Hatchlings are a dull brown color (NMFS 2009). The loggerhead feeds on mollusks, crustaceans, fish, and other marine animals.

The loggerhead may be found hundreds of miles out to sea, as well as in inshore areas such as bays, lagoons, salt marshes, creeks, ship channels, and the mouths of large rivers. Coral reefs, rocky places, and ship wrecks are often used as feeding areas. Within the Northwest Atlantic, the majority of nesting activity occurs from April through September, with a peak in June and July (Williams-Walls et al. 1983, Dodd 1988, Weishampel et al. 2006). Nesting occurs within the Northwest Atlantic along the coasts of North America, Central America, northern South America, the Antilles, Bahamas, and Bermuda, but is concentrated in the southeastern United States and on the Yucatán Peninsula in Mexico on open beaches or along narrow bays having suitable sand (Sternberg 1981, Ehrhart 1989, Ehrhart et al. 2003, NMFS and FWS 2008).

Critical habitat for the loggerhead sea turtle was designated on July 10, 2014 within the Northwest Atlantic Ocean DPS. Specifically, within the proposed action area critical habitat occurs in: 1) Bay, Charlotte, Collier, Escambia, Franklin, Gulf, Lee, Manatee, Monroe, and Sarasota Counties, Florida; 2) Baldwin County, Alabama; and 3) Jackson County, Mississippi. Within these areas, the primary constituent elements (PCEs) of the physical or biological features essential to the conservation of the Northwest Atlantic Ocean DPS of the loggerhead sea turtle are the extratidal or dry sandy beaches from the mean high-water line to the toe of the secondary dune, which are capable of supporting a high density of nests or serving as an expansion area for beaches with a high density of nests and that are well distributed within each state, or region within a state, and representative of total nesting, consisting of four components: 1) suitable nesting beach habitat that has relatively unimpeded nearshore access from the ocean to the beach for nesting females and from the beach to the ocean for both post-nesting females and hatchlings and is located above mean high water to avoid being inundated frequently by high tides; 2) sand that allows for suitable nest construction, is suitable for facilitating gas diffusion conducive to embryo development, and is able to develop and maintain temperatures and a moisture content conducive to embryo development; 3) suitable nesting beach habitat with sufficient darkness to ensure that nesting turtles are not deterred from emerging onto the beach and hatchlings and post-nesting females orient to the sea; and 4) natural coastal processes or artificially created or maintained habitat mimicking natural conditions.

Life history

Loggerheads are long-lived, slow-growing animals that use multiple habitats across entire ocean basins throughout their life history. This complex life history encompasses terrestrial, nearshore, and open ocean habitats. The three basic ecosystems in which loggerheads live are the:

1. Terrestrial zone (supralittoral) - the nesting beach where both oviposition (egg laying) and embryonic development and hatching occur.
2. Neritic zone - the inshore marine environment (from the surface to the sea floor) where water depths do not exceed 656 feet. The neritic zone generally includes the continental shelf, but in areas where the continental shelf is very narrow or nonexistent, the neritic zone conventionally extends to areas where water depths are less than 656 feet.
3. Oceanic zone - the vast open ocean environment (from the surface to the sea floor) where water depths are greater than 656 feet.

Maximum intrinsic growth rates of sea turtles are limited by the extremely long duration of the juvenile stage and fecundity. Loggerheads require high survival rates in the juvenile and adult stages, common constraints critical to maintaining long-lived, slow-growing species, to achieve positive or stable long-term population growth (Congdon et al. 1993, Heppell 1998, Crouse 1999, Heppell et al. 1999, 2003, Musick 1999).

Numbers of nests and nesting females are often highly variable from year to year due to a number of factors including environmental stochasticity, periodicity in ocean conditions, anthropogenic effects, and density-dependent and density-independent factors affecting survival, somatic growth, and reproduction (Meylan 1982, Hays 2000, Chaloupka 2001, Solow *et al.* 2002). Despite these sources of variation, and because female turtles exhibit strong nest site fidelity, a nesting beach survey can provide a valuable assessment of changes in the adult female population, provided that the study is sufficiently long and effort and methods are standardized (Meylan 1982, Gerrodette and Brandon 2000, Reina et al. 2002). Table 1 summarizes key life history characteristics for loggerheads nesting in the U.S.

Table 1. Typical values of life history parameters for loggerheads nesting in the U.S. (NMFS and FWS 2008).

Life History Trait	**Data**
Clutch size (mean)	100-126 eggs[1]
Incubation duration (varies depending on time of year and latitude)	Range = 42-75 days[2,3]
Pivotal temperature (incubation temperature that produces an equal number of males and females)	84˚F[5]
Nest productivity (emerged hatchlings/total eggs) x 100 (varies depending on site specific factors)	45-70 percent[2,6]
Clutch frequency (number of nests/female/season)	3-4 nests[7]
Internesting interval (number of days between successive nests within a season)	12-15 days[8]
Juvenile (<34 inches Curved Carapace Length) sex ratio	65-70 percent female[4]
Remigration interval (number of years between successive nesting migrations)	2.5-3.7 years[9]
Nesting season	late April-early September
Hatching season	late June-early November
Age at sexual maturity	32-35 years[10]
Life span	>57 years[11]

1 Dodd (1988).
2 Dodd and Mackinnon (1999, 2000, 2001, 2002, 2003, 2004).
3 Witherington (2006) (information based on nests monitored throughout Florida beaches in 2005, n = 865).
4 NMFS (2001); Foley (2005).
5 Mrosovsky (1988).
6 Witherington (2006) (information based on nests monitored throughout Florida beaches in 2005, n = 1,680).
7 Murphy and Hopkins (1984); Frazer and Richardson (1985); Hawkes et al. 2005; Scott 2006.
8 Caldwell (1962), Dodd (1988).
9 Richardson et al. (1978); Bjorndal et al. (1983).
10 Snover (2005).
11 Dahlen et al. (2000).

Loggerheads nest on ocean beaches and occasionally on estuarine shorelines with suitable sand. Nests are typically laid between the high tide line and the dune front (Routa 1968, Witherington 1986, Hailman and Elowson 1992). Wood and Bjorndal (2000) evaluated four environmental factors (slope, temperature, moisture, and salinity) and found that slope had the greatest influence on loggerhead nest-site selection on a beach in Florida. Loggerheads appear to prefer relatively narrow, steeply sloped, coarse-grained beaches, although nearshore contours may also play a role in nesting beach site selection (Provancha and Ehrhart 1987).

The warmer the sand surrounding the egg chamber, the faster the embryos develop (Mrosovsky and Yntema 1980). Sand temperatures prevailing during the middle third of the incubation period also determine the sex of hatchling sea turtles (Mrosovsky and Yntema 1980). Incubation temperatures near the upper end of the tolerable range produce only female hatchlings while incubation temperatures near the lower end of the tolerable range produce only male hatchlings.

Loggerhead hatchlings pip and escape from their eggs over a 1- to 3-day interval and move upward and out of the nest over a 2- to 4-day interval (Christens 1990). The time from pipping to emergence ranges from 4 to 7 days with an average of 4.1 days (Godfrey and Mrosovsky 1997). Hatchlings emerge from their nests en masse almost exclusively at night, and presumably using decreasing sand temperature as a cue (Hendrickson 1958, Mrosovsky and Shettleworth 1968, Witherington et al. 1990). Moran et al. (1999) concluded that a lowering of sand temperatures below a critical threshold, which most typically occurs after nightfall, is the most probable trigger for hatchling emergence from a nest. After an initial emergence, there may be secondary emergences on subsequent nights (Carr and Ogren 1960, Witherington 1986, Ernest and Martin 1993, Houghton and Hays 2001).

Hatchlings use a progression of orientation cues to guide their movement from the nest to the marine environments where they spend their early years (Lohmann and Lohmann 2003). Hatchlings first use light cues to find the ocean. On naturally lighted beaches without artificial lighting, ambient light from the open sky creates a relatively bright horizon compared to the dark silhouette of the dune and vegetation landward of the nest. This contrast guides the hatchlings to the ocean (Daniel and Smith 1947, Limpus 1971, Salmon et al. 1992, Witherington and Martin 1996, Witherington 1997, Stewart and Wyneken 2004).

Population dynamics

The loggerhead occurs throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd 1988). However, the majority of loggerhead nesting is at the western rims of the Atlantic and Indian Oceans. The most recent reviews show that only two loggerhead nesting beaches have greater than 10,000 females nesting per year (Baldwin et al. 2003, Ehrhart et al. 2003, Kamezaki et al. 2003, Margaritoulis et al. 2003): Peninsular Florida (U.S.) and Masirah (Oman).

The loggerhead is commonly found throughout the North Atlantic including the GOM, the northern Caribbean, the Bahamas archipelago, and eastward to West Africa, the western Mediterranean, and the west coast of Europe.

The major nesting concentrations in the U.S. are found in south Florida. However, loggerheads nest from Texas to Virginia. Total estimated nesting in the U.S. has fluctuated between 49,000 and 90,000 nests per year from 1999-2010 (NMFS and Service 2008, FWC/FWRI 2010a).

About 80 percent of loggerhead nesting in the southeast U.S. occurs in six Florida counties (Brevard, Indian River, St. Lucie, Martin, Palm Beach, and Broward Counties). Adult loggerheads are known to make considerable migrations between foraging areas and nesting beaches (Schroeder et al. 2003, Foley et al. 2008). During non-nesting years, adult females from U.S. beaches are distributed in waters off the eastern U.S. and throughout the Gulf of Mexico, Bahamas, Greater Antilles, and Yucatán.

Status and distribution

Five recovery units have been identified in the northwest Atlantic based on genetic differences and a combination of geographic distribution of nesting densities, geographic separation, and geopolitical boundaries (NMFS and Service 2008). Recovery units are subunits of a listed species that are geographically or otherwise identifiable and essential to the recovery of the species. Recovery units are individually necessary to conserve genetic robustness, demographic robustness, important life history stages, or some other feature necessary for long-term sustainability of the species. The five recovery units identified in the Northwest Atlantic are:

1. Northern Recovery Unit (NRU) - defined as loggerheads originating from nesting beaches from the Florida-Georgia border through southern Virginia (the northern extent of the nesting range);
2. Peninsula Florida Recovery Unit (PFRU) - defined as 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 (DTRU) - defined as loggerheads originating from nesting beaches throughout the islands located west of Key West, Florida;
4. Northern Gulf of Mexico Recovery Unit (NGMRU) - defined as loggerheads originating from nesting beaches from Franklin County on the northwest Gulf coast of Florida through Texas; and
5. Greater Caribbean Recovery Unit (GCRU) - composed of loggerheads originating from all other nesting assemblages within the Greater Caribbean (Mexico through French Guiana, The Bahamas, Lesser Antilles, and Greater Antilles).

The PFRU is the largest loggerhead recovery unit within the Northwest Atlantic Ocean DPS and represents approximately 87 percent of all nesting effort in the DPS (Ehrhart et al. 2003). A near-complete nest census of the PFRU undertaken from 1989 to 2007 revealed a mean of 64,513 loggerhead nests per year representing approximately 15,735 females nesting per year (4.1 nests per female [Murphy and Hopkins 1984; FWC 2008; NMFS and Service 2008]). This near-complete census provides the best statewide estimate of total abundance, but because of variable survey efforts, these numbers cannot be used to assess trends. Loggerhead nesting trends are best assessed using standardized nest counts made at Index Nesting Beach Survey (INBS) sites surveyed with constant effort over time. In 1979, the Statewide Nesting Beach Survey (SNBS) program was initiated to document the total distribution, seasonality, and abundance of sea turtle nesting in Florida. In 1989, the INBS program was initiated in Florida to measure seasonal productivity, allowing comparisons between beaches and between years (FWC 2009b). Of the 190 SNBS surveyed areas, 33 participate in the INBS program (representing 30 percent of the SNBS beach length).

Using INBS nest counts, a significant declining trend was documented for the PFRU, where nesting declined 26 percent over the 20-year period from 1989–2008, and declined 41 percent over the period 1998-2008 (NMFS and Service 2008). However, with the addition of nesting data through 2010, the nesting trend for the PFRU did not show a nesting decline statistically different from zero (76 FR 58868, September 22, 2011).

The NGMRU is the third largest nesting assemblage among the four U.S. recovery units. Nesting surveys conducted on approximately 186 miles of beach within the NGMRU (Alabama and Florida only) were undertaken between 1995 and 2007 (statewide surveys in Alabama began in 2002). The mean nest count during this 13-year period was 906 nests per year, which equates to about 221 females nesting per year (4.1 nests per female, Murphy and Hopkins 1984, (FWC 2008, NMFS and Service 2008). Evaluation of long-term nesting trends for the NGMRU is difficult because of changed and expanded beach coverage. Loggerhead nesting trends are best assessed using standardized nest counts made at INBS sites surveyed with constant effort over time. Using Florida INBS data for the NGMRU (FWC 2008), a log-linear regression showed a significant declining trend of 4.7 percent annually from 1997-2008 (NMFS and Service 2008).

The DTRU, located west of the Florida Keys, is the smallest of the identified recovery units. A near-complete nest census of the DTRU was undertaken from 1995 to 2004, excluding 2002, (9 years surveyed) revealed a mean of 246 nests per year, which equates to about 60 females nesting per year (4.1 nests per female, Murphy and Hopkins 1984) (FWC 2008, NMFS and Service 2008). The nesting trend data for the DTRU are from beaches that are not part of the INBS program, but are part of the SNBS program. A simple linear regression of 1995-2004 nesting data, accounting for temporal autocorrelation, revealed no trend in nesting numbers. Because of the annual variability in nest totals, it was determined that a longer time series is needed to detect a trend (NMFS and Service 2008).

Kemp's ridley sea turtle

Species/critical habitat description

The Kemp's ridley sea turtle was federally listed as endangered on December 2, 1970 (35 FR 18320). The Kemp's ridley, along with the flatback sea turtle (*Natator depressus*), has the most geographically restricted distribution of any sea turtle species. The range of the Kemp's ridley includes the Gulf coasts of Mexico and the U.S., and the Atlantic coast of North America as far north as Nova Scotia and Newfoundland.

Adult Kemp's ridleys and olive ridleys are the smallest sea turtles in the world. The weight of an adult Kemp's ridley is generally between 70 to 108 pounds with a carapace measuring approximately 24 to 26 inches in length (Heppell et al. 2005). The carapace is almost as wide as it is long. The species' coloration changes significantly during development from the grey-black dorsum and plastron of hatchlings, a grey-black dorsum with a yellowish-white plastron as post-pelagic juveniles and then to the lighter grey-olive carapace and cream-white or yellowish plastron of adults. Their diet consists mainly of swimming crabs, but may also include fish, jellyfish, and an array of mollusks.

The Kemp's ridley has a restricted distribution. Nesting is essentially limited to the beaches of the western GOM, primarily in Tamaulipas, Mexico (NMFS et al. 2011). Nesting also occurs in Veracruz and a few historical records exist for Campeche, Mexico (Marquez-Millan 1994). Nesting also occurs regularly in Texas and infrequently in a few other U.S. states. However, historic nesting records in the U.S. are limited to south Texas (Werler 1951, Carr 1961, Hildebrand 1963).

Most Kemp's ridley nests located in the U.S. have been found in south Texas, especially Padre Island (Shaver and Caillouet 1998; Shaver 2002, 2005). Nests have been recorded elsewhere in Texas (Shaver 2005, 2006a, 2006b, 2007, 2008), and in Florida (Johnson et al. 1999, Foote and Mueller 2002, Hegna et al. 2006, FWC/FWRI 2010b), Alabama (J. Phillips, FWS, personal communication, 2007 cited in NMFS et al. 2011; J. Isaacs, Service, personal communication, 2008 cited in NMFS et al. 2011), Georgia (Williams et al. 2006), South Carolina (Anonymous 1992), and North Carolina (Marquez et al. 1996), but these events are less frequent. Kemp's ridleys inhabit the GOM and the Northwest Atlantic Ocean, as far north as the Grand Banks (Watson et al. 2004) and Nova Scotia (Bleakney 1955). They occur near the Azores and eastern north Atlantic (Deraniyagala 1938, Brongersma 1972, Fontaine et al. 1989, Bolten and Martins 1990) and Mediterranean (Pritchard and Marquez 1973, Brongersma and Carr 1983, Tomas and Raga 2007, Insacco and Spadola 2010).

Hatchlings, after leaving the nesting beach, are believed to become entrained in eddies within the GOM. Most Kemp's ridley post-hatchlings likely remain within the GOM. Others are transported into the northern GOM and then eastward, with some continuing southward in the Loop Current, then eastward on the Florida Current into the Gulf Stream (Collard and Ogren 1990, Putman et al. 2010). Juvenile Kemp's ridleys spend on average 2 years in the oceanic zone (NMFS SEFSC unpublished preliminary analysis, July 2004, as cited in NMFS et al. 2011) where they likely live and feed among floating algal communities. They remain here until they reach about 7.9 inches in length (approximately 2 years of age), at which size they enter coastal shallow water habitats (Ogren 1989); however, the time spent in the oceanic zone may vary from 1 to 4 years or perhaps more (TEWG 2000, Baker and Higgins 2003, Dodge et al. 2003).

No critical habitat has been designated for the Kemp's ridley sea turtle.

Life history

Nesting occurs primarily from April into July. Nesting often occurs in synchronized emergences, known as "arribadas" or "arribazones," which may be triggered by high wind speeds, especially north winds, and changes in barometric pressure (Jimenez et al. 2005). Nesting occurs primarily during daylight hours. Clutch size averages 100 eggs and eggs typically take 45 to 58 days to hatch depending on incubation conditions, especially temperatures (Marquez-Millan 1994, Rostal 2007).

Females lay an average of 2.5 clutches within a season (TEWG 1998) and inter-nesting interval generally ranges from 14 to 28 days (Donna Shaver, Padre Island National Seashore, personal communication, 2007 as cited in NMFS et al. 2011). The mean remigration interval for adult females is 2 years, although intervals of 1 and 3 years are not uncommon (Marquez et al. 1982;

TEWG 1998, 2000). Males may not be reproductively active on an annual basis (Wibbels et al. 1991). Age at sexual maturity is believed to be between 10 to 17 years (Snover et al. 2007).

Population dynamics

Most Kemp's ridleys nest on the beaches of the western GOM, primarily in Tamaulipas, Mexico. Nesting also occurs in Veracruz and Campeche, Mexico, although a small number of Kemp's ridleys nest consistently along the Texas coast (NMFS et al. 2011). In addition, rare nesting events have been reported in Alabama, Florida, Georgia, South Carolina, and North Carolina. Historical information indicates that tens of thousands of ridleys nested near Rancho Nuevo, Mexico, during the late 1940s (Hildebrand 1963). The Kemp's ridley population experienced a devastating decline between the late 1940s and the mid-1980s. The total number of nests per nesting season at Rancho Nuevo remained below 1,000 throughout the 1980s, but gradually began to increase in the 1990s. In 2009, 16,273 nests were documented along the 18.6 miles of coastline patrolled at Rancho Nuevo, and the total number of nests documented for all the monitored beaches in Mexico was 21,144 (Service 2010). In 2011, a total of 20,570 nests were documented in Mexico, 81 percent of these nests were documented in the Rancho Nuevo beach (Burchfield and Peña 2011). In addition, 153 and 199 nests were recorded during 2010 and 2011, respectively, in the United States, primarily in Texas.

Status and distribution

Nesting aggregations of Kemp's ridleys at Rancho Nuevo were discovered in 1947, and the adult female population was estimated to be 40,000 or more individuals based on a film by Andres Herrera (Hildebrand 1963, Carr 1963). Within approximately three decades, the population had declined to 924 nests and reached the lowest recorded nest count of 702 nests in 1985. Since the mid-1980s, the number of nests observed at Rancho Nuevo and nearby beaches has increased 15 percent per year (Heppell et al. 2005), allowing cautious optimism that the population is on its way to recovery. This increase in nesting can be attributed to full protection of nesting females and their nests in Mexico resulting from a bi-national effort between Mexico and the U.S. to prevent the extinction of the Kemp's ridley, the requirement to use Turtle Excluder Devices (TEDs) in shrimp trawls both in the U.S. and Mexico, and decreased shrimping effort (NMFS et al. 2011, Heppell et al. 2005).

Analysis of the species/critical habitat likely to be affected

The Service and the NMFS share Federal jurisdiction for sea turtles under the Act. The Service has responsibility for sea turtles on the nesting beach. NMFS has jurisdiction for sea turtles in the marine environment. In accordance with the Act, the Service completes consultations with all federal agencies for actions that may adversely affect sea turtles on the nesting beach. The Service's analysis only addresses activities that may impact nesting sea turtles, their nests and eggs, and hatchlings as they emerge from the nest and crawl to the sea. NMFS assesses and consults with federal agencies concerning potential impacts to sea turtles in the marine environment.

The proposed action has the potential to adversely affect nesting loggerhead and Kemp's ridley females, nests, and hatchlings within the proposed project area. Critical habitat for loggerhead sea turtles may also be impacted within the action area; however, because critical habitat has not been designated for the Kemp's ridley, none will be affected. The effects of the proposed action on sea turtles will be considered further in the remaining sections of this BO. Potential sources of impacts to nesting loggerhead and Kemp's ridley sea turtles from existing and proposed oil and gas activities are loss of nesting habitat and disturbance of nests, and trash and debris.

ENVIRONMENTAL BASELINE (Loggerhead and Kemp's ridley sea turtle)

Status of the species within the action area

Kemp's ridley sea turtles have been documented in recent years along South Padre Island National Seashore in Texas. On the Texas coast 251 Kemp's ridley nests were recorded from 2002 to 2006. For the 2007 nesting season, 128 nests had been recorded. For 2008, 2009, 2010, 2011, 2012, 2013, and 2014 the nests in Texas totaled 195, 197, 141, 199, 209, 153, and 119, respectively (National Park Service [NPS] 2015). The 2015 5-year Summary and Evaluation for the Kemp's ridley (NMFS and Service 2015) states the 2014 population of the species within the GOM as: 7,272 nests in Rancho Nuevo; 1,381 in Tepehuajes; and 2,333 in Playa Dos, Mexico. NMFS and Service (2015) have the 2014 Padre Island National population as 119 nests.

As discussed above, there are five recovery units within the Northwest Atlantic Ocean (NWA) DPS for the loggerhead sea turtle. Of those five recovery units, the NGMRU and the PFRU are contained within the action area. The 2009 Turtle Expert Working Group report indicated that the average number of nests for the four recovery units within the United States was 72,311 nests with the NGMRU accounting for 1,000 nests or 1.4% of the NWA DPS. The NGMRU is the most at-risk recovery unit within the DPS. Peninsular Florida represents the largest loggerhead nesting aggregation in the Atlantic Ocean, representing as much as 80% of all nesting and producing 90% of all hatchlings. The number of nests declined since peaking in 1998 (with 59,918 nests on index beaches). These beaches represent about 25% of all nesting habitat but about 70% of the total number of nests. In 2010, Alabama had reported 421 loggerhead nests. A total of 84 nests in 2011 along the Gulf coast were discovered. Tropical Storm Lee, however, inundated several nests. In the 2012, 2013, 2014, and 2015 seasons, a total of 149, 81, 80, and 109 nests were reported along the Alabama shores, respectively (Share the Beach 2015). Louisiana and Mississippi have few if any nests.

Factors affecting species environment within the action area

For at least two decades, several factors have contributed to the decline of sea turtle populations along the Atlantic and Gulf coasts. Turtles have been victims of commercial over-utilization of eggs and turtle parts, incidental catches during commercial fishing operations, disturbance of nesting beaches by coastal housing, and marine pollution and debris. The reproductive strategy of sea turtles involves producing large numbers of offspring to compensate for high natural mortality through the first several years of life; however, excessive exploitation of turtles has increased mortality beyond that which can be compensated for by high natality. Therefore, activities that continue to affect the survivability of turtles on their remaining nesting beaches, particularly the high-density nesting beaches, will seriously reduce the Service's ability to conserve sea turtles. Today, under strict protection, the populations appear to be in the early stages of recovery.

EFFECTS OF THE ACTION (Loggerhead and Kemp's ridley sea turtle)

Direct effects

Our evaluation only addresses nesting loggerhead and Kemp's ridley sea turtles. Consultation with the NMFS will address the effects to sea turtles in the marine environment. Potential sources of impacts to nesting loggerhead and Kemp's ridley sea turtles from existing and proposed oil and gas activities are habitat loss and fragmentation, disturbance from aircraft and boat vessel traffic, effects from trash and debris, and OCS-related air emissions.

Sea turtles have not been observed feeding during the nesting process; thus, ingestion of trash and debris associated with oil and gas activities should not impact nesting sea turtles. Loggerhead nesting and hatchling emergence occur at night; therefore, disturbance from aircraft and/or boat vessel traffic should be minimal because traffic activity then is minimal to negligible. Since Kemp's ridley nesting and hatching emergence occur during daylight hours, it is not known how this affects nesting Kemp's ridley sea turtles; however, minimal disturbance could be expected. Air emissions are not expected to be a concern for nesting adult sea turtles because of the amount of time spent on the beach; however, impacts to developing embryos and hatchlings in the nest are unknown. Offshore pipeline installation is not expected to impact nesting sea turtles; however, onshore beach installation could have an impact if onshore pipeline installation on sea turtle nesting beaches is conducted during the nesting season from May 1 through October 31. New pipelines that go ashore require an environmental analysis before approval. Where pipeline landfall occurs, the permitting process encourages the use of directional boring to greatly reduce and potentially eliminate impacts to barrier beaches.

Indirect effects

Oil Spills

Oil spills impacting the nesting beaches of Kemp's ridley and loggerhead sea turtles are of concern and could have a significant impact, depending upon the geographic location of the spill; hydrocarbon type, dosage, and weathering; impact area; oceanographic and meteorological conditions; season; and life stages of animals exposed to the hydrocarbons (National Research Council [NRC] 1985). Sea turtles are vulnerable to the amount of weathering the oil has undergone, the height of deposition on the beach, and the stage of nesting (Fritts and McGehee 1982). Experiments on the physiologic and clinicopathologic effects of hydrocarbons have shown that major body systems of sea turtles are adversely affected by short exposure to weathered oil. Sea turtles accidentally exposed to oil or tarballs may suffer inflammatory dermatitis, ventilator disturbance, salt gland dysfunction or failure, red blood cell disturbances, immune responses, and digestive disorders or blockages (Vargo et al. 1986; Lutcavage et al. 1995). Although disturbances may be temporary, long-term effects remain unknown, and chronically ingested oil may accumulate in organs. Exposure to hydrocarbons may be fatal, particularly to juvenile and hatchling sea turtles.

Direct contact with oil may also harm developing turtle embryos. Effects of petroleum on the development and survival of marine turtle embryos are variable. Impacts will be different if the oil impacts the beach before nesting, during nest preparation, or during incubation and migration of hatchlings to the sea. Study results indicate that oil remaining on the beach approximately 1 year after a spill did not cause significant mortality in sea turtle eggs; however, fresh crude oil deposited on sand above a nest can cause extensive mortality to incubating sea turtle eggs (Fritts and McGehee 1982). Fritz and McGehee (1982) noted that sea turtle eggs were damaged by contact with weathered oil released from the Ixtoc spill in 1979, which oiled the Rancho Nuevo beaches. In 1994, when beaches on St. Vincent NWR in the Florida panhandle became fouled with tar, female sea turtles crawled through the tar to lay nests, transferring the tar to the nest. However, no tar was found on the eggs in the nest. It has also been proposed that olfactory clues are imprinted on the hatchlings and guide them back to their natal beach for nesting when they reach maturity. Oil on the beach could interfere with these chemical guides (Lutz et al. 1985; Ogren 1990; Possardt 1990).

Oil collecting at beaches through which nesting adults or retreating hatchlings must pass can also affect the survivability of turtles in several ways. Damage can occur by toxic ingestion, with blockage of the digestive tract or internal and external inflammatory responses including infection and poisoning. Most impacts are believed to be sublethal, but little is known about the impacts of chronically ingested oil accumulating in organs. Studies have also indicated that sea turtles do not seem to avoid oil slicks and may ingest or become fouled with the oil. There is concern that long-term chronic impacts will affect the survivability of turtles, both young and old. More definitive information is needed to assess the impacts of oiling on sea turtle nesting beaches.

The OSRA modeling results (10- and 30-day probabilities) indicate that a large spill (>1,000 barrels) in federal offshore waters, would have a 3-5 percent and 9-16 percent probability (from a spill originating in the CPA) and a 5-8 percent and 8-14 percent (from a spill originating in the WPA) of impacting Texas state waters. State waters in Louisiana are divided into east and west Louisiana. West Louisiana has a 10-18 percent and 14-25 percent probability (from a spill originating in the CPA) and a <0.5 and a 1 percent probability (from a spill originating in the WPA), while east Louisiana has a 2-4 percent and 3-5 percent probability (from a spill originating in the CPA) and <0.5 percent and <0.5 percent probability (from a spill originating in the WPA) from reaching state waters. The OSRA model projected a spill risk of <0.5-2 percent for impacting state waters eastward of Louisiana as a result of a spill in either the WPA or CPA. The OSRA modeling results also indicated that a similar spill (i.e., >1,000 barrels), if it were to occur in EPA federal offshore waters, would have <0.5 percent probability of impacting Texas state waters. East and west Louisiana both have a <0.5-1 percent probability of impacts to state waters from such a spill. Waters eastward of Louisiana have a spill risk of <0.5 percent.

The OSRA modeling also produced probabilities of a large spill (>1,000 barrels) contacting coastal counties/parishes. Should a spill occur near the coast within the CPA, a total of 15 counties/parishes (extending from St. Bernard Parish, Louisiana to Kennedy County, Texas) are predicted to be impacted with a <0.5 and <0.5-5 percent probability. Should a spill occur near the coast in the WPA, a total of 10 counties/parishes (extending from Cameron Parish, Louisiana to Kennedy County, Texas) are predicted to be impacted with a >0.5 and <0.5-3 percent probability; while only one parish (Plaquemines) has a spill risk larger than 0.5 percent for a spill in the EPA.

The BOEM/BSEE, USEPA, and USCG have regulations, requirements, and recommendations that should prevent or reduce the likelihood of a spill occurring and prevent or reduce impacts to sea turtles if a spill occurs. This, and the weathering of oil in the environment, should significantly minimize potential impacts on sea turtles and their nesting habitat if a spill occurs.

Should a spill contact a barrier beach (including loggerhead sea turtle critical habitat), oiling is expected to be light, and sand removal during cleanup activities minimal. No significant impacts to the morphology of barrier beaches and associated dunes are expected to occur as a result of a proposed action. Because loggerhead and Kemp's ridley sea turtles nest on high-energy beaches, it is assumed that complete recovery of the coastal beach ecosystem should occur within 1 to 3 years after the spill. Recovery time could vary depending on the severity of the spill, time of year, and cleanup methods used.

Spill Response Activities

Spill-response activities could adversely affect sea turtles and sea turtle habitats, causing displacement from suitable habitat to less suitable areas. Impacting factors include artificial lighting from night operations, booms, machine and human activity, equipment on beaches and in intertidal areas, sand removal and cleaning, and changed beach landscape and composition. Some of the resulting impacts from cleanup could be interrupted or deterred nesting behavior, crushed nests, entanglement in booms, and increased mortality of hatchlings due to predation during the increased time required to reach the water (Newell 1995; Lutcavage et al. 1997). Untended booms could wash ashore and become a barrier to sea turtle adults and hatchlings (U.S. Department of Commerce [USDOC] NOAA 2011). During the response activities individual turtles covered with oil have been captured, cleaned, rehabilitated and released (USDOC NMFS 2013).

Female sea turtles seasonally emerge during the warmer summer months to nest on beaches. Thousands of sea turtles nest along the Gulf coast and turtles could build nests on oiled beaches. Hatchlings, with a naturally high mortality rate, could traverse the beach through oiled sand and swim through oiled water to reach preferred habitats of sargassam floats. Response efforts could include mass movements of eggs from hundreds of nests or thousands of hatchlings from Gulf coast beaches to the east coast of Florida or to the open ocean to prevent hatchlings entering oiled waters (Jernelöv and Lindén 1981). Due to poorly understood mechanisms that guide female sea turtles back to the beaches where they hatched, it is uncertain if relocated hatchlings would eventually return to the Gulf coast to nest. Therefore, shoreline oiling and response efforts may affect future population levels and reproduction.

As mandated by the Oil Pollution Act of 1990 (OPA 90), required spill contingency plans include special notices to minimize adverse effects from vehicular traffic during cleanup activities and to maximize protection efforts to prevent contact of these areas with spilled oil. Sea turtle nesting areas would also be expected to receive special cleanup considerations under these regulations.

Effects summary

Oil spills and spill response activities reasonably expected to result from BOEM/BSEE activities have the potential to impact small to large numbers of sea turtles in the GOM, depending on the magnitude and frequency of accidents, the ability to respond to accidents, the location and date of accidents, and various meteorological and hydrological factors. During their lifetimes, populations of sea turtles in the northern GOM may be exposed to residuals of oils spilled as a result of BOEM and BSEE activities. The greatest concern to nesting sea turtles is the threat of an oil spill reaching nesting habitat during the nesting season.

The probabilities, developed by BOEM/BSEE, of an oil spill occurring and contacting habitat where nesting loggerhead or Kemp's ridley sea turtles nest are low. Furthermore, contact with habitat does not necessarily mean contact with individual organisms, therefore, the likelihood of adverse impacts are further reduced. They also overestimate contact probability because they do not account for naturally occurring events such as weathering and activities included in the proposed action (e.g., clean up, and containment). Although the reduction in those probabilities could not be quantified, it is the Service's belief that those reductions make the likelihood of contact extremely low, but not zero.

As discussed earlier, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to sea turtles associated with a spill of this magnitude are not addressed in this BO.

STATUS OF THE SPECIES/CRITICAL HABITAT

Cape Sable seaside sparrow (*Ammodramus maritimus mirabilis*)

Species/habitat description

The Cape Sable seaside sparrow (CSSS) is one of eight extant subspecies of seaside sparrow in North America. Its distribution is limited to the short-hydroperiod wetlands, or marl prairies, located at the southern end of the greater Everglades ecosystem, on the southern tip of mainland Florida. Unlike most other subspecies of seaside sparrow, which occupy primarily brackish tidal systems (Post and Greenlaw 1994), this sparrow currently occurs primarily in the short hydroperiod wet prairies, also referred to as marl prairies. The sparrow is generally sedentary, secretive, and non-migratory, although sparrows are known to migrate between subpopulations (Lockwood et al. 2008; Virzi et al. 2009).

Life history

Breeding and Nesting

CSSS generally begin nesting in early March (Lockwood et al. 2001), but may begin territorial behavior, courtship, and nest-building in late February (Werner and Woolfenden 1983; Lockwood et al. 1997). This timing coincides with the dry season, and most areas within the marl prairies are either dry or only shallowly inundated at the beginning of the breeding season. During the dry portion of the breeding season (March to May), sparrows build nests above the ground, but relatively low in the vegetation (6.7 to 7.1 inches) (Werner 1975; Lockwood et al. 2001). During the wet portion of the sparrow breeding season (June to August), sparrows build their nests higher in the vegetation than during dry periods, an average of 8.3 inches above the ground surface (Lockwood et al. 2001). Wet-season nests probably occur in taller vegetation than during the dry season because even at the nest height, there must be sufficient height and density of vegetation remaining above the nest to cover and conceal nests.

CSSS lay three to four eggs per clutch (Werner 1978, Pimm et al. 2002) with a hatching rate ranging between 0.66 and 1.00 (Boulton et al. 2009). The sparrow nesting cycle, from nest construction to independence of young, lasts about 30 to 50 days (Werner 1975, Lockwood et al. 2001), and sparrows may renest following both successful and failed nesting attempts (Werner 1975, Post and Greenlaw 1994, Lockwood et al. 2001). Both parents rear and feed the young birds and may do so for an additional 10 to 20 days after the young fledge (Woolfenden 1956, Trost 1968). Sparrows are incapable of flight until they are about 17 days old; when approached flightless fledglings will freeze on a perch until the threat is less than approximately 3 feet away, and then run along the ground (Werner 1975, Lockwood et al. 1997).

Because of the potential for a long breeding season in southern Florida, sparrows may regularly nest several times within a year, and may be capable of successfully fledging two to four clutches, though few sparrows probably reach this level of success (Lockwood et al. 2001). Second and third nesting attempts may occur during the early portion of the wet season, and nests later in the season usually occur over water.

Nest success rates vary among years, and range from 12 to 60 percent, depending upon time within the breeding season (Lockwood et al. 2001, Baiser et al. 2008, Boulton et al. 2009). Substantially higher nest success rates occur within the early portion of the breeding season (prior to June 1) followed by a decline in success as the breeding season progresses to a low of about 20 percent after June 1. Nest predation is the primary documented cause of nest failure (Lockwood et al. 2001, Pimm et al. 2002, Baiser et al. 2008, Boulton et al. 2009, Virzi et al. 2009), accounting for more than 75 percent of all nest failures (Lockwood et al. 1997, Baiser et al. 2008). A complete array of nest predators has not been determined, however, raccoons, rice rats, and snakes, including exotic pythons may be the predominant predators (Lockwood et al. 1997, Post and Greenlaw 2000, Dean and Morrison 2001).

Outside of the breeding season, sparrows generally remain sedentary in the general vicinity of their breeding territories, but expand the area that they use compared to the breeding season territory (Dean and Morrison 2001). Average non-breeding season home range size was approximately 42 acres in size, and ranged from 14.1 to 137.1 acres (Dean and Morrison 2001). Some individuals make exploratory movements away from the area of their territories, and may occasionally relocate their territories and home ranges before resuming a sedentary movement pattern (Dean and Morrison 2001).

Sparrow subpopulations require large patches of contiguous open habitat (about 4,000 acres or larger). The minimum area required to support a population has not been specifically determined, but the smallest area that has remained occupied by sparrows for an extended period is about 4,000 acres. Individuals are area-sensitive, and generally avoid the edges where other habitat types meet the marl prairies. They will only occupy small patches (less than 100 acres) of marl prairie vegetation when they occur within large, expansive areas and are not close to forested boundaries (Dean and Morrison 2001). Large expanses of deep water or wooded habitat may act as barriers to long-range movements (Dean and Morrison 2001). Once sparrows establish a breeding territory, they exhibit high site fidelity, and each individual sparrow may only occupy a small area for the majority of its life (Werner 1975).

CSSS are generally short-lived, with an average individual annual survival rate of 66 percent (Lockwood et al. 2001). The average lifespan is probably 2 to 3 years. Consequently, a sparrow population requires favorable breeding conditions in most years to be self-sustaining, and cannot persist under poor conditions for extended periods (Lockwood et al. 1997, 2001, Pimm et al. 2002).

Feeding Behavior

While detailed information about the diet of CSSS is not known, invertebrates comprise the majority of their diet, though sparrows may also consume seeds when they are available (Werner 1975, Post and Greenlaw 1994). Howell (1932) identified the contents of 15 sparrow stomachs and primarily found remains of insects and spiders, as well as amphipods, mollusks, and plant matter. Primary prey items that are fed to nestlings during the breeding season include grasshoppers, moths and butterflies, dragonflies, and other common large insects (Post and Greenlaw 1994, Stevenson and Anderson 1994, Lockwood et al.1997, Pimm et al. 2002). Adult sparrows probably consume the same species during the nesting season.

Habitat Requirements

Sparrows inhabiting the action area occur mostly within the short-hydroperiod freshwater marl prairies of the southern Everglades that flank the deeper sloughs. The most commonly associated vegetation species in occupied freshwater habitat is muhly grass (Werner 1975, Kushlan and Bass 1983, Werner and Woolfenden 1983, Post and Greenlaw 1994, Stevenson and Anderson 1994). However, a variety of vegetation species occur within the freshwater marl prairies occupied by sparrows, including habitat where muhly grass is absent (Ross et al. 2006). Other dominant species that occur in these prairies include sawgrass, South Florida bluestem, black-topped sedge, and beak rushes (Werner and Woolfenden 1983, Ross et al. 2006).

Sparrows occupy these marl prairie communities year-round, and the vegetation must support all sparrow life stages. During the dry season when the habitat is typically dry, usually coinciding with the late winter and early spring (December to May), sparrows traverse the ground surface beneath the grasses, and only occasionally perch within the vegetation. During the wet season (June to November), the ground surface is inundated, with peak water depths occasionally exceeding two feet (Nott et al. 1998). During these periods, sparrows travel within the grasses, perching low in the clumps, hopping among the bases of dense grass clumps, and walking over matted grass litter. During the wet season sparrows fly more frequently, and regularly perch low in the vegetation, but generally remain inconspicuous (Dean and Morrison 2001).

Small tree islands and individual trees and shrubs occur throughout the areas occupied by the sparrows, but at a very low density. Sparrows do not appear to require woody vegetation during any aspect of their normal behavior, and generally avoid areas where shrubs and trees are either dense or evenly distributed.

Population dynamics

The first comprehensive, range-wide sparrow population survey was conducted in 1981, but was not repeated until 1992. Since that time, surveys have been conducted annually including twice in 1999 and 2000 (Pimm et al. 2002). The number of survey locations has changed through time, from a high of over 850 sites in 1992 to a low of 250 sites in 1995 (Cassey et al. 2007). Over this time period, there have been substantial demographic changes in most of the six subpopulations.

The 1981 sparrow survey provided a baseline on the distribution and abundance of sparrows at that time, and the 1992 survey results were similar, though there is no information available about how the populations may have changed during the intervening 12 years. In 1981, there were an estimated 6,656 sparrows distributed across six subpopulations, with the majority (85 percent) of the sparrows occurring within subpopulations A (40 percent), B (35 percent), and E (10 percent). By comparison, the last complete CSSS population survey for all the subpopulations (2014) resulted in an estimate of 2,720 sparrows, with the majority of birds occurring within subpopulation B (69 percent) and subpopulation E (25 percent).

The overall sparrow population has declined since 1992, and there has been no evidence of significant improvements. In addition to the decline in overall numbers, the distribution has declined. The sparrow subpopulations that have declined have also contracted toward the center of the remaining habitat patches (Cassey et al. 2007).

Status and distribution

The CSSS was listed as an endangered species on March 11, 1967, pursuant to the Endangered Species Preservation Act of 1966 (32 FR 4001). That protection was continued under the Endangered Species Conservation Act of 1969 and the Act of 1973, as amended in 1998 (87 Stat. 884; 16 U.S.C. 1531 et seq. The CSSS was listed because of its limited distribution and threats to its habitat posed by large-scale conversion of land in South Florida to agricultural uses.

Critical habitat for the CSSS was initially designated on August 11, 1977 (42 FR 42840). The critical habitat designation was revised on November 6, 2007 (50 FR 62736) and the revised habitat occurs within Miami-Dade County, Florida.

Analysis of the species/critical habitat likely to be affected

The CSSS and its critical habitat may be affected by the proposed action. The effects of the proposed action on the CSSS will be considered further in the remaining sections of this BO. Potential sources of impacts to this species from existing and proposed oil and gas activities are loss of nesting and critical habitat, disturbance of nests, and trash and debris.

ENVIRONMENTAL BASELINE

Status of the species within the action area

The CSSS is one of eight extant subspecies of seaside sparrow in North America. Its distribution is limited to the short-hydroperiod wetlands at the bottom of the greater Everglades system, on the southern tip of mainland Florida. The great majority of these sparrows occur within Everglades National Park (ENP), and only a small number are found on the adjacent state-owned Southern Glades Wildlife and Environmental Area. It was one of the first group of species listed under the Act. Critical habitat was first designated for this species in 1977, and revised critical habitat designation was published in November 2007. Unlike most other subspecies of seaside sparrow, which occupy primarily brackish tidal systems (Post and Greenlaw 1994), CSSS currently occurs primarily in the short-hydroperiod wet prairies, also referred to as marl prairies, though it may still occupy brackish marshes in some areas.

Factors affecting species environment within the action area

Hydrology

The Central and Southern Florida Project (C&SF) is a system-wide network of canals and water-control structures. The U.S Army Corps of Engineers (Corps) and South Florida Water Management District (District) operate the C&SF Project to achieve a variety of local and regional objectives including flood protection, water supply, and environmental benefits. Operations of the C&SF Project affect the hydrologic conditions of nearly all the wetland systems within southern Florida to some degree, including the habitat supporting the CSSS. In general, the closer wetland habitat is located to water control infrastructure, the greater the potential effect may be. The Service's 2002 BO prescribed the Interim Operational Plan (IOP) as a second reasonable and prudent alternative (RPA) with qualifications which included a hydrologic management regime to protect sparrow breeding by reducing water deliveries in western marl prairies that are too wet and increasing water deliveries to the eastern marl prairies that have been historically over drained prior to the expansion of ENP.

Under the IOP, hydrologic management provided reduced flows during the breeding season to sparrow habitat located in the western marl prairies. Construction and operation of several detention areas adjacent to sparrow habitat in the eastern subpopulations increased hydroperiods in some over-drained habitats such as subpopulation C. Many other hydrologic operations throughout the C&SF system that routinely occur have resulted in changes to hydrologic conditions in and adjacent to sparrow habitat. Pre-storm and post-storm operations, testing of hydrologic management operations, and other similar activities conducted by the Corps and District have also affected hydrologic conditions within sparrow habitat, mainly through alteration of the natural timing of wetting and drying events.

Fire

Fire is a natural or human-related factor that affects marl prairies occupied by the sparrow and most sparrow habitats have burned at some point during the past 30 to 40 years. The ENP, Big Cypress National Preserve, and FWC have all conducted prescribed burns within sparrow habitat on lands within their respective jurisdictions. In the short-term, fire typically renders sparrow habitat unsuitable for occupancy because it removes the vegetation that sparrows rely upon for cover and refugia especially during the breeding season. Following fire, vegetation normally begins to regenerate rapidly and reaches pre-burn density and species composition about 2 years later. Sparrows do not regularly occupy burned areas for 2 to 3 years after fire (La Puma et al. 2007). ENP has conducted prescribed fire in former sparrow habitat within the western marl prairies to facilitate habitat restoration. Prescribed burns have also been conducted along the eastern ENP boundary to reduce the likelihood of human-ignited fires spreading into sparrow habitat. Because fires reduce habitat suitability for up to 3 years, prescribed fires, human-induced fires and wildfires can all have adverse short-term effects on sparrow populations, but also may be necessary in the long-term for the maintenance of habitat that outweigh the adverse short-term effects (Taylor 1983; Pimm et al. 2002; Lockwood et al. 2003; LaPuma et al. 2007).

Changes in vegetation composition can result from changes in hydrologic conditions, changes in fire frequency, and change in management actions. Many areas of sparrow habitat have experienced vegetation change since monitoring was initiated. Over drying that results from maintaining artificially low water levels within areas of sparrow habitat, such as those that occur along the eastern boundary of the ENP, results in woody vegetation encroachment, which reduces the suitability of the habitat for sparrow occupancy. Extended hydroperiods and deep water depths may occur from managed water releases in combination with wet-season rainfall which can lead to vegetation changes from marl prairie species to marsh species, resulting in reduced habitat suitability.

Invasive and Exotic Species

Invasive and exotic species may also affect sparrows. Invasive plant species such as punk tree or paperbark tea tree, Australian pine, Brazilian pepper, and other woody species can become established in sparrow habitat and reduce habitat suitability. While limited information is available on the effects of invasive exotic plants and animals on sparrows, species like the Burmese python have become established in sparrow habitat and may depredate sparrows. There is also concern about competition with recently introduced non-native predators such as the Argentine black and white tegu, a known predacious and egg scavenging lizard.

Management of invasive woody plants has been conducted by the ENP, FWC, and District in and adjacent to sparrow habitat to reduce impacts of these species on sparrow habitat suitability. Herbicide treatment of large stands of exotic trees has reduced the spread of these species and has improved sparrow habitat in some areas. These invasive plant species regenerate rapidly requiring continued maintenance controls. Efforts to remove invasive exotic animals like the Burmese python have also been initiated, but to date these efforts have largely been opportunistic.

Water Quality

The Everglades was historically an oligotrophic system, lacking nutrients such as phosphorus, but having high levels of dissolved oxygen. Major portions have become rich in nutrients that promote excessive plant growth and deplete dissolved oxygen primarily due to anthropogenic sources of phosphorus and nitrogen (cultural eutrophication). Degradation of water quality, particularly runoff of phosphorus from agricultural and urban sources, is a concern because it can cause encroachment of cattail and other undesirable invasive and exotic species.

EFFECTS OF THE ACTION

Direct effects

Potential sources of direct impact to the CSSS from the proposed oil and gas activities are habitat loss and fragmentation, disturbance from aircraft and boat vessel traffic, effects from trash and debris, and OCS-related air emissions. Pipeline landfalls, terminals, and other onshore OCS-related infrastructure can destroy of fragment otherwise suitable CSSS habitat. These activities should only have a minimal effect on CSSS and their critical habitat because the range of this species encompasses an area where new construction is not anticipated. In addition, new pipelines that go ashore require an environmental analysis before approval. Where pipeline landfall occurs, the permitting process encourages the development of measures to minimize and potentially eliminate impacts to federally listed species.

Low-altitude aircraft overflights related to OCS oil and gas operations could affect the CSSS. CSSS may be susceptible to disturbance by low-altitude aircraft during nesting, foraging and resting periods. Those birds may leave and cease using their preferred nesting and feeding areas and possibly seek less desirable ones, resulting in decreased nest success, increased energy expenditure via flight and alertness, and reduced energy intake via lower feeding rates. The Service, FAA, NPS, and U.S. Bureau of Land Management (BLM) have an Interagency Agreement to reduce low-level flights over natural resource areas. The recommended minimum flight altitude is 2,000 feet above ground level. This limitation is included on aeronautical maps. The FAA (FAA Advisory Circular 91-36C) and corporate helicopter policy also states that helicopters must maintain a minimum altitude of 700 feet while in transit offshore and 500 feet while working between platforms. When flying over land, the specified minimum altitude is 1,000 feet over unpopulated areas or across coastlines and 2,000 feet over populated areas and biologically sensitive areas such as wildlife refuges and national parks. Within the EPA, BOEM/BSEE anticipates 0-2 helicopter trips annually. Service vessels, in support of OCS-related activities, are expected to use selected nearshore waters and existing coastal navigation waterways. Those activities are not anticipated to significantly increase the amount of existing routine vessel traffic within these waterways. Impacts to the CSSS from helicopter and vessel traffic should, therefore, be minimal.

There are numerous existing laws, regulations, and enforcement guidelines that prohibit and discourage the disposal of solid debris in Gulf waters that can impact listed species and their critical habitats. For example, BOEM/BSEE prohibits the disposal of equipment, containers, and other materials into offshore waters by lessees (30 CFR 250.300). Also, BSEE NTL No. 2015-G03 requires annual awareness training and the posting of placards to minimize the unintentional loss of debris from industry structures or vessels. BSEE inspectors routinely conduct site visits and issue citations for noncompliance. In addition, MARPOL, Annex V. Public Law 100-220 (101 Statute 1458), which prohibits the disposal of any plastics, garbage, and other solid wastes at sea or in coastal waters, went into effect January 1, 1989, and is enforced by the USCG. The MDRPR was enacted in December 2006. The purposes of the MDRPR are to help identify, determine sources of, assess, reduce and prevent marine debris and its adverse impacts on the marine environment and navigation safety; to reactivate the Interagency Marine Debris Coordinating Committee; and to develop a Federal marine debris information clearinghouse. The MDRPR established, within the NOAA, a Marine Debris Prevention and Removal Program to reduce and prevent the occurrence and adverse impacts of marine debris on the marine environment and navigation safety. Greatly improved handling of waste and trash by industry, along with the annual awareness training required by the marine debris mitigation, is decreasing OCS-related debris in the ocean and impacts to the CSSS are, therefore, expected to be negligible.

BOEM/BSEE anticipates minimal effects to air quality associated with OCS oil and gas emissions due to prevailing atmospheric conditions, emission heights and rates, and pollutant concentrations. Because emissions from OCS-related activities are not likely to impact ambient air quality, effects to the CSSS from decreased air quality are expected to be negligible.

Indirect effects

While oil spills represent the greatest potential impact to coastal and marine bird populations, the CSSS preferred habitat is open prairie that occurs between marsh and scrub/forest habitat. Because CSSS are an interior species and are not know to nest, forage, or rest within coastal habitats, contact with oil is unlikely. In addition, there is less than a 0.5 percent probability that an oil spill > 1,000 barrels would occur and contact CSSS or their habitat (including critical habitat) within 10 days in the EPA. Note again that those probabilities do not include clean-up activities and natural weathering of the spill. The BOEM/BSEE, USEPA, and USCG have regulations, requirements, and recommendations that should prevent or reduce the likelihood of a spill occurring and prevent or reduce impacts to CSSS if a spill occurs. This, and the weathering of oil in the environment, should further minimize potential impacts on CSSS if a spill occurs.

Effects summary

Activities occurring as a result of the proposed action may affect the CSSS and its critical habitat; however, no direct loss of habitat is anticipated. It is expected that the majority of the effects from the major-impact producing factors (i.e., habitat loss and fragmentation, aircraft noise and operation, vessel noise and operation, marine debris, and air emissions are sublethal and infrequent within CSSS habitat, causing discountable or insignificant effects. The greatest concern is the threat of an oil spill reaching CSSS nesting habitat during the nesting season. As stated above, however, CSSS habitat is buffered from coastal habitats and the probabilities, developed by BOEM/BSEE, of an oil spill occurring and contacting habitat where nesting CSSS occur are low. Furthermore, contact with habitat does not necessarily mean contact with individual organisms, therefore, the likelihood of adverse impacts are further reduced. They also overestimate contact probability because they do not account for naturally occurring events such as weathering and activities included in the proposed action (e.g., clean up, and containment). Although the reduction in those probabilities could not be quantified, it is the Service's belief that those reductions make the likelihood of contact extremely low, but not zero.

As discussed earlier, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to CSSS associated with a spill of this magnitude are not addressed in this BO.

STATUS OF THE SPECIES/CRITICAL HABITAT

Mississippi sandhill crane (*Gros Canadensis pulla*)

Species/critical habitat description

Mississippi sandhill cranes resemble great blue herons. A major distinguishing characteristic is that cranes are completely gray. When standing erect, cranes are about four feet tall. Male and female cranes are similar in appearance. All cranes have long necks, and adult cranes possess a bald red forehead. The species vocalizations are loud and clattering. Mississippi sandhill cranes are a non-migratory subspecies which have become reproductively isolated from other sandhill cranes. The only known wild population is on and near the Mississippi Sandhill Crane National Wildlife Refuge in Jackson County, Mississippi. The birds present range is restricted to an area defined by the Pascagoula River (east), to about the Jackson County line (west), to about Simmons Bayou (south), to 4 miles north of the town of Vancleave (north).

Mississippi sandhill cranes were listed as rare in the 1968 list of Rare and Endangered Fish and Wildlife of the United States. After being described as a subspecies in 1972, the Mississippi sandhill crane was added to the U.S. List of Endangered Fish and Wildlife on June 4, 1973 38 Fed. Reg. 14678. The first recovery plan was written in 1976 and the latest revision (3^{rd}) completed in 1991. In 1974 the Nature Conservancy purchased 1,709 acres which the Service acquired in 1975 to establish the Mississippi Sandhill Crane National Wildlife Refuge. Additional lands have been acquired such that the current total acreage of the Refuge is 19,273 acres. Reducing the likelihood of extinction will require a self-sustaining population of cranes and suitable habitat. Original estimates suggested the Refuge crane population may require a minimum of about 130 to 170 birds, consisting of about 60 nesting cranes per breeding season, for a continuous period of at least 10 years (Service 1991). Long term self-sustenance and stability will require a genetically viable population, high levels of natural recruitment, and cessation of the captive release program.

Critical habitat for the Mississippi sandhill crane was designated on September 8, 1977 42 Fed. Reg. 39985 (USFWS 1977). Current Service policy requires that the PCEs of critical habitat be defined. PCEs are those physical and biological features of a landscape that a species needs to survive and reproduce. However, the Final Rule that determined critical habitat for the Mississippi sandhill crane occurred prior to establishment of this policy. Nevertheless, we now define the PCEs to be those elements required to support appropriate foraging, roosting, and nesting habitat isolated from human disturbance by a minimum of 100 yards. The areas delineated for critical habitat cover about 26,000 acres of which 19,273 acres are currently protected on the Refuge. Most known breeding, summer feeding, and roosting sites are included in the 26,000 acres of critical habitat. Surveys indicate that approximately 90% of crane breeding sites and approximately 70% of their roosting sites occur in critical habitat. Since 1965, approximately 9% of documented nesting has been located off the Refuge. Scattered winter feeding areas are located both on the refuge (approximately 80%) and on neighboring farmlands outside of designated critical habitat. These sites cover a large area, and sporadic use of these areas by cranes varies with the planted crops. Although included within the critical habitat boundary, not all such areas actually possess all of the constituent elements of critical habitat.

Life history

Mississippi sandhill cranes are long lived. In the wild they do not reach reproductive age until around 4 to 5 years of age (sometimes not until their "teens"), have large nesting territories, and frequently raise only one chick per year (Service 1991).

Savannahs are the optimal habitat of the Mississippi sandhill crane and are inhabited year round. These wet grasslands are predominated by wiregrass, with scattered longleaf pine, slash pine, and cypress trees. Other associated plants include pitcher plants, sundew, clubmoss, and pipeworts. Cranes also utilize wooded depressions (swamps) dominated by cypress, longleaf: and slash pine trees with an understory of swamp cyrilla, buckwheat tree, wax myrtle, and several species of holly (Service 1991).

Cranes roost in shallow water in savannas, edges of wooded depressions or swamps, and ponds. Paired cranes roost near the nest during the breeding season. Mississippi sandhill cranes prefer to nest as far from sources of disturbance as possible. Ideally this is in open area of grasses and sedges adjacent to perennial shallow water. Such an area, surrounded by trees and shrubs, is typically large enough for the cranes to see potential predators and allow flight. Due to the economic growth of coastal Mississippi, construction of miles of access roads, and plantation pine forestry techniques, most of the original ideal nesting habitat has been destroyed.

Crane feeding habitats vary with the seasons. During the spring, summer, and early fall, cranes consume both plant and animal matter equally, including roots, tubers, fruits, insects, earthworms, other invertebrates, and occasionally a few frogs and other small vertebrates. During the cooler months, cranes switch some of their preferred items diet to products of upland agriculture including corn, seeds, and insects found in farms, pastures, and Refuge food plots. Chufa is planted in the spring or summer for the cranes on the Refuge and then cool season grasses and legumes are planted in the fall. As a result of human population growth, some agricultural areas in the vicinity of the Refuge that are now used by cranes for foraging (including some that have been utilized for decades and even generations), are being converted to high density residential or commercial development that is not suitable for cranes.

Population dynamics

Population estimates in 1929, 1949, and 1969 indicated that the crane population has been less than 100 since 1929 with evidence of continuing decline through 1980 (Seal et al. 1992). Since inception of the Refuge in 1975 and formal designation of critical habitat for the Mississippi sandhill crane (Service 1977), the population levels have increased from a low of 30-35 individuals and 5-6 nesting pairs to over 100 birds and over 20 nesting pairs. A survey conducted in 2015 on the Mississippi Sandhill Crane National Wildlife Refuge yielded 133 birds and 24 nesting pairs that produced a total of 35 nests with 11 renests. Six crane chicks fledged including one set of twins.

Supplementation of the population began in 1981 and has continued every year since. Approximately 95% of the current free ranging population is from captive hatched or captive bred birds (S. Hereford, pers. comm.). Recently, the population has been maintained purposely from 100-130 birds (S. Hereford, pers. comm.). Mapping of the habitat requirements of the crane (in the early 1990s) indicated that a population of about 130-150 birds was the maximum capacity of the refuge at that time, even with intensive site management (Seal et al. 1992). Ultimately, the carrying capacity of the Refuge will be limited by the habitat available for nesting territories. The addition of protected, managed, high quality crane habitat, particularly potential nesting areas, to the Refuge is vital for the recovery of the species.

Status and distribution

The Mississippi sandhill crane is a nonmigratory endangered subspecies which has become reproductively isolated from other sandhill cranes and is in danger of extinction. Major reasons for the decline include loss of habitat, human predation, and decreased natural recruitment. Mississippi sandhill cranes were once found all along the Gulf Coast with a total population possibly into the thousands. During the 1950's thousands of acres of the crane's favored savannah habitat were drained and converted to slash pine plantations. Dense understories developed underneath the mature pine trees, and the once open, undisturbed habitat became unsuitable for cranes. The latter part of the 20th century brought a human population explosion to the Mississippi coast, including residential and commercial development and infrastructure utilities to support that growth. Eight paved highways now transect or border the crane's range. These roads have further depleted habitat, caused pollution problems, and provided public access to the cranes, all of which have caused problems for the species.

Analysis of the species/critical habitat likely to be affected

The Mississippi sandhill crane and its critical habitat may be affected by the proposed action. The effects of the proposed action on this species will be considered further in the remaining sections of this biological opinion. Potential sources of impacts to this species from existing and proposed oil and gas activities are loss of nesting and critical habitat, disturbance of nests, and trash and debris.

ENVIRONMENTAL BASELINE

Status of the species within the action area

The range of the Mississippi sandhill crane is limited to the action area, in Jackson County, Mississippi, as described above. In addition, the entire designated critical habitat for the Mississippi sandhill crane occurs within the action area.

Factors affecting species environment within the action area

Previously described development has historically affected the species and its habitat throughout the action area (cranes' range), and continues to do so (with the exception of lands on the refuge). This includes development or other use of lands within the cranes range that are not designated as critical habitat (approximately 83% of their range is not designated as critical habitat).

Numerous enhancement/restoration projects designed to benefit Mississippi sandhill cranes and their habitat have occurred on the refuge. Specifically in 2015, the refuge: (1) conducted mechanical treatment on approximately 224 acres (via bush-hog, mulching machine, gyrotrack, chain saw) to enhance open savanna, (2) bush-hogged 450 acres, (3) conducted twenty-nine prescribed burns totaling 5,739 acres (4) chemically treated approximately 115 acres of cogongrass, and (5) released 10 captive-reared juveniles (of which 80% survived the first year). In addition, the contract trapper ran traps (foot-hold and snare) during January-June and October-December to protect nesting and released cranes. A total of 5,405 trap-nights were conducted removing 25 bobcats, 25 coyotes, 88 raccoons, one Canada goose, and 37 other.

EFFECTS OF THE ACTION

Direct effects

Potential sources of direct impact to the Mississippi sandhill crane from the proposed oil and gas activities are habitat loss and fragmentation, disturbance from aircraft and boat vessel traffic, effects from trash and debris, and OCS-related air emissions. Pipeline landfalls, terminals, and other onshore OCS-related infrastructure can destroy of fragment otherwise suitable Mississippi sandhill crane habitat. These activities should only have a minimal effect on Mississippi sandhill cranes and their critical habitat because the range of this species encompasses an area where new construction is not anticipated. In addition, new pipelines that go ashore require an environmental analysis before approval. Where pipeline landfall occurs, the permitting process encourages the development of measures to minimize and potentially eliminate impacts to federally listed species.

Low-altitude aircraft overflights related to OCS oil and gas operations could affect the Mississippi sandhill crane. Mississippi sandhill cranes may be susceptible to disturbance by low-altitude aircraft during nesting, foraging and resting periods. Those birds may leave and cease using their preferred nesting and feeding areas and possibly seek less desirable ones, resulting in decreased nest success, increased energy expenditure via flight and alertness, and reduced energy intake via lower feeding rates. The Service, FAA, NPS, and BLM have an Interagency Agreement to reduce low-level flights over natural resource areas. The recommended minimum flight altitude is 2,000 feet above ground level. This limitation is included on aeronautical maps. The FAA (FAA Advisory Circular 91-36C) and corporate helicopter policy also states that helicopters must maintain a minimum altitude of 700 feet while in transit offshore and 500 feet while working between platforms. When flying over land, the specified minimum altitude is 1,000 feet over unpopulated areas or across coastlines and 2,000 feet over populated areas and biologically sensitive areas such as wildlife refuges and national parks. Within the CPA, BOEM/BSEE anticipates 594,500-1,112,500 helicopter trips annually. Service vessels, in support of OCS-related activities, are expected to use selected nearshore waters and existing coastal navigation waterways. Those activities are not anticipated to significantly increase the amount of existing routine helicopter and vessel traffic within the CPA. Impacts to the Mississippi sandhill crane from helicopter and vessel traffic should, therefore, be minimal.

There are numerous existing laws, regulations, and enforcement guidelines that prohibit and discourage the disposal of solid debris in Gulf waters that can impact listed species and their critical habitats. For example, BSEE prohibits the disposal of equipment, containers, and other materials into offshore waters by lessees (30 CFR 250.300). Also, BSEE NTL No. 2015-G03 requires annual awareness training and the posting of placards to minimize the unintentional loss of debris from industry structures or vessels. BSEE inspectors routinely conduct site visits and issue citations for noncompliance. In addition, MARPOL, Annex V. Public Law 100-220 (101 Statute 1458), which prohibits the disposal of any plastics, garbage, and other solid wastes at sea or in coastal waters, went into effect January 1, 1989, and is enforced by the USCG. The MDRPR (P .L 109-449) was enacted in December 2006. The purposes of the MDRPR are to help identify, determine sources of, assess, reduce and prevent marine debris and its adverse impacts on the marine environment and navigation safety; to reactivate the Interagency Marine Debris Coordinating Committee; and to develop a Federal marine debris information clearinghouse. The MDRPR established, within the NOAA, a Marine Debris Prevention and Removal Program to reduce and prevent the occurrence and adverse impacts of marine debris on the marine environment and navigation safety. Greatly improved handling of waste and trash by industry, along with the annual awareness training required by the marine debris mitigations, is decreasing OCS-related debris in the ocean and impacts to the Mississippi sandhill crane are, therefore, expected to be negligible.

BOEM/BSEE anticipates minimal effects to air quality associated with OCS oil and gas emissions due to prevailing atmospheric conditions, emission heights and rates, and pollutant concentrations. Because emissions from OCS-related activities are not likely to impact ambient air quality, effects to the Mississippi sandhill crane from decreased air quality are expected to be negligible.

Indirect effects

While oil spills represent the greatest potential impact to coastal and marine bird populations, the Mississippi sandhill cranes preferred habitat is savannahs predominated by wiregrass, with scattered longleaf pine, slash pine, and cypress trees. Because Mississippi sandhill cranes are an interior species and are not know to nest, forage, or rest within coastal habitats, contact with oil is unlikely. In addition, there is a 0.5-1 percent probability that an oil spill > 1,000 barrels originating in the WPA or CPA would occur and contact Mississippi sandhill crane habitat (including critical habitat) within 10 days. There is < 0.5 percent probability that an oil spill >1,000 barrels originating in the EPA would occur and contact Mississippi sandhill cranes and their habitat (including critical habitat). Note again that those probabilities do not include clean-up activities and natural weathering of the spill. The BOEM/BSEE, USEPA, and USCG have regulations, requirements, and recommendations that should prevent or reduce the likelihood of a spill occurring and prevent or reduce impacts to Mississippi sandhill cranes if a spill occurs. This, and the weathering of oil in the environment, should further minimize potential impacts on Mississippi sandhill cranes if a spill occurs.

Effects summary

Activities occurring as a result of the proposed action may affect the Mississippi sandhill crane and its critical habitat; however, no direct loss of habitat is anticipated. It is expected that the majority of the effects from the major-impact producing factors (i.e., habitat loss and fragmentation, aircraft and vessel noise and operation, marine debris, and air emissions) are sublethal and infrequent within Mississippi sandhill crane habitat, causing discountable or insignificant effects. The greatest concern is the threat of an oil spill reaching Mississippi sandhill crane nesting habitat during the nesting season. As stated above, however, Mississippi sandhill crane habitat is buffered from coastal habitats and the probabilities, developed by BOEM/BSEE, of an oil spill occurring and contacting habitat where nesting Mississippi sandhill cranes occur are low. Furthermore, contact with habitat does not necessarily mean contact with individuals, therefore, the likelihood of adverse impacts are further reduced. They also overestimate contact probability because they do not account for naturally occurring events such as weathering and activities included in the proposed action (e.g., clean up, and containment). Although the reduction in those probabilities could not be quantified, it is the Service's belief that those reductions make the likelihood of contact extremely low, but not zero.

As discussed earlier, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to Mississippi sandhill cranes associated with a spill of this magnitude are not addressed in this BO.

STATUS OF THE SPECIES/CRITICAL HABITAT

Piping plover (*Charadrius melodus*)

Species/critical habitat description

The piping plover is a small (7 inches long), pale, sand-colored shorebird with a wingspan of 15 inches (Palmer 1967). On January 10, 1986, the piping plover was listed as endangered in the Great Lakes watershed and threatened elsewhere within its range, including migratory routes outside of the Great Lakes watershed and wintering grounds (Service 1985). Piping plovers were listed principally because of habitat destruction and degradation, predation, and human disturbance. Three separate breeding populations have been identified, each with its own recovery criteria: the northern Great Plains (threatened), the Great Lakes (endangered), and the Atlantic Coast (threatened). The piping plover winters in coastal areas of the U.S. from North Carolina to Texas, along the coast of eastern Mexico, on Caribbean islands from Barbados to Cuba, and in the Bahamas (Haig and Elliott-Smith 2004).

The Service has designated critical habitat for the piping plover on three occasions. Two of these designations protected different breeding populations. Critical habitat for the Great Lakes breeding population was designated May 7, 2001 (Service 2001a), and critical habitat for the northern Great Plains breeding population was designated September 11, 2002 (Service 2002). Piping plovers do not breed along the Gulf coast; therefore, critical habitat for breeding populations does not occur along the Gulf coast and will not be discussed further in this document.

The Service also designated critical habitat for wintering piping plovers on July 10, 2001 (Service 2001b). Wintering piping plovers may include individuals from the Great Lakes and northern Great Plains breeding populations as well as birds that nest along the Atlantic coast. Designated wintering piping plover critical habitat originally included 142 areas (the rule states 137 units; this is in error) encompassing about 1,793 miles of mapped shoreline and 165,211 acres of mapped areas along the coasts of North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Texas. Since the designation of wintering critical habitat, four units in North Carolina have been vacated and remanded back to the Service for reconsideration by Court order (<u>Cape Hatteras Access Preservation Alliance v. U.S. Department of Interior</u>, 344 F. Supp. 2d 108 (D.D.C. 2004)). The four critical habitat units vacated were NC-1, 2, 4, and 5, and all occurred within Cape Hatteras National Seashore. A revised designation for these four units was published on October 21, 2008 (Service 2008a). The Courts also vacated and remanded back to the Service for reconsideration, 19 units (TX- 3,4,7-10, 14-19, 22, 23, 27,28, and 31-33) in Texas (<u>Texas General Land Office v. U.S. Department of Interior</u>, Case No. V-06-CV-00032). On May 19, 2009, the Service published a final rule designating 18 revised critical habitat units in Texas, totaling approximately 139,029 acres (Service 2009a).

For wintering piping plovers, PCEs are those habitat components that support foraging, roosting, and sheltering and the physical features necessary for maintaining the natural processes that support these habitat components. These areas typically include coastal areas that support intertidal beaches and flats and associated dune systems and flats above annual high tide (Service

2001a). Specifically, PCEs of wintering piping plover critical habitat include sand or mud flats (or both) with no or sparse emergent vegetation. Adjacent unvegetated or sparsely vegetated sand, mud, or algal flats above high tide are also important PCEs, especially for roosting piping plovers (Service 2001a). PCEs of the beach/dune ecosystem include surf-cast algae, natural wrack, sparsely vegetated back beach and salterns, spits, and over-wash areas. Over-wash areas are broad, unvegetated zones, with little or no topographic relief, that are formed and maintained by the action of hurricanes, storm surge, or other extreme wave action. The units designated as critical habitat are those areas that had consistent use by piping plovers at the time of designation and that best meet the biological needs of the species.

Activities that affect PCEs include those that directly or indirectly alter, modify, or destroy the processes that are associated with the formation and movement of barrier islands, inlets, and other coastal landforms. Those processes include erosion, accretion, succession, and sea-level change. The integrity of the habitat components also depends upon daily tidal events and regular sediment transport processes, as well as episodic, high-magnitude storm events (Service 2001b).

Life history

Piping plovers live an average of five years, although studies have documented birds as old as 11 (Wilcox 1959) and 15 years. Breeding activity begins in mid-March when birds begin returning to their nesting areas (Coutu et al. 1990; Cross 1990; Goldin et al. 1990; MacIvor 1990; Hake 1993). The female can lay up to four eggs, which hatch approximately 25 days later. Chicks fledge in three to four weeks after hatching. Plovers are known to begin breeding as early as one year of age (MacIvor 1990; Haig 1992); however, the percentage of birds that breed in their first adult year is unknown. Piping plovers generally fledge only a single brood per season, but may re-nest several times if previous nests are lost.

Cryptic coloration is a primary defense mechanism for piping plovers. Nests, adults, and chicks all blend in with their typical beach surroundings. Piping plovers on wintering and migration grounds respond to intruders (pedestrian, avian, and mammalian) usually by squatting, running, and flushing (flying).

Migration

Plovers depart their breeding grounds for their wintering grounds from July through late August, but southward migration extends through November. Piping plovers spend up to 10 months of their life cycle on their migration and winter grounds, generally July 15 through as late as May 15. Piping plovers migrate through and winter in coastal areas of the U.S. from North Carolina to Texas and in portions of Mexico and the Caribbean. The pattern of both fall and spring counts at many Atlantic Coast sites demonstrates that many piping plovers make intermediate stopovers lasting from a few days up to one month during their migrations (Noel et al. 2005; Stucker and Cuthbert 2006). Use of inland stopovers during migration is also documented (Pompei and Cuthbert 2004). The source breeding population of a given wintering individual cannot be determined in the field unless it has been banded or otherwise marked. Information from observation of color-banded piping plovers indicates that the winter ranges of the breeding populations overlap to a significant degree.

Foraging (nonbreeding portion of annual cycle)

Behavioral observation of piping plovers on the wintering grounds suggests that they spend the majority of their time foraging (Nicholls and Baldassarre 1990a; Drake 1999a, 1999b). Feeding activities may occur during all hours of the day and night (Staine and Burger 1994; Zonick 1997), and at all stages in the tidal cycle (Goldin 1993; Hoopes 1993). Wintering plovers primarily feed on invertebrates such as polycheate marine worms, various crustaceans, fly larvae, beetles, and occasionally bivalve mollusks (Bent 1929; Nicholls 1989; Zonick and Ryan 1995). They peck these invertebrates on top of the soil or just beneath the surface. Plovers forage on moist substrate features such as intertidal portions of ocean beaches, over-wash areas, mudflats, sand flats, algal flats, shoals, wrack lines, sparse vegetation, shorelines of coastal ponds, lagoons, ephemeral pools and adjacent to salt marshes, as well as bay-side islands and beaches with abundant prey items (Gibbs 1986; Zivojnovich 1987; Nicholls 1989; Nicholls and Baldassarre 1990a, 1990b; Coutu et al. 1990; Hoopes et al. 1992; Loegering 1992; Goldin 1993; Elias-Gerken 1994; Wilkinson and Spinks 1994; Zonick 1997; Service 2001a; Cohen et al. 2006).

Roosting

Piping plovers roost in unvegetated or sparsely vegetated areas, which may have debris, detritus, or micro-topographic relief offering refuge to plovers from high winds and cold weather. Several studies identified that wrack (organic material including seaweed, seashells, driftwood, and other materials deposited on beaches by tidal action) is also an important component of roosting habitat for nonbreeding piping plovers (Lott et al. 2009; Maddock et al. 2009; Smith 2007; Drake 1999a, 1999b). Plovers will also roost on intertidal habitat, backshore coastline (defined as a zone of dry sand, shell, cobble and beach debris from mean high water line up to the toe of the dune), over-wash and ephemeral pools (Maddock et al. 2009; Smith 2007), as well as sea grass debris (bay-shore wrack) (Drake 1999b).

Population dynamics

Populations on all three portions of the breeding range have increased since listing. The Atlantic Coast breeding population has increased an estimated 234 percent, from approximately 790 pairs in 1986 to 1,762 in 2011 (Service 2009b; Service 2012a). Likewise, the Great Lakes breeding population has increased from an estimated 12 pairs in 1984 to 58 nesting pairs in 2012, most of which nested in Michigan (Service 2009b; Service 2012b). The northern Great Plains breeding population is the largest with an estimated 2,953 individuals in 1991 (1,981 in the U.S. excluding Canada) and an estimated 4,662 individuals in 2006 (2,959 in the U.S. excluding Canada) (Ferland and Haig 2002, Elliott-Smith et al. 2009).

Various population viability analyses conducted for piping plovers indicate that small declines in adult and juvenile survival rates can cause substantial increases in extinction risk (Ryan et al. 1993; Melvin and Gibbs 1996; Plissner and Haig 2000; Wemmer et al. 2001; Larson et al. 2002; Amirault et al. 2005; Calvert et al. 2006; Brault 2007). This suggests that maximizing productivity on the breeding grounds does not ensure population increases. Efforts to partition survival within the annual cycle are beginning to receive more attention, but current information remains limited. Thus, survival during migration and on the wintering grounds remains an important concern for the stability of piping plover breeding populations.

Status and distribution

Nonbreeding (migrating and wintering) Range

Piping plovers spend up to 10 months of their life cycle on their migration and wintering grounds, generally July 15 through as late as May 15. Piping plover migration routes and habitats overlap breeding and wintering habitats, and, unless banded, migrants passing through a site usually are indistinguishable from breeding or wintering piping plovers. Review of published records of piping plover sightings throughout North America by Pompei and Cuthbert (2004) found more than 3,400 fall and spring stopover records at 1,196 sites. Published reports indicated that piping plovers do not concentrate in large numbers at inland sites and that they seem to stop opportunistically. In most cases, reports of birds at inland sites were single individuals. In general, distance between stopover locations and duration of stopovers throughout the coastal migration range remains poorly understood.

Piping plovers migrate through and winter in coastal areas of the U.S. from North Carolina to Texas and in portions of Mexico and the Caribbean. Five range-wide, mid-winter (late January to early February) International Piping Plover Census population surveys, conducted at five-year intervals starting in 1991, are summarized in Table 2 (Ferland and Haig 2002, Haig et al. 2005, Elliott-Smith et al. 2009, 2015). Total numbers have fluctuated over time, with some areas experiencing increases and others decreases. About 89 percent of birds that are known to winter in the U.S. do so along the Gulf Coast (Texas to Florida), while eight percent winter along the Atlantic Coast (North Carolina to Florida). Results from the 2011 International Piping Plover Census indicate that the Bahamas are also an important wintering area for piping plovers.

Table 2. Results of the 1991, 1996, 2001, 2006, and 2011 International Piping Plover Censuses of wintering birds.

Location	1991	1996	2001	2006	2011
Virginia	NS*	NS*	NS*	1	1
North Carolina	20	50	87	84	43
South Carolina	51	78	78	100	86
Georgia	37	124	111	212	63
Florida	551	375	416	454	306
Alabama	12	31	30	29	38
Mississippi	59	27	18	78	88
Louisiana	750	398	511	226	86[†]
Texas	1,904	1,333	1,042	2,090	2,145
Puerto Rico	0	0	6	NS*	2
U.S. Total	3,384	2,416	2,299	3,355	2,858
Mexico	27	16	NS*	76	30
Bahamas	29	17	35	417	1,066
Cuba	11	66	55	89	19
Other Caribbean Islands	0	0	0	28	NS*
GRAND TOTAL	3,451	2,515	2,389	3,884	3,973
Percent of Total International Piping Plover Breeding Census	62.9%	42.4%	40.2%	48.2%	69.4%

*NS = not surveyed.

[†] Data from Louisiana is incomplete because of the Deepwater Horizon oil spill. This spill occurred in spring and summer of 2010, affecting the entire northern Gulf of Mexico coastline and in particular the Louisiana Mississippi River Delta (Mendelssohn et al. 2012). Through the Natural Resource Damage Assessment and Restoration (NRDAR) process, some piping plover habitat was closed to the public but subject to special studies to determine oil spill impacts. Data collected as part of the NRDAR process include counts of piping plovers, but these data were not released in time to be included in the 2011 census report. At some time, we should be able to fill in the data gaps, which could add 200 or more piping plovers to the winter count (based on prior census results).

Threats to piping plovers/critical habitat

For the sake of brevity and efficiency, we provide a summary analysis of threats to piping plovers in their migration and wintering range in the following sections. A more in-depth explanation of the threats mentioned here can be found in *Volume II: Draft Revised Recovery Plan for the Wintering Range of the Northern Great Plains Piping Plover (Charadrius melodus) and Comprehensive Conservation Strategy for the Piping Plover in its Coastal Migration and Wintering Range in the Continental United States* (Service 2015a). With minor exceptions, this analysis is focused on threats to piping plovers within the continental U.S. portion of their migration and wintering range. Threats in the Caribbean and Mexico remain largely unknown.

To help the reader determine the relative importance of each threat, we ranked them as low, medium, or high based on how much of a threat they are to the wintering population (Table 3).

Table 3. Piping plover wintering grounds threats matrix. The threats are ranked according to their overall potential impact on the population. The chart represents an overall ranking on the wintering population based on the amount of information currently known, the amount of habitat affected, and the difficulty in ameliorating the threat (Service 2015a).

Threat	Threat Level			
	Low	**Medium**	**High**	**Unknown**
Loss, modification, and degradation of habitat				
Development and construction			X	
Dredging and sand mining			X	
Inlet stabilization and relocation			X	
Groins			X	
Seawalls and revetments			X	
Sand placement projects		X[1]		
Loss of macroinvertebrate prey base due to shoreline stabilization		X		
Invasive vegetation			X[2]	
Wrack removal and beach cleaning		X		
Accelerating sea level rise and other climate change impacts			X	
Weather events				
Storm events	X			
Severe cold weather	X			
Disturbance from recreational activities		X[3]		
Oil spills and other contaminants				
Oil spills		X		
Pesticides and other contaminants	X			
Energy development				
Land-based oil and gas exploration and development	X			
Wind turbines				X
Predation	X			
Military operations	X			
Disease	X			

[1] The threat level of sand placement projects varies among sites and projects. In areas where the loss of critical habitat is imminent due to sea level rise and subsidence, well-designed, infrequent sand placement projects can provide overall benefits to critical habitat once the benthic fauna recovers and natural processes are allowed to reshape the beach and dune system.
[2] The impact and extent of invasive vegetation varies across the range. Regionally, invasive plant growth can have a large impact on habitat availability, while in other parts of the wintering range invasive species are not an issue.
[3] At some sites recreational disturbance would be considered a higher level of threat if the disturbance in essence makes the site unavailable or marginally useful to the plovers.

Loss, Modification, and Degradation of Habitat

The wide, flat, sparsely vegetated barrier beaches, spits, sandbars, and bayside flats preferred by piping plovers in the U.S. are formed and maintained by natural forces and are thus susceptible to degradation caused by development and shoreline stabilization efforts. Development on barrier islands and beachfronts, inlet and shoreline stabilization, inlet dredging, beach maintenance and nourishment activities, seawall installations, and mechanical beach grooming continue to alter natural coastal processes throughout the range of migrating and wintering piping plovers. Dredging of inlets can affect spit formation adjacent to inlets, as well as ebb and flood tidal shoal formation. Jetties stabilize inlets and cause island widening and subsequent vegetation growth on the up-drift inlet shores; they also cause island narrowing and/or erosion on the downdrift inlet shores. Seawalls and revetments restrict natural island movement and exacerbate erosion. Although dredge and fill projects that place sand on beaches and dunes may restore lost or degraded habitat in some areas, in other areas these projects may degrade habitat quality by altering the natural sediment composition, depressing the invertebrate prey base, hindering habitat migration with sea level rise, and replacing the natural habitats of the dune-beach-near-shore system with artificial geomorphology. Construction of any of these projects during months when piping plovers are present also causes disturbance that disrupts the birds' foraging and roosting behaviors. Accelerating sea level rise, which increases erosion and habitat loss where existing development and hardened stabilization structures prevent the natural migration of the beach and/or barrier island exacerbates these threats. Although threats from sea level rise are often considered separately, sea level rise has specific synergistic effects on threats from coastal development and artificial coastal stabilization.

Development and Construction

Development and associated construction threaten the piping plover in its migration and wintering range by degrading, fragmenting, and eliminating habitat. Constructing buildings and infrastructure adjacent to the beach can eliminate roosting and loafing habitat within the development's footprint and degrade adjacent habitat by replacing sparsely vegetated dunes or back-barrier beach areas with landscaping, pools, fences, etc. In addition, the development of bayside or estuarine shorelines, with finger canals and their associated bulkheads, docks, buildings, and landscaping, leads to direct loss and degradation of plover habitat. Finger canals can lead to water pollution, fish kills, loss of aquatic nurseries, saltwater intrusion of groundwater, disruption of surface flows, island breaching due to the funneling of storm surge, and a perpetual need for dredging and disposal of dredged material in order to keep the canals navigable for property owners (Morris et al. 1978; Bush et al. 1996). High-value plover habitat becomes fragmented as lots are developed or coastal roads are built between ocean-side and bayside habitats. Development activities can also include lowering or removing natural dunes to improve views or grade building lots, planting vegetation to stabilize dunes, and erecting sand fencing to establish or stabilize continuous dunes in developed areas. Such activities can further degrade, fragment, and eliminate sparsely vegetated and unvegetated habitats used by the piping plover and other wildlife. Development and construction of other infrastructure in close proximity to barrier beaches often creates economic and social incentives for subsequent shoreline stabilization projects, such as shoreline hardening and beach nourishment. Developed beaches are also highly vulnerable to further habitat loss because they cannot migrate in response to sea level rise.

Approximately 40 percent of the sandy beach shoreline in the migration and wintering range is already developed, and Rice (2012a) has identified over 900 miles (43 percent) of sandy beaches in the wintering range that are currently "preserved" through either public ownership, ownership by non-governmental conservation organizations, or conservation easements. These beaches may be subject to some erosion as they migrate in response to sea level rise or if sediment is removed from the coastal system, and they are vulnerable to recreational disturbance. However, the "preserved" shoreline areas are most likely to maintain the geomorphic characteristics of suitable piping plover habitat. The remaining 17 percent of shoreline habitat in the migration and wintering range (that which is currently undeveloped but not preserved) is susceptible to future loss to development and the attendant threats from shoreline stabilization activities. Nonetheless, the entire coastline regardless of whether it is developed or not is susceptible to sea level rise.

Dredging and Sand Mining

The dredging and mining of sediment from inlet complexes threatens the piping plover on its wintering grounds through habitat loss and degradation. The maintenance of navigation channels by dredging, especially deep shipping channels, can significantly alter the natural coastal processes on inlet shorelines of nearby barrier islands (Otvos 2006; Morton 2008; Otvos and Carter 2008; Beck and Wang 2009; Stockdon et al. 2010). Forty-four percent of the tidal inlets within the U.S. wintering range of the piping plover have been or continue to be dredged, primarily for navigational purposes (Service 2015a). The dredging of navigation channels or relocation of inlet channels for erosion-control purposes contributes to the cumulative effects of inlet habitat modification by removing or redistributing the local and regional sediment supply; the maintenance dredging of deep shipping channels can convert a natural inlet that normally bypasses sediment from one shoreline to the other into a sediment sink, where sediment no longer bypasses the inlet. Additionally, dredging can occur on an annual basis or every two to three years and the volume of sediment removed can be major, resulting in continual perturbations and modifications to inlet and adjacent shoreline habitat.

As sand sources for beach nourishment projects have become more limited, the mining of ebb tidal shoals for sediment has increased (Cialone and Stauble 1998). Exposed shoals and sandbars are valuable to piping plovers, as they tend to receive less human recreational use (because they are only accessible by boat) and therefore provide relatively less disturbed habitats for birds. Removing these sand sources can alter depth contours and change wave refraction as well as cause localized erosion (Hayes and Michel 2008). Ebb shoals are especially important because they act as "sand bridges" that connect beaches and islands by transporting sediment via longshore transport from one side (updrift) to the other (downdrift) side of an inlet. The mining of sediment from these shoals upsets the inlet system equilibrium and can lead to increased erosion of the adjacent inlet shorelines (Cialone and Stauble 1998). Rice (2012b) noted that this mining of material from inlet shoals for use as beach fill is not equivalent to the natural sediment bypassing that occurs at unmodified inlets for several reasons, most notably for the massive volumes involved that are "transported" virtually instantaneously instead of gradually and continuously and for the placement of the material outside of the immediate inlet vicinity, where it would naturally bypass. The mining of inlet shoals can also remove massive amounts of sediment. Cialone and Stauble (1998) found that monitoring of the impacts of ebb shoal mining has been insufficient, and in one case the mining pit was only 66 percent recovered after five years; they conclude that the larger the volume of sediment mined from the shoals, the larger the perturbation to the system and the longer the recovery period.

Information is limited on the effects to piping plover habitat of the deposition of dredged material, and the available information is inconsistent (Drake et al. 2001; Zonick et al. 1998; Zdravkovic and Durkin 2011; Cohen et al. 2008a). Studies have found instances where birds will and will not use islands created from dredged material throughout the wintering range. Research is needed to understand why piping plovers use some dredge material islands, but are not regularly found using others.

In summary, the removal of sediment from inlet complexes via dredging and sand mining for beach fill has modified nearly half of the tidal inlets within the continental wintering range of the piping plover, leading to habitat loss and degradation. Many of these inlet habitat modifications have become permanent, existing for over 100 years. The expansion of several harbors and ports to accommodate deeper draft ships poses an increasing threat as more sediment is removed from the inlet system, causing larger perturbations and longer recovery times; maintenance dredging conducted annually or every few years may prevent full recovery of the inlet system. Sand removal or sediment starvation of shoals, sandbars and adjacent shoreline habitat has resulted in habitat loss and degradation, which may reduce the system's ability to maintain a full suite of inlet habitats as sea level continues to rise at an accelerating rate. Rice (2012b) noted that the adverse impacts of this threat to piping plovers may be mitigated, however, by eliminating dredging and mining activities in inlet complexes with high habitat value, extending the interval between dredging cycles, discharging dredged material in near-shore downdrift waters so that it can accrete more naturally than when placed on the subaerial beach, and designing dredged material islands to mimic natural shoals and flats.

Inlet Stabilization and Relocation

Many navigable mainland or barrier island tidal inlets along the Atlantic and GOM coasts are stabilized with jetties, groins, or by seawalls and/or adjacent industrial or residential development. Jetties are structures built perpendicular to the shoreline that extend through the entire near-shore zone and past the breaker zone to prevent or decrease sand deposition in the channel (Hayes and Michel 2008). Inlet stabilization with rock jetties and associated channel dredging for navigation alter the dynamics of long-shore sediment transport and affect the location and movement rate of barrier islands (Camfield and Holmes 1995), typically causing down-drift erosion. Sediment is then dredged and added back to the islands which are subsequently widened. Once the island becomes stabilized, vegetation encroaches on the bayside habitat, thereby diminishing and eventually destroying its value to piping plovers. Accelerated erosion may compound future habitat loss, depending on the degree of sea-level rise. Unstabilized inlets naturally migrate, re-forming important habitat components over time, whereas jetties often trap sand and cause significant erosion of the down-drift shoreline. These combined actions affect the availability of piping plover habitat (Cohen et al. 2008b).

Tidal inlet relocation and artificial closures can cause loss and/or degradation of piping plover habitat, and although less permanent than construction of hard structures, effects can persist for years. The relocation of inlets or the creation of new inlets often leads to immediate widening of the new inlet and loss of adjacent habitat, among other impacts (Mason and Sorenson 1971; Masterson et al. 1973; Corps 1992; Cleary and Marden 1999; Cleary and Fitzgerald 2003; Erickson et al. 2003; Kraus et al. 2003; Wamsley and Kraus 2005; Kraus 2007). The artificial opening and closing of inlets typically creates very different habitats from those found at inlets that open or close naturally (Rice 2012b). Artificially created inlets tend to need hard structures to remain open or stable. Inlets have been artificially closed, some in response to oil spills and others as part of coastal restoration efforts. Most artificial inlet closures in Louisiana are part of barrier island restoration projects, because much of that state’s barrier islands are disintegrating (Otvos 2006; Morton 2008; Otvos and Carter 2008). Inlets closed during coastal restoration projects in Louisiana are purposefully designed to approximate low, wide naturally closed inlets and to allow over-wash in the future. By contrast, most artificially closed inlets have higher elevations and tend to have a constructed berm and dune system. Over-wash may occur periodically at a naturally closed inlet but is prevented at an artificially closed inlet by the constructed dune ridge, hard structures, or sandbags (Rice 2012b).

Groins

Groins pose an ongoing threat to piping plover beach habitat within the continental wintering range. Groins (structures made of concrete, rip rap, wood, or metal built perpendicular to the beach in order to trap sand) are typically found on developed beaches with severe erosion. Although groins can be individual structures, they are often clustered along the shoreline. Groins can act as barriers to long-shore sand transport and cause down-drift erosion (Hayes and Michel 2008), which prevents piping plover habitat creation by limiting sediment deposition and accretion. The resulting beach typically becomes scalloped in shape, thereby fragmenting plover habitat over time. Groins and groin fields are found throughout the southeastern Atlantic and Gulf Coasts, and although most were in place prior to the piping plover's 1986 listing under the Act, installation of new groins continues to occur, perpetuating the threat to migrating and wintering piping plovers. As sea level rises at an accelerating rate, the threat of habitat loss, fragmentation and degradation from groins and groin fields may increase as communities and beachfront property owners seek additional ways to protect infrastructure and property.

Seawalls and Revetments

Seawalls and revetments are vertical hard structures built parallel to the beach in front of buildings, roads, and other facilities to protect them from erosion. However, these structures often accelerate erosion by causing scouring in front of and down-drift from the structure (Hayes and Michel 2008), which can eliminate intertidal foraging habitat and adjacent roosting habitat. Physical characteristics that determine microhabitats and biological communities can be altered after installation of a seawall or revetment, thereby depleting or changing composition of benthic communities that serve as the prey base for piping plovers. At four California study sites, each comprised of an unarmored segment and a segment seaward of a seawall, Dugan and Hubbard (2006) found that armored segments had narrower intertidal zones, smaller standing crops of macrophyte wrack, and lower shorebird abundance and species richness.

The repair of existing armoring structures and installation of new structures continues to degrade, destroy, and fragment beachfront plover habitat throughout its continental wintering range. As sea level rises at an accelerating rate, the threat of habitat loss, fragmentation and degradation from hard erosion-control structures is likely to increase as communities and property owners seek to protect their beachfront development. As coastal roads become threatened by rising sea level and increasing storm damage, additional lengths of beachfront habitat may be modified by riprap, revetments, and seawalls.

Sand Placement Projects

In the wake of episodic storm events, managers of lands under public, private, and county ownership often protect coastal structures using emergency storm berms; this is frequently followed by beach nourishment or renourishment activities (nourishment projects are considered "soft" stabilization versus "hard" stabilization such as seawalls). Berm placement and beach nourishment deposit substantial amounts of sand along Gulf of Mexico and Atlantic beaches to protect local property in anticipation of preventing erosion and what otherwise will be considered natural processes of over-wash and island migration (Schmitt and Haines 2003). On unpopulated islands, the addition of sand and creation of marsh are sometimes used to counteract the loss of roosting and nesting habitat for shorebirds and wading birds as a result of erosional storm events.

Past and ongoing stabilization projects may fundamentally alter the naturally dynamic coastal processes that create and maintain beach strand and bayside habitats, including those habitat components that piping plovers rely upon. Although impacts may vary depending on a range of factors, stabilization projects may directly degrade or destroy piping plover roosting and foraging habitat in several ways. Front beach habitat may be used to construct an artificial berm that is densely planted in grass, which can directly reduce the availability of roosting habitat. Over time, if the beach narrows due to erosion, additional roosting habitat between the berm and the water can be lost. Over-wash is an essential process, necessary to maintain the integrity of many barrier islands and to create new habitat (Donnelly et al. 2006). Berms can also prevent or reduce the natural over-wash that creates roosting habitats by converting vegetated areas to open sand areas. The vegetation growth caused by impeding natural over-wash can also reduce the maintenance and creation of bayside intertidal feeding habitats. In addition, stabilization projects may indirectly encourage further development of coastal areas and increase the threat of disturbance.

In Louisiana, the sustainability of the coastal ecosystem is threatened by the inability of the barrier islands to maintain geomorphologic functionality (Corps 2011). Consequently, most of the planned sediment placement projects are conducted as environmental restoration projects by various federal and state agencies because without the sediment many areas would erode below sea level since the Louisiana coastal systems are starved for sediment sources. Agencies conducting coastal restoration projects aim to design projects that mimic the natural existing elevations of coastal habitats (e.g., beach, dune, and marsh) in order to allow their projects to work within and be sustained by the natural ecosystem processes that maintain those coastal habitats. Due to the low elevation of barrier islands and coastal headlands, placement of additional sediment in those areas generally does not reach an elevation that would prevent the formation of over-wash areas or impede natural coastal processes, especially during storm events. Such careful design of these restoration projects allows daily tidal processes or storm events to re-work the sediments to reform the Gulf/beach interface and create over-wash areas, sand flats, and mud flats on the bay-side of the islands, as well as sand spits on the ends of the islands; thus, the added sediment aids in sustaining the barrier island system.

Loss of Macroinvertebrate Prey Base due to Shoreline Stabilization

Wintering and migrating piping plovers depend on the availability and abundance of macroinvertebrates as an important food item. Studies of invertebrate communities have found that communities are richer (greater total abundance and biomass) on protected (bay or lagoon) intertidal shorelines than on exposed ocean beach shorelines (McLachlan 1990; Cohen et al. 2006; Defeo and McLachlan 2011). Polychaete worms tend to have a more diverse community and be more abundant in more protected shoreline environments, and mollusks and crustaceans such as amphipods thrive in more exposed shoreline environments (McLachlan and Brown 2006). Polychaete worms comprise the majority of the shorebird diet (Kalejta 1992; Mercier and McNeil 1994; Tsipoura and Burger 1999; Verkuil et al. 2006); and of the piping plover diet in particular (Hoopes 1993; Nicholls 1989; Zonick and Ryan 1995).

The quality and quantity of the macroinvertebrate prey base is threatened by shoreline stabilization activities, including beaches that have received sand placement of various types. The addition of dredged sediment can temporarily affect the benthic fauna of intertidal systems. Invertebrates may be crushed or buried during project construction. Some benthic species can burrow through a thin layer (38-89 cm for different species) of additional sediment since they are adapted to the turbulent environment of the intertidal zone; however, thicker layers (i.e., >1 meter) of sediment are likely to smother the benthic fauna (Greene 2002). Numerous studies of such effects indicate that the recovery of benthic fauna after beach nourishment or sediment placement projects can take anywhere from six months to two years, and possibly longer in extreme cases (Thrush et al. 1996; Peterson et al. 2000; Zajac and Whitlatch 2003; Bishop et al. 2006; Peterson et al. 2006).

Invertebrate communities may also be affected by changes in the physical environment resulting from shoreline stabilization activities that alter the sediment composition or degree of exposure. Shoreline armoring with hard stabilization structures such as seawalls and revetments can alter the degree of exposure of the macroinvertebrate prey base by modifying the beach and intertidal geomorphology, or topography. Seawalls typically result in the narrowing and steepening of the beach and intertidal slope in front of the structure, eventually leading to complete loss of the dry and intertidal beach as sea level continues to rise (Pilkey and Wright 1988; Hall and Pilkey 1991; Dugan and Hubbard 2006; Defeo et al. 2009; Kim et al. 2011). Sand placement projects bury the natural beach with new sediment, and grade the new beach and intertidal zone with heavy equipment to conform to a predetermined topographic profile, which can lead to compaction of the sediment (Nelson et al. 1987; Corps 2008; Defeo et al. 2009).

Invasive Vegetation

An identified threat to piping plover habitat, not described in the listing rule or older recovery plans, is the spread of coastal invasive plants into suitable piping plover habitat. Like most invasive species, coastal exotic plants reproduce and spread quickly and exhibit dense growth habits, often outcompeting native plant species. If left uncontrolled, invasive plants cause a habitat shift from open or sparsely vegetated sand to dense vegetation, resulting in the loss or degradation of piping plover roosting habitat, which is especially important during high tides and migration periods. The propensity of these exotic species to spread, and their tenacity once established, make them a persistent threat, partially countered by increasing landowner awareness and willingness to undertake eradication activities.

Many invasive species are either currently affecting or have the potential to affect coastal beaches and thus plover habitat. Beach vitex is a woody vine introduced into the southeastern U.S. as a dune stabilization and ornamental plant which has spread to coastal communities throughout the southeastern U.S. from Virginia to Florida, and west to Texas (Westbrooks and Madsen 2006). Unquantified amounts of crowfoot grass grow invasively along portions of the Florida coastline. It forms thick bunches or mats that may change the vegetative structure of coastal plant communities and alter shorebird habitat. The Australian pine also changes the vegetative structure of the coastal community in south Florida and islands within the Bahamas. Shorebirds prefer foraging in open areas where they are able to see potential predators, and tall trees provide good perches for avian predators. Australian pines potentially impact shorebirds, including the piping plover, by reducing attractiveness of foraging habitat and/or increasing avian predation. Early detection and rapid response are the keys to controlling this and other invasive plants (Westbrooks 2011 pers. comm.).

Wrack Removal and Beach Cleaning

Wrack on beaches and baysides provides important foraging and roosting habitat for piping plovers (Drake 1999a; Smith 2007; Maddock et al. 2009; Lott et al. 2009) and many other shorebirds on their winter, breeding, and migration grounds. Because shorebird numbers are positively correlated with wrack cover and biomass of their invertebrate prey that feed on wrack (Tarr and Tarr 1987; Hubbard and Dugan 2003; Dugan et al. 2003), beach grooming has been shown to decrease bird numbers (Defeo et al. 2009).

Although beach cleaning and raking machines effectively remove human-made debris, these efforts also remove accumulated wrack, topographic depressions, emergent foredunes and hummocks, and sparse vegetation nodes used by roosting and foraging piping plovers (Nordstrom 2000; Dugan and Hubbard 2010). Removal of wrack also reduces or eliminates natural sand-trapping, further destabilizing the beach. Cathcart and Melby (2009) found that beach grooming and raking beaches "fluffs the sand" whereas heavy equipment compacts the sand below the top layer; the fluffed sand is then more vulnerable to erosion by storm water runoff and wind. In addition, sand adhering to seaweed and trapped in the cracks and crevices of wrack is removed from the beach. Beach cleaning or grooming can result in abnormally broad unvegetated zones that are inhospitable to dune formation or plant colonization, thereby enhancing the likelihood of erosion (Defeo et al. 2009).

Tilling beaches to reduce soil compaction, as sometimes required by the Service for sea turtle protection after beach nourishment activities, also has similar impacts. Recently, the Service improved sea turtle protection provisions in Florida; these provisions now require tilling, when needed, to be above the primary wrack line, rather than within it.

Accelerating sea level rise and other climate change impacts

Numerous studies have documented accelerating rise in sea levels worldwide (Rahmstorf et al. 2007; Douglas et al. 2001 as cited in Hopkinson et al. 2008; USCCSP 2009; Pilkey and Young 2009; Vermeer and Rahmstorf 2009; Pilkey and Pilkey 2011). Predictions include a sea level rise of between 50 and 200 cm above 1990 levels by the year 2100 (Rahmstorf 2007; Pfeffer et al. 2008; Vermeer and Rahmstorf 2009; Grinsted et al. 2010; Jevrejeva et al. 2010) and potential conversion of as much as 33 percent of the world's coastal wetlands to open water by 2080 (Intergovernmental Panel on Climate Change (IPCC) 2007; USCCSP 2008). Potential effects of sea level rise on piping plover roosting and foraging habitats may vary regionally due to subsidence or uplift, the geological character of the coast and near-shore, and the influence of management measures such as beach nourishment, jetties, groins, and seawalls (USCCSP 2009, Galbraith et al. 2002; Gutierrez et al. 2011). Sea level rise along the U.S. Gulf Coast exceeded the global average by 13-15 cm because coastal lands there are subsiding (EPA 2009). The rate of sea level rise in Louisiana is particularly high (Louisiana Coastal Wetlands Conservation and Restoration Task Force and the Wetlands Conservation and Restoration Authority 1999). Sediment compaction and oil and gas extraction compound tectonic subsidence along the GOM coastline (Penland and Ramsey 1990; Morton et al. 2003; Hopkinson et al. 2008).

Low elevations and proximity to the coast make all non-breeding piping plover foraging and roosting habitats vulnerable to the effects of rising sea level. Areas with small tidal ranges are the most vulnerable to loss of intertidal wetlands and flats (EPA 2009). Sea level rise was cited as a contributing factor in the 68 percent decline in tidal flats and algal mats in the Corpus Christi, Texas region (i.e., Lamar Peninsula to Encinal Peninsula) between the 1950s and 2004 (Tremblay et al. 2008). Mapping by Titus and Richman (2001) showed that more than 80 percent of the lowest land along the Atlantic and Gulf coasts was in Louisiana, Florida, Texas, and North Carolina.

Inundation of piping plover habitat by rising seas could lead to permanent loss of habitat, especially if those shorelines are armored with hardened structures (Brown and McLachlan 2002; Dugan and Hubbard 2006; Fish et al. 2008; Defeo et al. 2009). Over-wash and sand migration are impeded on the developed portions of sandy ocean beaches (Smith et al. 2008) that comprise 40 percent of the U.S. non-breeding range (Rice 2012b). As the sea level rises, the ocean-facing beaches erode and attempt to migrate inland. Buildings and artificial sand dunes then prevent sand from washing back toward the lagoons (i.e., bayside), and the lagoon side becomes increasingly submerged during extreme high tides (Scavia et al. 2002). Barrier beach shorebird habitat and natural features that protect mainland developments are both diminished as a result.

Weather Events

Storm Events

Storms are a component of the natural processes that form coastal habitats used by migrating and wintering piping plovers, and positive effects of storm-induced over-wash and vegetation removal have been noted in portions of the wintering range. Hurricane Katrina (2005) over-washed the mainland beaches of Mississippi, creating many tidal flats where piping plovers were subsequently observed (Winstead 2008 pers. comm.). Hurricane Katrina also created a new inlet and improved habitat conditions on some areas of Dauphin Island, Alabama (LeBlanc 2009 pers. comm.).

Conversely, localized storms, since Katrina, have induced habitat losses on Dauphin Island (LeBlanc 2009 pers. comm.). Following Hurricane Ike in 2008, Arvin (2009) reported decreased numbers of piping plovers at some heavily eroded Texas beaches in the center of the storm impact area and increases in plover numbers at sites about 100 miles to the southwest. However, piping plovers were observed later in the season using tidal lagoons and pools that Ike created behind the eroded beaches (Arvin 2009).

The adverse effects on piping plovers attributed to storms are sometimes due to a combination of storms and other environmental changes or human use patterns. For example, four hurricanes between 2002 and 2005 are often cited in reference to rapid erosion of the Chandeleur Islands, a chain of low-lying islands in Louisiana where the 1991 International Piping Plover Census tallied more than 350 piping plovers. Comparison of imagery taken three years before and several days after Hurricane Katrina found that the Chandeleur Islands lost 82 percent of their surface area (Sallenger et al. 2009), and a review of aerial photography prior to the 2006 International Piping Plover Census suggested little piping plover habitat remained (Elliott-Smith et al. 2009). However, Sallenger et al. (2009) noted that habitat changes in the Chandeleur Islands stem not only from the effects of these storms but rather from the combined effects of the storms, long-term (i.e., greater than 1,000 years) diminishing sand supply, and sea-level rise relative to the land.

Other storm-induced adverse effects include post-storm acceleration of human activities such as beach nourishment, sand scraping, and berm and seawall construction. Such stabilization activities can result in the loss and degradation of feeding and resting habitats. Storms also can cause widespread deposition of debris along beaches. Removal of debris often requires large machinery, which can cause extensive disturbance and adversely affect habitat elements such as wrack.

Recent climate change studies indicate a trend toward increasing hurricane numbers and intensity (Emanuel 2005; Webster et al. 2005). When combined with predicted effects of sea-level rise, there may be increased cumulative impacts from future storms. Storms can create or enhance piping plover habitat while causing localized losses elsewhere in the wintering and migration range. Available information suggests that some birds may have resiliency to storms and move to unaffected areas without harm, while other reports suggest birds may perish from storm events. Significant concerns include disturbance to piping plovers and habitats during cleanup of debris along shorelines and post-storm acceleration of shoreline stabilization activities, which can cause persistent habitat degradation and loss.

Severe Cold Weather

Several sources suggest the potential for adverse effects of severe winter cold on survival of piping plovers. The 1996 Atlantic Coast Recovery Plan mentioned high mortality of coastal birds and a drop from approximately 30-40 to 15 piping plovers following an intense 1989 snowstorm along the North Carolina coast (Fussell 1990). A preliminary analysis of survival rates for Great Lakes piping plovers found that the highest variability in survival occurred in spring and correlated positively with minimum daily temperature (weighted mean based on proportion of the population wintering near five weather stations) during the preceding winter (Roche 2010, 2012 pers. comm.). Catlin (2012 pers. comm.) reported that the average mass of ten piping plovers captured in Georgia during unusually cold weather in December 2010 was 5.7 grams (g) less than the average for nine birds captured in October of the same year (46.6 g and 52.4 g, respectively; $p = 0.003$).

Disturbance from Recreational Activities

Increasing human disturbance is a major threat to piping plovers in their coastal migration and wintering range. Intense human disturbance in shorebird winter habitat can be functionally equivalent to habitat loss if the disturbance prevents birds from using an area for a significant amount of time (Goss-Custard et al. 1996), which can lead to roost abandonment and local population declines (Burton et al. 1996). Disturbance can also cause shorebirds to spend less time roosting or foraging and more time in alert postures or fleeing from the disturbances (Johnson and Baldassarre 1988; Burger 1991; Burger 1994; Elliott and Teas 1996; Lafferty 2001a, 2001b; Thomas et al. 2002), which limits the local abundance of piping plovers (Zonick and Ryan 1995; Zonick 2000). Shorebirds that are repeatedly flushed in response to disturbance expend energy on costly short flights (Nudds and Bryant 2000) and may not feed enough to support migration and/or subsequent breeding efforts (Puttick 1979; Lafferty 2001b

Off-road vehicles (ORVs) can also significantly degrade piping plover habitat (Wheeler 1979) or disrupt the birds' normal behavior patterns (Zonick 2000). The 1996 Atlantic Coast Recovery Plan cites tire ruts crushing wrack into the sand, making it unavailable as cover or as foraging substrate (Hoopes 1993; Goldin 1993). The plan also notes that the magnitude of the threat from ORVs is particularly significant, because ORVs extend impacts to remote stretches of beach where human disturbance will otherwise be very slight. Godfrey et al. (1978, 1980 as cited in Lamont et al. 1997) postulated that vehicular traffic along the beach may compact the substrate and kill marine invertebrates that are food for the piping plover. Zonick (2000) found that the density of ORVs negatively correlated with abundance of roosting piping plovers on the ocean beach. Although there is some variability among states, disturbance poses a moderate to high and escalating threat to migrating and wintering piping plovers.

Oil Spills

Piping plovers may accumulate contaminants from point and non-point sources at migratory and wintering sites. Depending on the type and degree of contact, contaminants can have lethal and sub-lethal effects on birds, including behavioral impairment, deformities, and impaired reproduction (Rand and Petrocelli 1985; Gilbertson et al. 1991; Hoffman et al. 1996). Notwithstanding documented cases of lightly oiled piping plovers that have survived and successfully reproduced (Amirault-Langlais et al. 2007; Amos 2009, 2012 pers. comm.), contaminants have both the potential to cause direct toxicity to individual birds and to negatively impact their invertebrate prey base (Chapman 1984; Rattner and Ackerson 2008). Piping plovers' extensive use of the intertidal zone puts them in constant contact with coastal habitats likely to be contaminated by water-borne spills. Negative impacts can also occur during rehabilitation of oiled birds. Frink et al. (1996) describe how standard treatment protocols were modified to reflect the extreme susceptibility of piping plovers to handling and other stressors.

Following the Ixtoc spill, which began on June 3, 1979 off the coast of Mexico, approximately 350 metric tons of oil accumulated on South Texas barrier beaches, resulting in a 79 percent decrease in the total number of infaunal organisms on contaminated portions of the beach (Kindinger 1981; Tunnell et al. 1982). Chapman (1984) collected pre- and post-spill data on the abundance, distribution, and habitat use of shorebirds on the beaches in the affected area and saw declines in the numbers of birds as well as shifts in the habitats used. Shorebirds avoided the intertidal area of the beach, occupying the backshore or moving to estuarine habitats when most of the beach was coated. Chapman surmised that the decline in infauna probably contributed to the observed shifts in habitats used. His observations indicated that all the shorebirds, including piping plovers, avoided the contaminated sediments and concentrated in oil-free areas. Amos, however, reported that piping plovers ranked second to sanderlings in the numbers of oiled birds he observed on the beach, although there was no recorded mortality of plovers due to oil (Amos 2009, 2012 pers. comm.). Oiled birds were seen for a year or more following the initial spill, likely due to continued washing in of sunken tar; but there were only occasional subsequent observations of oiled or tarred plovers (Amos 2009 pers. comm.).

According to government estimates, the 2010 Deepwater Horizon Mississippi Canyon Well #252 oil spill discharged more than 200 million gallons of oil into the GOM (McNutt et al. 2011). Containment activities, recovery of oil-water mix, and controlled burning removed some oil, but additional impacts to natural resources may stem from the 1.84 million gallons of dispersant that were applied to the spill (U.S. Government 2010). Approximately 1,100 miles of shoreline was estimated to be oiled in the GOM. This included approximately 665 miles in Louisiana, 160 miles in Mississippi, 95 miles in Alabama, and 175 miles in Florida (Michel et al. 2013). These numbers do not address cumulative impacts or include shoreline that was cleaned earlier. The USCG, the states, and responsible parties that form the Unified Command (with advice from federal and state natural resource agencies) initiated protective measures and clean-up efforts as provided in contingency plans for each state's coastline. The contingency plans identified sensitive habitats, including all Act-listed species' habitats, which received a higher priority for response actions.

Efforts to prevent shoreline oiling and cleanup response activities can disturb piping plovers and their habitat. Although most piping plovers were on their breeding grounds in May, June, and early July when the Deepwater well was discharging oil, oil was still washing onto Gulf beaches when the plovers began arriving back on the Gulf in mid-July. Ninety percent of piping plovers detected during the prior four years of surveys in Louisiana were in the Deepwater Horizon oil spill impact zone, and Louisiana's Department of Wildlife and Fisheries reported significant disturbance to birds and their habitat from response activities. Wrack lines were removed, and sand washing equipment "cleansed" beaches (Seymour 2011 pers. comm.). Potential long-term adverse effects stem from the construction of sand berms and closing of at least 32 inlets (Rice 2012b). Implementation of prescribed best management practices reduced, but did not negate, disturbance to plovers (and to other beach-dependent wildlife) from cleanup personnel, all-terrain vehicles, helicopters, and other equipment. Service and state biologists present during cleanup operations provided information about breeding, migrating, and wintering birds and their habitat protection needs. However, high staff turnover during the extended spill response period necessitated continuous education and training of clean up personnel (Bimbi 2011 pers. comm.). Limited clean-up operations were still on-going throughout the spill area in November 2012 (Herod 2012 pers. comm.).

More subtle but cumulatively damaging sources of oil and other contaminants are leaking vessels located offshore or within the bays on the Atlantic and Gulf coasts, offshore oil rigs and undersea pipelines in the GOM, pipelines buried under the bay bottoms, and onshore facilities such as petroleum refineries and petrochemical plants. In Louisiana, about 2,500-3,000 oil spills are reported in the Gulf region each year, ranging in size from very small to thousands of barrels (Carver 2011 pers. comm.). Chronic spills of oil from rigs and pipelines and natural seeps in the GOM generally involve small quantities of oil. The oil from these smaller leaks and seeps, if they occur far enough from land, will tend to wash ashore as tar balls. In cases such as this, the impact is limited to discrete areas of the beach, whereas oil slicks from larger spills coat longer stretches of the shoreline (Rice 2009 pers. comm.). In late July and early August 2009, for example, oil suspected to have originated from an offshore oil rig in Mexican waters was observed on plumage or legs of 14 piping plovers in south Texas (Cobb 2012 pers. comm.).

Pesticides and Other Contaminants

Absent identification of contaminated substrates or observation of direct mortality of shorebirds on a site used by migrating and wintering piping plovers, detection of contaminants threats is most likely to occur through analysis of unhatched eggs. Contaminants in eggs can originate from any point in the bird's annual cycle, and considerable effort may be required to ascertain where in the annual cycle exposure occurred (see, for example, Dickerson et al. 2011 characterizing contaminant exposure of mountain plovers).

There has been limited opportunistic testing of piping plover eggs. Polychlorinated biphenol (PCB) concentrations in several composites of Great Lakes piping plover eggs tested in the 1990s had potential to cause reproductive harm. Analysis of prey available to piping plovers at representative Michigan breeding sites indicated that breeding areas along the upper Great Lakes region were not likely the major source of contaminants to this population (Best 1999 pers. comm. in Service 2003). Relatively high levels of PCB, dichloro diphenyl dichloroethylene (DDE), and polybrominated diphenyl ether (PBDE) were detected in one of two clutches of Ontario piping plover eggs analyzed in 2009 (Cavalieri 2011 pers. comm.). Results of opportunistic egg analyses to date from Atlantic Coast piping plovers did not warrant follow-up investigation (Mierzykowski 2009, 2010, 2012; Mierzykowski 2012 pers. comm.). No recent testing has been conducted for contaminants in the Northern Great Plains piping plover population.

Energy Development

Land-based Oil and Gas Exploration and Development

Various oil and gas exploration and development activities occur along the Gulf Coast. Examples of conservation measures prescribed to avoid adverse effects on piping plovers and their habitats include conditions on driving on beaches and tidal flats, restrictions on discharging fresh water across unvegetated tidal flats, timing exploration activities during times when the plovers are not present, and use of directional drilling from adjacent upland areas (Service 2008b; Firmin 2012 pers. comm.). With the implementation of appropriate conditions, threats to non-breeding piping plovers from land-based oil and gas extraction are currently very low.

Wind Turbines

Wind turbines are a potential future threat to piping plovers in their coastal migration and wintering range. Relatively small single turbines have been constructed along the beachfront in at least a few locations (e.g., South Carolina; Caldwell 2012 pers. comm.). Current risk to piping plovers from several wind farms located on the mainland north and west of several bays in southern Texas is deemed low during months of winter residency because the birds are not believed to traverse these areas in their daily movements (Newstead 2012 pers. comm.). To date, no piping plovers have been reported from post-construction carcass detection surveys at these sites (Clements 2012 pers. comm.). However, Newstead (2012a pers. comm.) has raised questions about collision risk during migration departure, as large numbers of piping plovers have been observed in areas of the Laguna Madre east of the wind farms during the late winter.

Furthermore, there is concern that, as sea level rises, the intertidal zone (and potential piping plover activity) may move closer to these sites. Several off-shore wind farm proposals in South Carolina are in various stages of early scoping (Caldwell 2012 pers. comm.).

Predation

The impact of predation on migrating or wintering piping plovers remains largely unknown and is difficult to document. Avian and mammalian predators are common throughout the species' wintering range. Predatory birds are relatively common during fall and spring migration, and it is possible that raptors occasionally take piping plovers (Drake et al. 2001). The 1996 Atlantic Coast Recovery Plan summarized evidence that human activities affect types, numbers, and activity patterns of some predators, thereby exacerbating natural predation on breeding piping plovers. It has been noted, however, that the behavioral response of crouching when in the presence of avian predators may minimize avian predation on piping plovers (Morrier and McNeil 1991; Drake 1999a; Drake et al. 2001).

Military Operations

Five of the eleven coastal military bases located in the U.S. continental range of non-breeding piping plovers have consulted with the Service about potential effects of military activities on plovers and their habitat (Service 2009b). Overall, project avoidance and minimization actions currently reduce threats from military activities to wintering and migrating piping plovers to a minimal threat level.

Disease

No instances of disease have been documented in piping plovers outside the breeding range. The 2009 5-Year Review concluded that West Nile virus and avian influenza remain minor threats to piping plovers on their wintering and migration grounds.

STATUS OF THE SPECIES/CRITICAL HABITAT

Red knot (*Calidris canutus rufa*)

Species/critical habitat description

There are six recognized subspecies of red knots (*Calidris canutus*), and on December 11, 2014, the Service published the final rule listing the rufa subspecies of red knot (*Calidris canutus rufa*) as a threatened species under the Act (Service 2014a); that rule became effective on January 12, 2015. (Throughout this document, the "rufa red knot" will be referred to as the "red knot" unless there is specific reference to a distinct subspecies.) The red knot is a medium-sized shorebird about 9 to 11 inches in length. The red knot is easily recognized during the breeding season by its distinctive rufous (red) plumage. Nonbreeding plumage is dusky gray above and whitish below. Juveniles resemble nonbreeding adults, but the feathers of the scapulars and wing coverts are edged with white and have narrow, dark bands, giving the upperparts a scalloped appearance (Davis 1983).

The Service has determined that the rufa red knot is threatened due to loss of both breeding and nonbreeding habitat; likely effects related to disruption of natural predator cycles on the breeding grounds; reduced prey availability throughout the nonbreeding range; and increasing frequency and severity of asynchronies (‘‘mismatches’’) in the timing of the birds’ annual migratory cycle relative to favorable food and weather conditions. Main threats to the rufa red knot in the U.S. include: reduced forage base at the Delaware Bay migration stopover; decreased habitat availability from beach erosion, sea level rise, and shoreline stabilization in Delaware Bay; reduction in or elimination of forage due to shoreline stabilization, hardening, dredging, beach replenishment, and beach nourishment in Massachusetts, North Carolina, and Florida; and beach raking which diminishes red knot habitat suitability. Critical habitat has not been proposed or designated for the red knot at the time of this document’s writing.

Life history

Breeding

The red knot’s typical life span is at least 7 years (Parvin 2014 pers. comm.; Niles et al. 2008), with the oldest known wild bird at least 21 years old as of 2014 (Bauers 2014; Jordan 2014), and age of first breeding is at least 2 years (Koch 2014 pers. comm.; Niles 2014 pers. comm.; Porter 2014 pers. comm.; Harrington 2001). Red knots nest in the Canadian Arctic in dry, slightly elevated tundra locations, often on windswept slopes with little vegetation. Breeding territories are located inland, but near arctic coasts, and foraging areas are located near nest sites in freshwater wetlands (Niles et al. 2008; Harrington 2001). Breeding occurs in June (Niles et al. 2008), and flocks of red knot sometimes arrive at breeding latitudes before snow-free habitat is available. Upon arrival or as soon as favorable conditions exist, male and female red knots occupy breeding habitat, and territorial displays begin (Harrington 2001). In red knots, pair bonds form soon after arrival on the breeding grounds and remain intact until shortly after the eggs hatch (Niles et al. 2008). Female red knots lay only one clutch per season, and, as far as is known, do not lay a replacement clutch if the first is lost. The usual clutch size is four eggs, though three-egg clutches have been recorded. The incubation period lasts approximately 22 days from the last egg laid to the last egg hatched, and both sexes participate equally in egg incubation. Young are precocial, leaving the nest within 24 hours of hatching and foraging for themselves (Niles et al. 2008). No information is available regarding chick survival rates (Niles et al. 2008). Females are thought to leave the breeding grounds and start moving south soon after the chicks hatch in mid-July. Thereafter, parental care is provided solely by the males, but about 25 days later (around August 10) they also abandon the newly fledged juveniles and move south. Not long after, they are followed by the juveniles (Niles et al. 2008).

Migration

The red knot migrates annually between its breeding grounds in the Canadian Arctic and several wintering regions, including the Southeast United States (Southeast), the Northwest Gulf of Mexico, northern Brazil, and Tierra del Fuego at the southern tip of South America. Departure from the breeding grounds begins in mid-July and continues through August. Red knots tend to migrate in single-species flocks that are generally greater than 50 birds (Niles et al. 2008), with departures typically occurring in the few hours before twilight on sunny days. Based on the duration and distance of migratory flight segments estimated from geolocator results, red knots are inferred to migrate during both day and night (Normandeau Associates, Inc. 2011).

Red knots make one of the longest distance migrations known in the animal kingdom, traveling up to 19,000 miles annually, and may undertake long flights that span thousands of miles without stopping. As red knots prepare to depart on long migratory flights, they undergo several physiological changes. Before takeoff, the birds accumulate and store large amounts of fat to fuel migration and undergo substantial changes in metabolic rates. In addition, leg muscles, gizzard, stomach, intestines, and liver all decrease in size, while pectoral (chest) muscles and heart increase in size. Due to these physiological changes, red knots arriving from lengthy migrations are not able to feed maximally until their digestive systems regenerate, a process that may take several days. Because stopovers are time-constrained, red knots require stopovers rich in easily digested food to achieve adequate weight gain (Niles et al. 2008; van Gils et al. 2005a, 2005b; Piersma et al. 1999) that fuels the next leg of migratory flight and, upon arrival in the Arctic, fuels a body transformation to breeding condition (Morrison 2006). At each stopover when heading south, the adults gradually replace their red breeding plumage with white and gray, but generally they do not molt their flight or tail feathers until they reach their wintering areas (Niles et al. 2008; Morrison and Harrington 1992).

During both the northbound (spring) and southbound (fall) migrations, red knots use key staging and stopover areas to rest and feed. Major spring stopover areas along the Atlantic coast include Río Gallegos, Península Valdés, and San Antonio Oeste in Patagonia, Argentina; Lagoa do Peixe in eastern Brazil; Maranhão in northern Brazil; the Virginia barrier islands in the United States; and Delaware Bay in Delaware and New Jersey, United States (Cohen et al. 2009; Niles et al. 2008; González 2005). Important fall stopover sites include southwest Hudson Bay (including the Nelson River delta), James Bay, the north shore of the St. Lawrence River, the Mingan Archipelago, and the Bay of Fundy in Canada; the coasts of Massachusetts and New Jersey and the mouth of the Altamaha River in Georgia, U.S.; the Caribbean (especially Puerto Rico and the Lesser Antilles); and the northern coast of South America from Brazil to Guyana (Newstead et al. 2013; Niles 2012a; Mizrahi 2011 pers. comm.; Niles et al. 2010; Schneider and Winn 2010; Niles et al. 2008; Harrington 2006 pers. comm.; Antas and Nascimento 1996; Morrison and Harrington 1992; Spaans 1978). However, large and small groups of red knots, sometimes numbering in the thousands, may occur in suitable habitats all along the Atlantic and Gulf coasts from Argentina to Canada during migration (Niles et al. 2008).

Available data indicate that red knots use both coastal and inland migration routes. Red knots wintering in the Southeastern U.S. will move north along the coast to the mid-Atlantic before departing for the Artic and some will depart overland for the Artic directly from the Southeast coast (Bimbi et al. 2014; South Carolina Department of Natural Resources 2013; Niles et al. 2012a; Harrington 2005a; Morrison and Harrington 1992). These eastern red knots typically make a short stop at James Bay in Canada, but may also stop briefly along the Great Lakes, perhaps in response to weather conditions (Niles et al. 2008; Morrison and Harrington 1992). Small numbers of red knots are also reported annually across the interior U.S. (i.e., greater than 25 miles from the Gulf or Atlantic Coasts) during spring and fall migration. Such reported sightings are concentrated along the Great Lakes, but multiple reports have been made from nearly every interior State (eBird.org 2012). Red knots wintering in Texas follow an inland flyway to and from the breeding grounds, using spring and fall stopovers along western Hudson Bay in Canada and in the northern Great Plains (Newstead et al. 2013; Skagen et al. 1999). Thus, red knots from different wintering areas appear to employ different migration strategies, including differences in timing, routes, and stopover areas. However, full segregation of migration strategies, routes, or stopover areas does not occur among red knots from different wintering areas.

Wintering

Red knots occupy all known wintering areas from December to February, but may be present in some wintering areas as early as September or as late as May. Wintering areas for the red knot include the Atlantic coasts of Argentina and Chile (particularly the island of Tierra del Fuego that spans both countries), the north coast of Brazil (particularly in the State of Maranhão), the Northwest Gulf of Mexico from the Mexican State of Tamaulipas through Texas (particularly at Laguna Madre) to Louisiana, and the Southeast United States from Florida (particularly the central Gulf coast) to North Carolina (Newstead et al. 2013; Patrick 2012 pers. comm.; Niles et al. 2008). Smaller numbers of knots winter in the Caribbean, and along the central Gulf coast (Alabama, Mississippi), the mid-Atlantic, and the Northeast United States. Red knots are also known to winter in Central America and northwest South America, but it is not yet clear if those birds are the *rufa* subspecies. Little information exists on where juvenile red knots spend the winter months, and there may be at least partial segregation of juvenile and adult red knots on the wintering grounds.

Migration and Wintering Habitat

Long-distance migrant shorebirds are highly dependent on the continued existence of quality habitat at a few key staging areas. These areas serve as stepping stones between wintering and breeding areas. Habitats used by red knots in migration and wintering areas are generally coastal marine and estuarine habitats with large areas of exposed intertidal sediments. In many wintering and stopover areas, quality high-tide roosting habitat (i.e., close to feeding areas, protected from predators, with sufficient space during the highest tides, free from excessive human disturbance) is limited (Kalasz 2012 pers. comm.; Niles 2012c pers. comm.). The supra-tidal (above the high tide) sandy habitats of inlets provide important areas for roosting, especially at higher tides when intertidal habitats are inundated (Harrington 2008). In some localized areas, red knots will use artificial habitats that mimic natural conditions, such as nourished beaches, dredged spoil sites, elevated road causeways, or impoundments; however, there is limited information regarding the frequency, regularity, timing, or significance of red knots' use of such artificial habitats.

In North America, wintering red knots are commonly found along sandy, gravel, or cobble beaches, tidal mudflats, salt marshes, peat banks, and shallow coastal impoundments, ponds, and lagoons along the Atlantic coast (Cohen et al. 2010; Cohen et al. 2009; Niles et al. 2008; Harrington 2001; Truitt et al. 2001). In Florida, the birds also use mangrove and brackish lagoons. Along the Texas coast, red knots forage on beaches, oyster reefs, and exposed bay bottoms and roost on high sand flats, reefs, and other sites protected from high tides. Red knots also show some fidelity to particular migration staging areas between years (Duerr et al. 2011; Harrington 2001).

Foraging

The red knot is a specialized molluscivore, eating hard-shelled mollusks, sometimes supplemented with easily accessed softer invertebrate prey, such as shrimp- and crab-like organisms, marine worms, and horseshoe crab eggs (Piersma and van Gils 2011; Harrington 2001). Mollusk prey are swallowed whole and crushed in the gizzard (Piersma and van Gils 2011). From studies of other subspecies, Zwarts and Blomert (1992) concluded that the red knot cannot ingest prey with a circumference greater than 1.2 inches. Foraging activity is largely dictated by tidal conditions, as the red knot rarely wades in water more than 0.8 to 1.2 inches deep (Harrington 2001). Due to bill morphology, the red knot is limited to foraging on only shallow-buried prey, within the top 0.8 to 1.2 inches of sediment (Gerasimov 2009; Zwarts and Blomert 1992).

On the breeding grounds, the red knot's diet consists mostly of terrestrial invertebrates such as insects (Harrington 2001). In non-breeding habitats, the primary prey of the red knot include blue mussel spat (juveniles); Donax and Darina clams; snails, and other mollusks, with polycheate worms, insect larvae, and crustaceans also eaten in some locations. A prominent departure from typical prey items occurs each spring when red knots feed on the eggs of horseshoe crabs, particularly during the key migration stopover within the Delaware Bay of New Jersey and Delaware. Delaware Bay serves as the principal spring migration staging area for the red knot because of the availability of horseshoe crab eggs (Clark et al. 2009; Harrington 2001; Harrington 1996; Morrison and Harrington 1992), which provide a superabundant source of easily digestible food.

Population dynamics

<u>Northwest GOM</u>

Except for localized areas, there have been no long-term systematic surveys of red knots in Texas or Louisiana, and no information is available about the number of knots that winter in northeastern Mexico. From survey work in the 1970s, Morrison and Harrington (1992) reported peak winter counts of 120 red knots in Louisiana and 1,440 in Texas, although numbers in Texas between December and February were typically in the range of 100 to 300 birds. Records compiled by Skagen et al. (1999) give peak counts of 2,838 and 2,500 red knots along the coasts of Texas and Louisiana, respectively, between January and June over the period 1980 to 1996, but these figures could include spring migrants. Morrison et al. (2006) estimated only about 300 red knots wintering along the Texas coast, based on surveys in January 2003 (Niles et al. 2008). Higher counts of roughly 700 to 2,500 knots have recently been made on Padre Island, Texas, during October, which could include wintering birds (Newstead et al. 2013; Niles et al. 2009). There are no current estimates for the size of the Northwest GOM wintering group as a whole (Mexico to Louisiana). The best available current estimates for portions of this wintering region are about 2,000 in Texas (Niles 2012a), or about 3,000 in Texas and Louisiana, with about half in each State and movement between them (Hunter 2012 pers. comm.).

Winter occurrences in Louisiana are erratic, and intense survey coverage may be needed to detect knots. Nonetheless they are regarded as somewhat regular in winter. Their gregarious habits (they are frequently found in flocks of 15 to 100 individuals) contribute to their "spotty" distribution in Louisiana (Purrington 2012). The birds seem to disappear in the coldest winters, perhaps moving down the Texas coast or even farther south. Most wintering birds are recorded from the Grand Terre/Grand Isle region west to Raccoon Island, Terrebonne Parish, but presumably some may winter offshore on the seldom-visited Chandeleur Island chain. A high count of 70 knots was recorded on Timbalier Island in February 2011, with more typical winter counts of 1 to 10 birds. Wintering birds appear to be largely absent from the southwestern Louisiana beaches where they are regular during spring and fall migration. The Christmas Bird Count at Grand Isle recorded red knots in 7 of the 10 years from 2004 to 2013, ranging from 0 to 92 birds, and averaging 13.6 birds per year (Russell 2014).

Southeastern United States and Caribbean

Harrington et al. (1988) reported that the mean count of birds wintering in Florida was 6,300 birds based on four aerial surveys conducted from October to January in 1980 to 1982. These surveys covered the Florida Gulf coast from Dunedin to Sanibel-Captiva, sometimes going as far south as Cape Sable (Harrington 2012 pers. comm.). Based on those surveys and other work, the Southeast wintering group was estimated at roughly 10,000 birds in the 1970s and 1980s (Harrington 2005a).

Two recent winter estimates are available for the central GOM. During the International Piping Plover Censuses in 2006 and 2011 (Patrick 2012 pers. comm.), 250 to 500 knots were counted from Alabama to Louisiana. From work related to the Deepwater Horizon oil spill, an estimated 900 red knots were reported from the Florida Panhandle to Mississippi (Hunter 2012 pers. comm.). Older surveys recorded similar numbers from the central Gulf coast, with peak counts of 752 red knots in Alabama (1971) and 40 knots in Mississippi (1979) (Morrison and Harrington 1992).

Status and distribution

The red knot's range spans 40 states, 24 countries, and their administrative territories or regions extending from their breeding grounds in the Canadian Arctic to migration stopover areas along the Atlantic and Gulf coasts of North America to wintering grounds throughout the Southeastern U.S., the Gulf coast, and South America (reaching as far south as Tierra del Fuego at the southern tip of South America). In Delaware Bay and Tierra del Fuego, the era of modern surveys for the red knot and other shorebird species began in the early 1980s. Systematic red knot surveys of other areas began later, and for many portions of the knot's range, available survey data are patchy. Prior to the 1980s, numerous natural history accounts are available, but provide mainly qualitative or localized population estimates. Nonetheless, a consistent narrative emerges across many historical accounts that red knots were extremely abundant in the early 1800s, decreased sharply starting in the mid-1800s, and may have begun to recover by the mid-1900s. Most writers agree the cause of that historical decline was intensive sport and market hunting. It is unclear whether the red knot population fully recovered its historical numbers following the period of unregulated hunting (Harrington 2001).

The current geographic distribution of the red knot has not changed relative to that recorded in historical writings with the notable exception of Delaware Bay. Several early writers reported that red knots breed in the Arctic and winter along the U.S. Gulf coast and in South America including Brazil and Tierra del Fuego (Lowery 1974; Hellmayr and Conover 1948; Bent 1927; Ridgway 1919; Forbush 1912; Eaton 1910; Shriner 1897; Mackay 1893; Audubon 1844). Bent (1927) included Jamaica and Barbados as part of the possible wintering range of red knots, and described knots as "rarely" wintering in parts of Louisiana and Florida. Several writers described the red knot as occurring primarily along the coasts with relatively few sightings inland, but interior migration routes through the central U.S. were also known (Lowery 1974; Hellmayr and Conover 1948; Bent 1927; Ridgway 1919; Forbush 1912; Eaton 1910; Audubon 1844). As with the geographic distribution, a number of historical accounts suggest that the timing of the red knot's spring and fall migrations along the Atlantic coast was generally the same in the past as it is today (Myers and Myers 1979; Urner and Storer 1949; Stone 1937; Bent 1927; Forbush 1912; Shriner 1897; Dixon 1895 in Barnes and Truitt 1997; Mackay 1893; Stearns and Coues 1883; Roosevelt 1866; Giraud 1844; Wilson 1829).

Although the large-scale geographic distribution of migration stopover habitats does not seem to have changed, some authors have noted regional changes in the patterns of red knot stopover habitat usage along the U.S. Atlantic coast. For example, based on a review of early literature, Cohen et al. (2008c) suggest that red knots had a more extensive spring stopover range a century ago than now, with thousands of birds noted in spring in Massachusetts, New York, New Jersey, and Virginia. Harrington et al. (2010) found changes in the regional patterns of stopover habitat usage in Massachusetts, as well as a shift in the wintering destination of birds stopping in Massachusetts during fall migration.

Threats to Red Knots and Their Habitat

In this section, we provide an analysis of threats to red knots and their habitat in their migration and wintering range, with some specific references to their breeding range. Although the red knot's range extends farther than the piping plover's, some similarities exist in habitat use between the species within the U.S. portion of their migration and wintering ranges. Subsequently, there are similarities in the threats to those shared habitat features. The information presented in this section, however, is specific to the red knot and may cover a broader area and/or spectrum of similar threats than the information presented in the *Threats to piping plover/critical habitat* section.

Climate change

The natural history of Arctic-breeding shorebirds makes this group of species particularly vulnerable to global climate change (e.g., Meltofte et al. 2007; Piersma and Lindström 2004; Rehfisch and Crick 2003; Piersma and Baker 2000; Zöckler and Lysenko 2000; Lindström and Agrell 1999). Relatively low genetic diversity, which is thought to be a consequence of survival through past climate-driven population bottlenecks, may put shorebirds at more risk from human-induced climate variation than other avian taxa (Meltofte et al. 2007). Low genetic diversity may result in reduced adaptive capacity as well as increased risks when population sizes drop to low levels.

In the short term, red knots may benefit if warmer temperatures result in fewer years of delayed horseshoe crab spawning in Delaware Bay (Smith and Michaels 2006) or fewer occurrences of late snow melt on the breeding grounds (Meltofte et al. 2007). However, there are indications that changes in the abundance and quality of red knot prey are already under way (Escudero et al. 2012; Jones et al. 2010), and prey species face ongoing climate-related threats from warmer temperatures (Jones et al. 2010; Philippart et al. 2003; Rehfisch and Crick 2003), ocean acidification (NRC 2010; Fabry et al. 2008), and possibly increased prevalence of disease and parasites (Ward and Lafferty 2004). In addition, red knots face imminent threats from loss of habitat caused by sea level rise (NRC 2010; Galbraith et al. 2002; Titus 1990), and increasing asynchronies (‘‘mismatches’’) between the timing of their annual breeding, migration, and wintering cycles and the windows of peak food availability on which the birds depend (Smith et al. 2011; McGowan et al. 2011; Meltofte et al. 2007; van Gils et al. 2005a; Baker et al. 2004).

Several threats are related to the possibility of changing storm patterns. While variation in weather is a natural occurrence and is normally not considered a threat to the survival of a species, persistent changes in the frequency, intensity, or timing of storms at key locations where red knots congregate (e.g., key stopover areas) can pose a threat. Storms impact migratory shorebirds like the red knot both directly and indirectly. Direct impacts include blowing birds off course, energetic costs from a longer migration route as birds avoid storms, and outright mortality (Niles et al. 2010). Indirect impacts include changes to habitat suitability, storm-induced asynchronies between migration stopover periods and the times of peak prey availability, and possible prompting of birds to take refuge in areas where shorebird hunting is still practiced (Niles et al. 2012b; Dey et al. 2011; Nebel 2011).

With arctic warming, vegetation conditions in the red knot’s breeding grounds are expected to change, causing the zone of nesting habitat to shift and perhaps contract, but this process may take decades to unfold (Feng et al. 2012; Meltofte et al. 2007; Kaplan et al. 2003). Ecological shifts in the Arctic may appear sooner. High uncertainty exists about when and how changing interactions among vegetation, predators, competitors, prey, parasites, and pathogens may affect the red knot, but the impacts are potentially profound (Fraser et al. 2013; Schmidt et al. 2012; Meltofte et al. 2007; Ims and Fuglei 2005).

Due to background rates of sea level rise and the naturally dynamic nature of coastal habitats, we conclude that red knots are adapted to moderate (although sometimes abrupt) rates of habitat change in their wintering and migration areas. However, rates of sea level rise are accelerating beyond those that have occurred over recent millennia. In most of the red knot’s nonbreeding range, shorelines are expected to undergo dramatic reconfigurations over the next century as a result of accelerating sea level rise. Extensive areas of marsh are likely to become inundated, which may reduce foraging and roosting habitats. Marshes may be able to establish farther inland, but the rate of new marsh formation (e.g., intertidal sediment accumulation, development of hydric soils, colonization of marsh vegetation, etc.) may be slower than the rate of deterioration of existing marsh, particularly under higher sea level rise scenarios. The primary red knot foraging habitats (i.e., intertidal flats and sandy beaches) will likely be locally or regionally inundated, but replacement habitats are likely to reform along the shoreline in its new position. However, if shorelines experience a decades-long period of high instability and landward migration, the formation rate of new beach habitats may be slower than the inundation rate of existing habitats. In addition, low-lying and narrow islands (e.g., in the Caribbean and along the Gulf and Atlantic coasts) may disintegrate rather than migrate, representing a net loss of red knot habitat. Superimposed on these changes are widespread human attempts to stabilize the shoreline, which are known to exacerbate losses of intertidal habitats by blocking their landward migration. The cumulative loss of habitat across the nonbreeding range could affect the ability of red knots to complete their annual cycles, possibly affecting fitness and survival, and is thereby likely to negatively influence the long-term survival of the red knot.

Reduced food availability

Commercial harvest of horseshoe crabs has been implicated as a causal factor in the decline of the red knot populations in the 2000s, by decreasing the availability of horseshoe crab eggs in the Delaware Bay stopover (Niles et al. 2008). Due to harvest restrictions and other conservation actions, horseshoe crab populations showed some signs of recovery in the early 2000s, with apparent signs of red knot stabilization (survey counts, rates of weight gain) occurring a few years later (as might be expected due to biological lag times). Since about 2005, however, horseshoe crab population growth has stagnated for unknown reasons. Under the current management framework, the present horseshoe crab harvest is not considered a threat to the red knot. However, it is not yet known if the horseshoe crab egg resource will continue to adequately support red knot populations over the next 5 to 10 years. In addition, implementation of the current management framework could be impeded by insufficient funding.

The causal role of reduced Delaware Bay food supplies in driving red knot population declines shows the vulnerability of red knots to declines in the quality or quantity of their prey. In addition to the fact that horseshoe crab population growth has stagnated, red knots now face several emerging threats to their food supplies throughout their nonbreeding range. These threats include: small prey sizes (from unknown causes) at two key wintering sites on Tierra del Fuego; warming water temperatures that may cause mollusk population declines and range contractions (including the likely loss of a key prey species from the Virginia spring stopover within the next decade); ocean acidification to which mollusks are particularly vulnerable; physical habitat changes from climate change affecting invertebrate communities; possibly increasing rates of mollusk diseases due to climate change; invasive marine species from ballast water and aquaculture; and the burial and crushing of invertebrate prey from sand placement and recreational activities. Although threats to food quality and quantity are widespread, red knots in localized areas have shown some adaptive capacity to switch prey when the preferred prey species became reduced (Escudero et al. 2012; Musmeci et al. 2011). Nonetheless, based on the combination of documented past impacts and a spectrum of ongoing and emerging threats, we conclude that reduced quality and quantity of food supplies is a threat to the rufa red knot at the subspecies level, and the threat is likely to continue into the future.

Shoreline stabilization and coastal development

Much of the U.S. coast within the range of the red knot is already extensively developed. Direct loss of shorebird habitats occurred over the past century as substantial commercial and residential developments were constructed in and adjacent to ocean and estuarine beaches along the Atlantic and Gulf coasts. In addition, red knot habitat was also lost indirectly, as sediment supplies were reduced and stabilization structures were constructed to protect developed areas. The damming of rivers, bulk-heading of highlands, and armoring of coastal bluffs have reduced erosion in natural source areas and consequently the sediment loads reaching coastal areas. Although it is difficult to quantify, the cumulative reduction in sediment supply from human activities may contribute substantially to the long-term shoreline erosion rate. Along coastlines subject to sediment deficits, the amount of sediment supplied to the coast is less than that lost to storms and coastal sinks (inlet channels, bays, and upland deposits), leading to long-term shoreline recession (Coastal Protection and Restoration Authority of Louisiana 2012; Florida Oceans and Coastal Council 2010; USCCSP 2009; Defeo et al. 2009; Morton et al. 2004; Morton 2003; Herrington 2003; Greene 2002).

In addition to reduced sediment supplies, other factors such as stabilized inlets, shoreline stabilization structures, and coastal development can exacerbate long-term erosion (Herrington 2003). Coastal development and shoreline stabilization can be mutually reinforcing. Coastal development often encourages shoreline stabilization because stabilization projects cost less than the value of the buildings and infrastructure. Conversely, shoreline stabilization sometimes encourages coastal development by making a previously high-risk area seem safer for development (USCCSP 2009). Protection of developed areas is the driving force behind ongoing shoreline stabilization efforts. Large-scale shoreline stabilization projects became common in the past 100 years with the increasing availability of heavy machinery. Shoreline stabilization methods change in response to changing new technologies, coastal conditions, and preferences of residents, planners, and engineers. Along the Atlantic and Gulf coasts, an early preference for shore-perpendicular structures (e.g., groins) was followed by a period of construction of shore-parallel structures (e.g., seawalls), and then a period of beach nourishment, which is now favored (Morton et al. 2004; Nordstrom 2000).

Past and ongoing stabilization projects fundamentally alter the naturally dynamic coastal processes that create and maintain beach strand and bayside habitats, including those habitat components that red knots rely upon. Past loss of stopover and wintering habitat likely reduce the resilience of the red knot by making it more dependent on those habitats that remain, and more vulnerable to threats (e.g., disturbance, predation, reduced quality or abundance of prey, increased intraspecific and interspecific competition) within those restricted habitats.

Hard structures

Hard structures constructed of stone, concrete, wood, steel, or geotextiles have been used for centuries as a coastal defense strategy (Defeo et al. 2009). The most common hard stabilization structures fall into two groups: structures that run parallel to the shoreline (e.g., seawalls, revetments, bulkheads) and structures that run perpendicular to the shoreline (e.g., groins, jetties). Groins are often clustered in groin fields, and are intended to protect a finite section of beach, while jetties are normally constructed at inlets to keep sand out of navigation channels and provide calm-water access to harbor facilities (Corps 2002). Descriptions of the different types of stabilization structures can be found in Rice (2009), Herrington (2003), and Corps (2002, Parts V and VI).

Prior to the 1950s, the general practice in the United States was to use hard structures to protect developments from beach erosion or storm damages, but the pace of constructing new hard stabilization structures has since slowed considerably (Corps 2002). Many states within the range of the red knot now discourage or restrict the construction of new, hard oceanfront protection structures, although the hardening of bayside shorelines is generally still allowed (Kana 2011; Greene 2002; Titus 2000). Most existing hard oceanfront structures continue to be maintained, and some new structures continue to be built. While some states have restricted new construction, hard structures are still among the alternatives in the Federal shore protection program (Corps 2002).

Hard shoreline stabilization projects are typically designed to protect property (and its human inhabitants) not beaches (Kana 2011; Pilkey and Howard 1981). Through effects on waves and currents, sediment transport rates, Aeolian (wind) processes, and sand exchanges with dunes and offshore bars, hard structures change the erosion/accretion dynamics of beaches and constrain the natural migration of shorelines (USCCSP 2009; Defeo et al. 2009; Morton 2003; Scavia et al. 2002; Nordstrom 2000). There is ample evidence of accelerated erosion rates, pronounced breaks in shoreline orientation, and truncation of the beach profile down-drift of perpendicular structures, and of reduced beach widths (relative to unprotected segments) where parallel structures have been in place over long periods of time (Hafner 2012; USCCSP 2009; Morton 2003; Scavia et al. 2002; Corps 2002; Nordstrom 2000; Pilkey and Wright 1988). In addition, marinas and port facilities built out from the shore can have effects similar to hard stabilization structures (Nordstrom 2000).

Structural development along the shoreline and manipulation of natural inlets upset the naturally dynamic coastal processes and result in loss or degradation of beach habitat (Melvin et al. 1991). As beaches narrow, the reduced habitat can directly lower the diversity and abundance of biota, especially in the upper intertidal zone. Shorebirds may be impacted both by reduced habitat area for roosting and foraging, and by declining intertidal prey resources, as has been documented in California (Defeo et al. 2009; Dugan and Hubbard 2006). In addition to directly eliminating red knot habitat, hard structures interfere with the creation of new shorebird habitats by interrupting the natural processes of over-wash and inlet formation. Where hard stabilization is installed, the eventual loss of the beach and its associated habitats is virtually assured (Rice 2009), absent beach nourishment, which may also impact red knots. Where they are maintained, hard structures are likely to significantly increase the amount of red knot habitat lost as sea levels continue to rise.

Mechanical sediment transport

Several types of sediment transport are employed to stabilize shorelines, protect development, maintain navigation channels, and provide for recreation (Gebert 2012; Kana 2011; Corps 2002). The effects of these projects are typically expected to be relatively short in duration, usually less than 10 years, but often these actions are carried out every few years in the same area, resulting in a more lasting impact on habitat suitability for shorebirds. Mechanical sediment transport practices include beach nourishment, sediment back-passing, sand scraping, and dredging.

Since the 1970s, 90 percent of the Federal appropriation for shore protection has been for beach nourishment (Corps 2002), which has become the preferred course of action to address shoreline erosion in the U.S. (Kana 2011; Morton and Miller 2005; Greene 2002). Beach nourishment requires an abundant source of sand that is compatible with the native beach material. The sand is trucked to the target beach or hydraulically pumped using dredges (Hafner 2012). Sand for beach nourishment operations can be obtained from dry land-based sources; estuaries, lagoons, or inlets on the backside of the beach; sandy shoals in inlets and navigation channels; near-shore ocean waters; or offshore ocean waters; with the last two being the most common sources (Greene 2002).

Where shorebird habitat has been severely reduced or eliminated by hard stabilization structures, beach nourishment may be the only means available to replace any habitat for as long as the hard structures are maintained (Nordstrom and Mauriello 2001), although such habitat will persist only with regular nourishment episodes (typically on the order of every two to six years). In Delaware Bay, beach nourishment has been recommended to prevent loss of spawning habitat for horseshoe crabs (Kalasz 2008; Carter et al. in Guilfoyle et al. 2007; ASMFC 1998), and is being pursued as a means of restoring shorebird habitat in Delaware Bay following Hurricane Sandy (Niles et al. 2013; Corps 2012). However, red knots may be directly disturbed if beach nourishment takes place while the birds are present. In addition to causing disturbance during construction, beach nourishment often increases recreational use of the widened beaches that, without careful management, can increase disturbance of red knots. Beach nourishment can also temporarily depress, and sometimes permanently alter, the invertebrate prey base on which shorebirds depend. Using sediment dissimilar to the native beach sand can also affect the red knot's ability to locate and capture prey (Peterson et al. 2006).

In addition to disturbing the birds and impacting the prey base, beach nourishment can affect the quality and quantity of red knot habitat (Bimbi 2012 pers. comm.; Greene 2002). The artificial beach created by nourishment may provide only suboptimal habitat for red knots, as a steeper beach profile is created when sand is stacked on the beach during the nourishment process. In some cases, nourishment is accompanied by the planting of dense beach grasses, which can directly degrade habitat, as red knots require sparse vegetation to avoid predation. By precluding over-wash and Aeolian transport, especially where large artificial dunes are constructed, beach nourishment can also lead to further erosion on the bayside and promote bayside vegetation growth, both of which can degrade the red knot's preferred foraging and roosting habitats. Preclusion of over-wash also impedes the formation of new red knot habitats. Beach nourishment can also encourage further development, bringing further habitat impacts, reducing future alternative management options such as a retreat from the coast, and perpetuating the developed and stabilized conditions that may ultimately lead to inundation where beaches are prevented from migrating (Bimbi 2012 pers. comm.; Greene 2002).

Sediment back-passing is a technique that reverses the natural migration of sediment by mechanically (via trucks) or hydraulically (via pipes) transporting sand from accreting, downdrift areas of the beach to eroding, up-drift areas of the beach (Kana 2011; Chasten and Rosati 2010). Currently less prevalent than beach nourishment, sediment back-passing is an emerging practice because traditional nourishment methods are beginning to face constraints on budgets and sediment availability (Hafner 2012; Chase 2006). Beach bulldozing or scraping is the process of mechanically redistributing beach sand from the littoral zone (along the edge of the sea) to the upper beach to increase the size of the primary dune or to provide a source of sediment for beaches that have no existing dune; no new sediment is added to the system (Kana 2011; Greene 2002; Lindquist and Manning 2001). Beach scraping tends to be a localized practice. Many of the effects of sediment back-passing and beach scraping are similar to those for beach nourishment (Service 2011b; Lindquist and Manning 2001), including disturbance during and after construction, alteration of prey resources, reduced habitat area and quality, and precluded formation of new habitats. Relative to beach nourishment, sediment back-passing and beach scraping can involve considerably more driving of heavy trucks and other equipment on the beach including areas outside the sand placement footprint, potentially impacting shorebird prey resources over a larger area (Service 2011b).

Sediments are also manipulated to maintain navigation channels. Many inlets in the U.S. range of the red knot are routinely dredged and sometimes relocated. In addition, near-shore areas are routinely dredged to obtain sand for beach nourishment. Regardless of the purpose, inlet and near-shore dredging can affect red knot habitats. Dredging often involves removal of sediment from sand bars, shoals, and inlets in the near-shore zone, directly impacting optimal red knot roosting and foraging habitats (Harrington 2008; Harrington in Guilfoyle et al. 2007; Winn and Harrington in Guilfoyle et al. 2006). These ephemeral habitats are even more valuable to red knots because they tend to receive less recreational use than the main beach strand. In addition to causing this direct habitat loss, the dredging of sand bars and shoals can preclude the creation and maintenance of red knot habitats by removing sand sources that would otherwise act as natural breakwaters and weld onto the shore over time (Hayes and Michel 2008; Morton 2003). Further, removing these sand features can cause or worsen localized erosion by altering depth contours and changing wave refraction (Hayes and Michel 2008), potentially degrading other nearby red knot habitats indirectly because inlet dynamics exert a strong influence on the adjacent shorelines.

Wrack removal and beach cleaning

The effects of wrack removal and beach cleaning to red knot migration and wintering habitat are similar to those described in the *Threats to piping plovers/critical habitat* section of this document. The occurrence of beach raking in the Southeast and along the Gulf coast was also discussed in that section. Therefore, that information will not be reiterated here.

Invasive vegetation

The effects of invasive vegetation to red knot migration and wintering habitat are similar to those described in the *Threats to piping plovers/critical habitat* section of this document. Therefore, that information will not be reiterated here and we provide the following summary.

Red knots require open habitats that allow them to see potential predators and that are away from tall perches used by avian predators. Invasive species, particularly woody species, degrade or eliminate the suitability of red knot roosting and foraging habitats by forming dense stands of vegetation. The propensity of invasive species to spread, and their tenacity once established, make them a persistent problem that is only partially countered by increasing awareness and willingness of beach managers to undertake control efforts (Service 2012c). Although the extent of the threat is uncertain, that uncertainty may be due to poor survey coverage more than an absence of species invasions. Even though they are not a primary cause of habitat loss, invasive species can be a regionally important contributor to the overall loss and degradation of the red knot's nonbreeding habitat.

Aquaculture and agriculture

In some localized areas within the red knot's range, aquaculture or agricultural activities are impacting habitat quality and quantity. In the United States, Luckenbach (2007) found that aquaculture of clams in the lower Chesapeake Bay occurs in close proximity to shorebird foraging areas. The current distribution of clam aquaculture in the very low intertidal zone minimizes the amount of direct overlap with shorebird foraging habitats, but if clam aquaculture expands farther into the intertidal zone, more shorebird impacts (e.g., habitat alteration) may occur.

Disease

Red knots are exposed to parasites and disease throughout their annual cycle. Susceptibility to disease may be higher when the energy demands of migration have weakened the immune system. Studying red knots in Delaware Bay in 2007, Buehler et al. (2010) found that several indices of immune function were lower in birds recovering protein after migration than in birds storing fat to fuel the next leg of the migration. These authors hypothesized that fueling birds may have an increased rate of infection or may be bolstering immune defense, or recovering birds may be immuno-compromised because of the physical strain of migratory flight or as a result of adaptive energy tradeoffs between immune function and migration, or both (Buehler et al. 2010). A number of known parasites (e.g., sporozoans, hookworms, flatworms, and ectoparasites) and viruses (e.g., avian influenza and avian paramyxovirus) have been documented in red knots, but we have no evidence that disease is a current threat to the red knot.

Predation

In wintering and migration areas, the most common predators of red knots are peregrine falcons, harriers, accipiters, merlins, short-eared owls, and greater black-backed gulls (Niles et al. 2008). In addition to greater black-backed gulls, other large gulls (e.g., herring gulls) are anecdotally known to prey on shorebirds (Breese 2010). Predation by a great horned owl has been documented in Florida (Schwarzer 2013 pers. comm.). Nearly all documented predation of wintering red knots in Florida has been by avian, not terrestrial, predators (Schwarzer 2013 pers. comm.). However in migration areas like Delaware Bay, terrestrial predators such as red foxes and feral cats may be a threat to red knots by causing disturbance, but direct mortality from these predators may be low (Niles et al. 2008).

In wintering and migration areas, predation is not directly impacting red knot populations despite some direct mortality. At key stopover sites, however, localized predation pressures are likely to exacerbate other threats to red knot populations, such as habitat loss, food shortages, and asynchronies between the birds' stopover period and the occurrence of favorable food and weather conditions. Predation pressures worsen these threats by pushing red knots out of otherwise suitable foraging and roosting habitats, causing disturbance, and possibly causing changes to stopover duration or other aspects of the migration strategy.

Human disturbance

The effects of human disturbance to red knot migration and wintering habitat are similar to those described in the *Threats to piping plovers/critical habitat* section of this document. Therefore, that information will not be reiterated here and we provide the following summary.

In some wintering and stopover areas, red knots and recreational users (e.g., pedestrians, ORVs, dog walkers, boaters) are concentrated on the same beaches (Niles et al. 2008; Tarr 2008). Recreational activities affect red knots both directly and indirectly. These activities can cause habitat damage (Schlacher and Thompson 2008; Anders and Leatherman 1987), cause shorebirds to abandon otherwise preferred habitats, negatively affect the birds' energy balances, and reduce the amount of available prey. Effects to red knots from vehicle and pedestrian disturbance can also occur during construction of shoreline stabilization projects including beach nourishment. Red knots can also be disturbed by motorized and non-motorized boats, fishing, kite surfing, aircraft, and research activities (Kalasz 2011 pers. comm.; Niles et al. 2008; Peters and Otis 2007; Harrington 2005b; Meyer et al. 1999; Burger 1986) and by beach raking. In Delaware Bay, red knots could also potentially be disturbed by hand-harvest of horseshoe crabs during the spring migration stopover period, but under the current management of this fishery, state waters from New Jersey to coastal Virginia are closed to horseshoe crab harvest and landing from January 1 to June 7 each year (ASMFC 2012); thus, disturbance from horseshoe crab harvest is no longer occurring. Active management can be effective at reducing and minimizing the adverse effects of recreational disturbance (Burger and Niles 2013; Forys 2011; Burger et al. 2004), but such management is not occurring throughout the red knot's range.

Red knots are exposed to disturbance from recreational and other human activities throughout their nonbreeding range. Excessive disturbance has been shown to preclude shorebird use of otherwise preferred habitats and can impact energy budgets. Both of these effects are likely to exacerbate other threats to the red knot, such as habitat loss, reduced food availability, asynchronies in the annual cycle, and competition with gulls (such competition is greater in Delaware Bay when foraging on horseshoe crab eggs; in other areas, the two species' diets do not tend to overlap).

Harmful algal blooms

A harmful algal bloom (HAB) is the proliferation of a toxic or nuisance algal species (which can be microscopic or macroscopic, such as seaweed) that negatively affects natural resources or humans (FWC 2015a). The primary groups of microscopic species that form HABs are flagellates (including dinoflagellates), diatoms, and blue-green algae (cyanobacteria). Blooms can appear green, brown, or red-orange, or may be colorless, depending upon the species blooming and environmental conditions. Although HABs are popularly called ''red tides,'' this name can be misleading, as it includes many blooms that discolor the water but cause no harm, while also excluding blooms of highly toxic cells that cause problems at low (and essentially invisible) concentrations (Woods Hole 2012).

For shorebirds, shellfish are a key route of exposure to algal toxins. When toxic algae are filtered from the water as food by shellfish, their toxins accumulate in those shellfish to levels that can be lethal to animals that eat the shellfish (Anderson 2007). Several shellfish poisoning syndromes that occur prominently within the range of the red knot include: Amnesic Shellfish Poisoning (ASP), occurring in Atlantic Canada, caused by *Pseudo-nitzchia* spp.; Neurotoxic Shellfish Poisoning (NSP, also called ‘‘red tide’’), occurring on the U.S. coast from Texas to North Carolina, caused by *Karenia brevis* and other species; and Paralytic Shellfish Poisoning (PSP), occurring in Atlantic Canada, the U.S. coast in New England, Argentina, and Tierra del Fuego, caused by *Alexandrium* spp. and others (Woods Hole 2012; Food and Agriculture Organization of the United Nations (FAO) 2004). Algal toxins may be a direct cause of death in seabirds and shorebirds via an acute or lethal exposure, or birds can be exposed to chronic, sub-lethal levels of a toxin over the course of an extended bloom. Sub-acute doses may contribute to mortality due to an impaired ability to forage productively, disrupted migration behavior, reduced nesting success, or increased vulnerability to predation, dehydration, disease, or injury (van Deventer 2007).

To date, direct impacts to red knots from HABs have been documented only in Texas, although a large die-off in Uruguay may have also been linked to a HAB. We conclude that some level of undocumented red knot mortality from HABs likely occurs most years, based on probable underreporting of shorebird mortalities from HABs and the direct exposure of red knots to algal toxins (particularly via contaminated prey) throughout the knot’s nonbreeding range. We have no documented evidence that HABs were a driving factor in red knot population declines in the 2000s. However, HAB frequency and duration have increased and do not show signs of abating over the next few decades. Combined with other threats, ongoing and possibly increasing mortality from HABs may affect the red knot at the population level.

Oil spills

The red knot has the potential to be exposed to oil spills and leaks throughout its migration and wintering range. Oil, as well as spill response activities, can directly and indirectly affect both the bird and its habitat through several pathways. Red knots can be exposed to petroleum products via spills from shipping vessels, leaks or spills from offshore oil rigs or undersea pipelines, leaks or spills from onshore facilities such as petroleum refineries and petrochemical plants, and beach-stranded barrels and containers that can fall from moving cargo ships or offshore rigs. Several key red knot wintering or stopover areas also contain large-scale petroleum extraction, transportation, or both activities. With regard to potential effects on red knot habitats, the geographic location of a spill, weather conditions (e.g., prevailing winds), and type of oil spilled are as important, if not more so, than the volume of the discharge.

Red knots are exposed to large-scale petroleum extraction and transportation operations in many key wintering and stopover habitats including Tierra del Fuego, Patagonia, the Gulf of Mexico, Delaware Bay, and the Gulf of St. Lawrence. To date, the documented effects to red knots from oil spills and leaks have been minimal. See the *Threats to piping plovers/critical habitat* section of this document for further details regarding potential impacts related to the Deepwater Horizon oil spill.

Wind energy development

Within the red knot's U.S. wintering and migration range, substantial development of offshore wind facilities is planned, and the number of wind turbines installed on land has increased considerably over the past decade. The rate of wind energy development will likely continue to increase into the future as the United States looks to decrease reliance on the traditional sources of energy (e.g., fossil fuels). Wind turbines can have a direct (e.g., collision mortality) and indirect (e.g., migration disruption, displacement from habitat) impact on shorebirds.

We are not aware of any documented red knot mortalities at any wind turbines to date, but low levels of red knot mortality from turbine collisions may be occurring now based on the number of turbines along the red knot's migratory routes and the frequency with which red knots traverse these corridors. Based on the current number and geographic distribution of turbines, if any such mortality is occurring, it is likely not causing subspecies-level effects. However, our primary concern is that, as build-out of wind energy infrastructure progresses, especially near the coast, increasing mortality from turbine collisions may contribute to a subspecies-level effect due to the red knot's modeled vulnerability to direct human-caused mortality (Watts 2010). We anticipate that the threat to red knots from wind turbines will be primarily related to collision or behavioral changes during migratory or daily flights. Unless facilities are constructed at key stopover or wintering habitats we do not expect wind energy development to cause significant direct habitat loss or degradation or displacement of red knots from otherwise suitable habitats.

Analysis of the species/critical habitat likely to be affected

Piping plovers and red knots that winter in the coastal/beach areas along the Gulf coast may be affected by the proposed action. Critical habitat has been designated for the piping plovers within this area and may also be affected by the proposed action. Because critical habitat has not yet been designated for the red knot within the action area, none will be affected. The effects of the proposed action on piping plovers and red knots and their habitat will be considered further in the remaining sections of this opinion.

ENVIRONMENTAL BASELINE

Because the piping plover and red knot share similar coastal habitats along the Gulf Coast, the habitat environmental baseline and effect of the action are essentially the same for both species. Therefore, in order to produce an efficient and effective consultation, the following sections discuss the mutual environmental baseline conditions for both species. Any differences that may occur between the species' habitat descriptions are indicated.

Status of the species within the action area

The piping plover occurs along the Gulf Coast from Florida to Texas within the action area. It occurs in two counties in Alabama, 18 counties throughout Florida, eight counties in Louisiana, and 131 counties in Texas (26 of which are coastal counties within the action area) (Service 2015b). About 89 percent of birds that are known to winter in the U.S. do so along the Gulf Coast (Texas to Florida). Results from the 2011 International Piping Plover Census indicate that the Bahamas are also an important wintering area for piping plovers. It inhabits sand beaches (sand, mud, or algal flats) or over-wash passes in their wintering areas (Service 2009a). In 2001, 142 areas along the coasts of North Carolina, South Carolina, Georgia, Florida, Alabama, Mississippi, Louisiana, and Texas were designated as critical habitat for the wintering population of the piping plover (Service 2001b). This designation included approximately 1,798 miles of mapped shoreline and approximately 165,211 acres of mapped areas along the Gulf and Atlantic coasts and along margins of interior bays, inlets, and lagoons. Subsequent designations were revised to include approximately 139,029 acres of critical habitat in nine counties in Texas.

The red knot also occurs along the Gulf Coast from Florida to Texas within the action area. In the action area, recent modeling suggests the southeast wintering group of red knots may number as high as 20,000. In the southeast wintering region, there was an apparent decline on Florida's Gulf coast population when comparing aerial surveys from 1980 to 1982 to surveys for 2006 to 2010. Two recent winter estimates counted 250 to 500 knots from Alabama to Louisiana. From work related to the DWH oil spill (Service 2015c), an estimated 900 red knots were reported from the Florida Panhandle to Mississippi (Service 2015d). The best available current estimates for portions of the northwest GOM wintering region are about 3,000 in Texas and Louisiana. No critical habitat has been proposed for the red knot at the time this document is being written, thus, none will be affected.

Factors affecting species environment within the action area

A variety of human-caused disturbance factors have been noted that may affect plover survival or utilization of wintering habitat. Those factors include recreational activities, inlet and shoreline stabilization, dredging of inlets that can affect spit formation, beach maintenance and renourishment, and pollution. In some areas, particularly Louisiana, erosion of barrier islands may also result in habitat loss. That loss may be the result of an ongoing process or significant events such as hurricanes. The Service represents the Department of Interior on the Louisiana Coastal Wetlands Conservation and Restoration Task Force. That Task Force oversees planning, evaluation, funding and implementation of projects expected to protect and restore coastal habitats (including wetlands and barrier beaches).

EFFECTS OF THE ACTION

Direct effects

Potential sources of direct impact to the piping plover and red knot from the proposed oil and gas activities are habitat loss and fragmentation, disturbance from aircraft and boat vessel traffic, effects from trash and debris, and OCS-related air emissions. Pipeline landfalls, terminals, and other onshore OCS-related infrastructure can destroy or fragment otherwise suitable piping plover and red knot habitat. These activities should only have a minimal effect on those species and piping plover critical habitat because the range of these species encompasses an area where new construction is not anticipated. In addition, new pipelines that go ashore require an environmental analysis before approval. Where pipeline landfall occurs, the permitting process encourages the use of directional boring to greatly reduce and potentially eliminate impacts to barrier beaches. Long-term impacts or impacts significantly affecting the ecological function of coastal beaches as a result of pipeline crossings are, therefore, not expected to occur.

Low-altitude aircraft overflights related to OCS oil and gas operations could affect the piping plover and red knot. Those species may be susceptible to disturbance by low-altitude aircraft during foraging and resting periods. Those birds may leave and cease using their preferred areas and possibly seek less desirable ones, resulting in decreased nest success, increased energy expenditure via flight and alertness, and reduced energy intake via lower feeding rates. The Service, FAA, NPS, and BLM have an Interagency Agreement to reduce low-level flights over natural resource areas. The recommended minimum flight altitude is 2,000 feet above ground level. This limitation is included on aeronautical maps. The FAA (FAA Advisory Circular 91-36C) and corporate helicopter policy also states that helicopters must maintain a minimum altitude of 700 feet while in transit offshore and 500 feet while working between platforms. When flying over land, the specified minimum altitude is 1,000 feet over unpopulated areas or across coastlines and 2,000 feet over populated areas and biologically sensitive areas such as wildlife refuges and national parks. Service vessels, in support of OCS-related activities, are expected to use selected nearshore waters and existing coastal navigation waterways. Those activities are not anticipated to significantly increase the amount of existing routine helicopter and vessel traffic within the action area. Impacts to the piping plover and red knot from helicopter and vessel traffic should, therefore, be minimal.

There are numerous existing laws, regulations, and enforcement guidelines that prohibit and discourage the disposal of solid debris in Gulf waters that can impact listed species and their critical habitats. For example, BSEE prohibits the disposal of equipment, containers, and other materials into offshore waters by lessees (30 CFR 250.300). Also, BSEE NTL No. 2015-G03 requires annual awareness training and the posting of placards to minimize the unintentional loss of debris from industry structures or vessels. BSEE inspectors routinely conduct site visits and issue citations for noncompliance. In addition, MARPOL, Annex V. Public Law 100-220 (101 Statute 1458), which prohibits the disposal of any plastics, garbage, and other solid wastes at sea or in coastal waters, went into effect January 1, 1989, and is enforced by the USCG. The MDRPR (P .L 109-449) was enacted in December 2006. The purposes of the MDRPR are to help identify, determine sources of, assess, reduce and prevent marine debris and its adverse impacts on the marine environment and navigation safety; to reactivate the Interagency Marine Debris Coordinating Committee; and to develop a Federal marine debris information clearinghouse. The MDRPR established, within NOAA, a Marine Debris Prevention and Removal Program to reduce and prevent the occurrence and adverse impacts of marine debris on the marine environment and navigation safety. Greatly improved handling of waste and trash by industry, along with the annual awareness training required by the marine debris mitigations, is decreasing OCS-related debris in the ocean and impacts to the piping plover and red knot are, therefore, expected to be negligible.

BOEM/BSEE anticipates minimal effects to air quality associated with OCS oil and gas emissions due to prevailing atmospheric conditions, emission heights and rates, and pollutant concentrations. Because emissions from OCS-related activities are not likely to impact ambient air quality, effects to the piping plover and red knot from decreased air quality are expected to be negligible.

Indirect effects

A potential source of indirect effects to the piping plover and red knot would be caused by fouling of wintering habitat from oil spills. Oil spills represent the greatest potential impact to piping plovers and red knots. These species may be among the more vulnerable species because they forage in intertidal areas.

Birds that are heavily oiled succumb to acute toxicity effects shortly after exposure (Clark 1984; Leighton 1993). If the physical oiling of birds occurs, some degree of both acute and chronic physiological stress associated with direct and secondary physiological stress associated with direct and secondary uptake of oil would be expected. Lightly oiled birds can sustain tissue and organ damage from oil ingested during feeding and grooming or from oil that is inhaled. Birds that are heavily oiled usually die. Even low levels of oil may have multiple deleterious effects, including the following: changes in behavior; interference with feeding drive and food detection; alteration of food preferences and ability to discriminate between poor versus ideal food items; predator detection and avoidance; and definition and defense of feeding territories.

Residual material that remains after evaporation and solubilization are water-in-oil emulsions (mousse), which are the primary pollutant onshore after oil from offshore spills actually reaches land. The mixing of mousse and sediments form aggregates that have the odor of oil and, after photo- and biological oxidation, form asphaltic “tarballs” and pavements (Briggs et al. 1996). Mousse emulsions may be the most toxic petroleum component because they are the most hydrophobic and will penetrate the hydrophobic core of the plasma membrane of cells and will cause disruption of the membrane and enter the cells as well (Briggs et al. 1996 and 1997). Common symptoms of exposed birds include dehydration, gastrointestinal problems, infections, arthritis, pneumonia, hemolytic anemias, cloacal impaction, and eye irritation.

When oil gets into vegetated or unvegetated sediment, low redox potentials, absence of light, and waterlogged substrate may result in oil that can neither be oxidized by bacteria and sunlight nor evaporate. The oil may also remain in its unweathered toxic state indefinitely; however, weathering-related effects on the oil from its path offshore to the coast ameliorates, to some extent, toxicity at the shoreline.

Under natural conditions, water does not penetrate through the vanes of feathers because air is present in the tiny pores in the lattice structure of the feather vane. Oil, with its reduced surface tension and hydrophobic characteristics, adheres to keratin and mats the feather barbules into clumps; the lattice opens up (breaks down) and water penetrates and displaces insulating air (Lambert et al. 1982; O’Hara and Morandin 2010). Oil also mats the feathers together, displacing insulating properties of trapped air (Jenssen 1994). Dispersants also reduce water surface tension in the feather lattice pores and render them water-attracting instead of water-repelling (Stephenson 1997; Stephenson and Andrews 1997). Thus, at a certain surface tension, water will penetrate the feathers and death from reduced thermoregulatory function may result (Lambert et al. 1982; Stephenson 1997; Stephenson and Andrews 1997).

Oil spill response activities also have to potential to impact wintering piping plovers and red knots. These activities can cause habitat damage, cause them to abandon otherwise preferred habitats, negatively affect the birds' energy balances, and reduce the amount of available prey. Timing (i.e., if peak periods in bird density overlap temporally with the spill; Fraser et al. 2006), location (high versus low bird density area), wind conditions, wave action, and distance to the shore may have a greater overall effect on bird mortality than spill volume and fluid type (Wilhelm et al. 2007; Castège et al. 2007; Byrd et al. 2009).

Rehabilitation of oiled birds may also be included in the oil spill response. Research on long-term survival and reproduction of rehabilitated, oiled birds is limited, and results to date are mixed (Anderson et al. 1996; Sharp 1996; Anderson and Labelle 2000; Golightly et al. 2002; Mazet et al. 2002, Underhill et al. 1999). Success of rehabilitation for oiled birds may be a function of capture and handling methods, overall oiling and exposure of the individual, facility design, and availability of food, water, and space while in captivity, as well as species-specific characteristics including body size, metabolism, and resting heart rate. It is critical that rehabilitated birds remain disease free while in captivity. A major concern for holding birds in facilities post-spill is the potential to expose the wild population to diseases once rehabilitated individuals are released.

Indirect impacts to piping plover critical habitat could result from contact by spilled oil if a spill were to occur. As discussed above, there is a low probability of oil, spilled as a result of the proposed action, contacting beaches where critical habitat has been designated. Should a spill contact a barrier beach, however, oiling is expected to be light and sand removal during cleanup activities will likely be minimized. No significant impacts to the physical shape and structure of barrier beaches and associated dunes are expected to occur as a result of a proposed action. Because their winter habitat is located in and adjacent to high-energy beaches, it is assumed that complete recovery of those habitats which become oiled would begin to occur within 1 to 2 years. Recovery time could vary depending on the severity of the spill, time of year, and cleanup methods used.

According to the OSRA, there is a 4 to 8 percent probability that an oil spill >1,000 barrels would occur and contact piping plover and red knot wintering habitats within 10 days in the CPA (note again that those probabilities do not include clean-up activities and natural weathering of the spill). That analysis also indicated that there is a 1 to 2 percent probability that an oil spill >1,000 barrels would occur and contact piping plover and red knot wintering habitats within 10 days in the WPA and a <0.5 percent probability in the EPA. The BOEM/BSEE, USEPA, and USCG have regulations, requirements, and recommendations to prevent or reduce the likelihood of a spill occurring and prevent or reduce impacts to piping plovers and red knots if a spill occurs. Those measures, and the weathering of oil in the environment, should significantly minimize potential impacts on wintering piping plovers and red knots if a spill occurs.

Effects summary

Activities occurring as a result of the proposed action to lease OCS submerged lands for oil and gas exploration, development, production and transportation may affect piping plovers and red knots. It is expected that the majority of the effects from the major-impact producing factors would be sublethal, causing discountable or insignificant effects. Some changes in local population numbers and distribution of wintering birds are assumed as a result of all of these factors; however, it is unknown what significance those changes could have overall on wintering piping plovers and red knots. No direct loss or permanent modification of piping plover critical habitat or its ecological function is anticipated as a result of the proposed action. The greatest potential concern is the threat of an oil spill reaching piping plover and red knot wintering habitat. The probabilities developed by BOEM/BSEE of an oil spill occurring and contacting habitat (including critical habitat) where piping plovers occur are low. They also overestimate contact probability because they do not account for naturally occurring events such as weathering, and activities included in the proposed action (e.g., clean up, containment, etc.). Although the reduction in those probabilities could not be quantified, it is the Service's belief that those reductions make the likelihood of contact extremely low. In addition, because contact with habitat does not necessarily mean contact with individual birds, the likelihood of take is not reasonably certain to occur. If a spill were to occur, the adverse effects that might occur to piping plover critical habitat would be temporary in nature and are of low probability. In addition, such potential adverse effects to the area of critical habitat most likely to be affected would not appreciably diminish the value of the entire designated critical habitat area in providing for either the long-term survival or the recovery of the species.

As discussed earlier, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to red knots or piping plovers associated with a spill of this magnitude are not addressed in this BO.

STATUS OF THE SPECIES/CRITICAL HABITAT

Whooping crane (*Grus americana*)

Species/critical habitat description

The whooping crane was listed by the Service as a federally endangered species on March 11, 1967. Critical habitat was designated for whooping cranes in 1978, at nine sites in seven states. Those sites are: 1) Monte Vista NWR, Colorado; 2) Alamosa NWR, Colorado; 3) Grays Lake NWR and vicinity, Idaho; 4) Cheyenne Bottoms State Waterfowl Management Area, Kansas; 5) Quivira National Wildlife Refuge, Kansas; 6) the Platte River bottoms between Lexington and Dehman, Nebraska; 7) Bosque del Apache NWR, New Mexico; 8) Salt Plains NWR, Oklahoma; and 9) Aransas NWR (ANWR) and vicinity, Texas.

The designated critical habitat in Texas is found in Aransas, Calhoun and Matagorda Counties within the ANWR and adjacent lands and waters. That area constitutes the entire wintering range of the whooping crane's reproducing wild population.

The whooping crane is the tallest North American bird and has a life span of 22 to 24 years. Males, which may approach 4.9 feet in height, are larger than females. Adults are white except for black primary feathers on the wings and a bare red face and crown. The bill is a dark olive-gray, which becomes lighter during the breeding season. The eyes are yellow and the legs and feet are gray-black. Immature cranes are a reddish cinnamon color that results in a mottled appearance as the white feather bases extend. The juvenile plumage is gradually replaced through the winter months and becomes predominantly white by the following spring as the dark red crown and face appear. Yearlings achieve the typical adult appearance by late in their second summer or fall. Whooping cranes are omnivorous feeders. They feed on insects, frogs, rodents, small birds, minnows, and berries in the summer. In the winter, they focus on predominantly animal foods, especially blue crabs and clams. They also, forage for acorns, snails, crayfish and insects in upland areas.

Life history

Whooping cranes are monogamous and form life-long pair bonds but will remate following the death of a mate. Whooping cranes return to the same breeding territory in Wood Buffalo National Park, Canada, in April and nest in the same general area each year. They construct nests of bulrush and lay one to three eggs (usually two) in late April and early May. The incubation period is about 29 to 31 days. Whooping cranes will renest if the first clutch is lost or destroyed before mid-incubation. Both sexes share incubation and brood-rearing duties. Despite the fact that most pairs lay two eggs, seldom does more than one chick reach fledging. Autumn migration begins in mid-September, and most birds arrive on the wintering grounds of ANWR on the Texas Gulf coast by late October to mid-November. Whooping cranes migrate singly, in pairs, in family groups or in small flocks, and are sometimes accompanied by sandhill cranes. They are diurnal migrants, stopping regularly to rest and feed. On the wintering grounds, pairs and family groups occupy and defend territories. Subadults and unpaired adult whooping cranes form separate flocks that use the same habitat but remain outside occupied territories. Subadults tend to winter in the area where they were raised their first year, and paired cranes often locate their first winter territories near their parents' winter territory. Spring migration is preceded by dancing, unison calling, and frequent flying. Family groups and pairs are the first to leave the refuge in late March to mid-April.

Juveniles and subadults return to the summer breeding grounds in the vicinity of their natal area, but are chased away by the adults during migration or shortly after arrival on the breeding grounds. Only one out of four hatched chicks survives to reach the wintering grounds. Whooping cranes generally do not produce fertile eggs until age four.

Population dynamics

In the mid 1800's, as many as 1,400 whooping cranes migrated across North America; however, by the late 1930s, the ANWR population had declined to 18 birds. Since then the population has slowly increased due to conservation efforts. By 1986 the population reached 110 birds, and 18 years later the population had reached 217 (Canadian Wildlife Service and Service 2007). In December of 2014, the population at ANWR was estimated to be 308 (Canadian Wildlife Service and Service 2015). Four geographically distinct populations exist: migratory populations moving between Canada and Texas on or near ANWR and between Wisconsin and Florida, and non-migratory flocks in the Kissimmee Prairie of Florida and in southwestern Louisiana. In 2014, a total of 451 wild whooping cranes were known to be in existence, with an additional 161 whooping cranes held in captivity (Canadian Wildlife Service and Service 2015).

The long-term recruitment rate of the whooping crane is 13.9% which is the highest of any other crane population in North America (Drewien et al. 1995). Since the late 1930's, the whooping crane population has been increasing at an average annual rate of more than 4% (Canadian Wildlife Service and Service 2007). Studies indicate that the whooping crane has a 10-year cycle in survivorship and has been correlated with that of boreal forest predator cycles by some researchers (Boyce and Miller 1985, Boyce 1987, Nedelman et al. 1987, Canadian Wildlife Service and Service 2007). For example, the whooping crane population increased from 75 to a high of 146 birds from 1983 to 1989, and then dropped to a 10-year low of 132 individuals in the winter of 1991-1992 (Canadian Wildlife Service and Service 2007).

Status and distribution

Reason for Listing

Whooping cranes are the rarest of the 15 species of cranes in the world (Ricketts et al. 2005). It has been estimated that over 10,000 whooping cranes were present in precolonial times prior to extensive human expansion and alteration of wetland habitats. Historically, growth of human populations in North America resulted in significant whooping crane habitat alteration and destruction. Whooping cranes declined or disappeared as agriculture claimed the northern Great Plains of the U.S. and Canada and destroyed wetland habitats (Allen 1952). Conversion of potholes and prairie to hay and grain production made much of the historic nesting and migration habitat unsuitable for whooping cranes. (Canadian Wildlife Service and Service 2007). Disruptive practices included draining, fencing, sowing, and the human activity associated with these actions. The shooting of hundreds of whooping cranes was documented by Allen (1952). By the mid-1900s, only one small nesting population survived in the wilderness in Wood Buffalo National Park. The species declined to an all-time low of just 15 birds in the Aransas-Wood Buffalo Population (AWBP) in 1941 (Canadian Wildlife Service and Service 2007, Stehn 2006).

The Whooping Crane Recovery Plan (2007) lists the following as current threats and reasons for listing: human settlement/development, insufficient freshwater inflows, shooting, disturbance, disease/parasites, predation, food availability/sibling aggression, severe weather, loss of genetic diversity, climate change, red tide, chemical spills, collisions with power lines, fences, and other structures, collisions with aircraft and pesticides.

Range-wide trend

The present form of the whooping crane appears to be the same as fossilized remains from the Upper Pliocene in Idaho (Miller 1944, Feduccia 1967), and from the Pleistocene in California, Kansas, and Florida (Wetmore 1931, 1956). The historical range extended from the Arctic coast south to central Mexico, and from Utah east to New Jersey, South Carolina, Georgia, and Florida (Allen 1952, Nesbitt 1982). Distribution of these fossil remains suggests a wider whooping crane distribution during the Pleistocene.

The major nesting area during the 19th and 20th centuries extended from Illinois, Iowa, Minnesota, and North Dakota to southwestern Manitoba, Saskatchewan and into east central Alberta. Some nesting apparently occurred at other sites such as Wyoming in the 1900's, but documentation is limited (Allen 1952). Allen (1952) believed the whooping cranes' principal wintering range was the tall grass prairies, in Louisiana, along the coast of Texas, and near the Rio Grande Delta in Mexico. Other significant wintering areas were the interior tablelands in west Texas and the high plateaus of central Mexico, where whooping cranes occurred among thousands of sandhill cranes (Canadian Wildlife Service 2007).

Whooping cranes currently exist in the wild at four locations and in captivity at thirteen facilities. In December 2014, the total whooping crane population in the wild was estimated at 451 individuals, an increase of about 43 percent in 11 years (315 were recorded in December 2003). This includes 308 individuals in the only self-sustaining wild flock from AWBP, approximately 11 individuals in a non-migratory Florida Population (FP), 93 individuals (introduced starting in 2001) that migrate between Wisconsin and Florida in an eastern migratory population (EMP), and 39 captive-raised birds surviving from annual releases at White Lake, Louisiana that started in February 2011. The December 2014 captive population of whooping cranes contained 161 birds, an increase of about 35 percent in eleven years (119 were reported in December 2003) with annual hatchling production from the Calgary Zoo, International Crane Foundation, Patuxent Wildlife Research Center (PWRC), Audubon Species Survival Center and the San Antonio Zoo. The total population, wild and captive, in December 2014 was 612 individual birds (Canadian Wildlife Service and Service 2015).

Four projects to re-introduce whooping cranes into the wild show increased promise but to date remain unsuccessful in establishing a self-sustaining population. From 1975-1989, whooping cranes cross-fostered at Graythanks Lake NWR, Idaho by placing whooping crane eggs into sandhill nests failed to pair up and breed due to an apparent imprinting problem. Captive-raised whooping cranes placed in Florida from 1993-2005 suffered high mortality, low productivity, and habitat loss; and efforts to establish this non-migratory flock were discontinued. Starting in 2001, captive-raised whooping cranes were placed at Necedah NWR in central Wisconsin and taught a migration to Florida by following ultra-light aircraft. Mortality for this project has been reasonable, but production has been extremely low. The cranes are pairing up and nesting, but nest abandonment is occurring which may be due to swarms of biting black flies. Efforts are being made to move the reintroduction into other parts of Wisconsin where black flies are less prevalent. Since 2011, annual cohorts of captive-raised whooping cranes have been released at White Lake Wetlands Conservation Area. Cohort sizes were 10-16 juvenile whooping cranes per year. Cohort size increased to a maximum of 35 juveniles per year with the addition of a second release area at Rockefeller Wildlife Refuge in 2015. As of December 2017, 125 whooping cranes have been released in Louisiana. The current population size as of February 18, 2019, is 69 individuals.

Whooping cranes are currently listed as endangered (32 FR 4001, 1967 March 11) except where NEPs exist (66 FR 33903-33917, 2001 June 26; 62 FR 38932-38939, 1997 July 21; and 58 FR 5647-5658, 1993 January 22; 76 FR 6066 6082, 2011 February 3) in 18 eastern states including the reintroduced population that migrates between Wisconsin and Florida. Critical habitat was designated in 1978 in the U.S Federal Register (Vol. 43, Number 94) at 9 sites in 7 states. In the U.S., the whooping crane was listed as threatened with extinction in 1967 (Fed. Reg. Vol. 32, Number 48, March 11), and as Endangered in 1970 (Fed. Reg. Vol. 35, Number 199, October 13). Both of these listings were "grandfathered" into the Act (U.S.C., 1531-1 543; 87 Stat. 884), which resulted in the establishment of the U.S. Whooping Crane Recovery Team and facilitated further conservation actions on behalf of the species. In Canada, the whooping crane was designated as endangered in 1978 by the Committee on the Status of Endangered Wildlife in Canada and listed as endangered under the SARA in 2003.

For the whooping crane to be reclassified to threatened (downlisted), the March 2007 International Recovery Plan for the Whooping Crane has set forth 2 primary objectives and measurable criteria that would have to be met. The first objective involves establishing and maintaining, for at least ten years prior to downlisting, self-sustaining populations of whooping cranes in the wild while the second objective involves maintaining a genetically stable captive population. If additional wild sustaining populations are not established, then the AWBP must remain above 1,000 individuals (250 productive pairs) for downlisting to occur. During the 2014 to 2015 wintering season, 308 whooping cranes were verified in the Texas wintering area.

New Threats

Many future threats to this species' continued existence, both natural and human related, are expected to impact summer, migration, and winter habitats. Threats mentioned in this section and/or listed in the Recovery Plan include loss of habitat, water diversions in rivers degrading habitat, reduced inflows to wintering area, erosion of winter habitat, increased development of shorelines and wetlands, collisions with power lines, fences, cell towers and wind turbines, collisions with aircraft, chemical spills, loss of genetics, disease, red tide, pesticides, increased human recreational pressures, increased human and domestic animal disturbance, shootings, sea level rise, climate change, and greater establishment of black mangrove into the winter range. The young chicks also face many hazards including predation, disease and sibling aggression (Stehn 2006).

Whooping cranes are faced with various natural obstacles and problems during their annual 2,400 mile migration. Snow and hail storms, low temperatures, and drought can present navigational handicaps or reduce food availability. Migrating cranes are also exposed to a variety of physical hazards such as collision with obstructions such as power lines, predation of young cranes by bobcats, disease and illegal shootings (Canadian Wildlife Service and Service 2007). Collisions with power lines have accounted for the death or serious injury of at least 49 whooping cranes since 1956 (Stehn 2011 pers. comm.). Hurricanes and drought can also create problems on the wintering grounds. The active portion of the hurricane season usually ends by October 31, before most whooping cranes arrive. A late season hurricane could place the cranes at risk due to high wind velocities and storm surge. Drought also influences availability and abundance of the natural food supply by altering salinity of tidal basins and estuaries, and the cranes are forced to move to less than optimal uplands to forage for food and find freshwater (Blankinship 1976).

Currently, expanding human populations throughout the range of the whooping cranes continue to threaten survival and recovery of the birds. Impacts are particularly severe on the winter grounds. A major threat to the whooping crane is the decrease in the suitability of the species' habitat due to accelerating development within and adjacent to the designated critical habitat in

Texas. In addition, human population growth along the U.S. coast creates an ever increasing demand for recreation in wetlands and tidal areas for fishing, crabbing, boating and hunting.

Freshwater inflows starting hundreds of miles inland primarily from the Guadalupe and San Antonio rivers that flow into whooping crane critical habitat at ANWR are needed to maintain the proper salinity gradients, nutrient loadings, and sediments that produce an ecologically healthy estuary (Texas Parks and Wildlife Department (TPWD) 1998). Inflows are essential to maintain the productivity of coastal waters and produce foods used by the whooping cranes. Coastal water with low saline levels that whooping cranes can drink rather than fly inland for freshwater are maintained by these freshwater in-stream flows. Upstream reservoir construction and water diversions for agriculture and human use reduce these inflows. In a report entitled Bays in Peril, a "Danger" ranking was given to San Antonio Bay because drought periods were predicted to increase by 250%, and years with low freshwater pulses in the spring were calculated to increase 26% from naturalized levels (National Wildlife Federation 2004). TPWD has made recommendations for target inflows needed to maintain the unique biological communities of the Guadalupe River estuary (TPWD 1998) that flows into whooping crane critical habitat. A simple inverse relationship exists between blue crab catch rates and mean salinity within an estuary (Longley 1994). By 2040, due to constructed diversions, a decrease of freshwater inflows into the crane's winter range is projected in an average year to cause an 8% decline in blue crab populations, the primary food of the whooping crane (Texas Department of Water Resources 1980). Inflows are already presumed to be insufficient and significantly reduced over historic levels. With projected losses, freshwater inflows would be insufficient to sustain the ecosystem in an average rainfall year. Long before the ecosystem collapsed for lack of inflows, significant adverse impacts to the primary winter food supply of the whooping crane would occur (Kretzschmar 1990). Texas Water Development Board data indicate natural droughts already threaten the Guadalupe ecosystem. Withdrawals of surface and groundwater for municipal and industrial growth will leave insufficient inflows to sustain the ecosystem in less than 50 years. The state water plan proposes a diversion at the mouth of the Guadalupe River, pumping at least 94,500 acre-feet annually back to San Antonio for municipal use (Canadian Wildlife Service and Service 2007). Recently, a nuclear power plant has been proposed near Victoria, Texas, which also may divert freshwater resources and the need to discharge contaminants that could affect whooping crane habitat quality.

Commercial vessels carrying dangerous, toxic chemicals travel the Gulf Intracoastal Waterway (GIWW) daily through the heart of whooping crane winter habitat. A spill or leak of these substances could contaminate or kill the cranes' food supply, or poison the cranes (Robertson et al. 1993). Spills that occur in summer, when whooping cranes are absent, could adversely affect survival by reducing productivity of the environment or leaving a toxic residue. Gulf Engineers and Consultants, Inc. (1992, as cited by Canadian Wildlife Service and Service 2007) assessed threats to the whooping crane and its habitat from spills of vessel fuels and cargoes. They concluded that the hazard of spill exists, but the probability of occurrence is low. Low probability events, such as a catastrophic spill, are difficult to predict. It is impossible to provide full protection for the cranes as long as chemicals are transported on the GIWW through the heart of winter range. Spills of hazardous chemicals may limit human approach to only those personnel wearing special protective suits, and breathing apparatus. Also, an event could occur at night or in bad weather and further slow response. Spill of gaseous materials could directly impact any cranes downwind. High winds greatly reduce the effectiveness of containment booms for products floating on the surface. If crane habitat becomes contaminated, attempts would be made to haze cranes, away from the spill area and to capture individuals that become seriously contaminated (Canadian Wildlife Service and Service 2007). However, the response of whooping cranes to spilled materials, and to humans trying to haze the cranes away, is currently unknown. Adult cranes are territorial; therefore, it is likely not possible to haze them from their large territories. Oiled cranes would be captured when possible and cleaned, although wild cranes are very difficult to capture and susceptible to death from capture myopathy, especially when young. The only self-sustaining wild population remains vulnerable to destruction through a hurricane event or contaminant spill, due to oil and gas activity and transportation of chemical and petroleum products in critical habitat areas and through other areas utilized by wintering whooping cranes (Canadian Wildlife Service and Service 2007, Stehn 2006).

Global climate change may have numerous impacts to whooping cranes throughout the year (Chavez-Ramirez and Wehtje, In Press). The water regime of Wood Buffalo National Park may be severely affected, with potentially severe impacts on whooping crane reproduction (Stehn 2006). Permanently lowered water tables, for example, would shrink wetlands, reduce the availability of quality nesting sites, reduce invertebrate food availability, and allow predators to access nests and young. A drying trend forecast for the Great Plains may reduce and degrade the amount of migration habitat available. At ANWR, the predicted warmer winters may allow black mangrove to extend its range northward into the crane area, shading out desirable crane forage plants including Carolina wolfberry. On the wintering area, a reduction in rainfall would reduce inflows and reduce the blue crab population that is the cranes' primarily source of prey.

Sea level rise combined with land subsidence are projected to be about 17 inches on the Texas coast over the next 100 years (Twilley et al. 2001). This would reduce suitability of salt marsh and open water areas, making much of the present acreage too deep for use by whooping cranes (Stehn, personal communication 2006).

A catastrophic event could eliminate the wild, self-sustaining AWBP because this population is characterized by low numbers of individuals, slow reproductive potential, and limited genetic diversity. Therefore, the recovery strategy as stated in the Recovery Plan includes protection and enhancement of the breeding, migration, and wintering habitat to allow the AWBP wild flock to grow and reach ecological and genetic stability (Canadian Wildlife Service and Service 2007). The numerical population criteria for the species (1000 individuals) can only be achieved if threats to the species' existence are sufficiently reduced or removed (Canadian Wildlife Service and Service 2007).

Analysis of the species/critical habitat likely to be affected

The whooping crane and its critical habitat may be affected by the proposed action. The effects of the proposed action on that species will be considered further in the remaining sections of this BO. Potential sources of impacts to this species from existing and proposed oil and gas activities are habitat loss and fragmentation, disturbance from aircraft and boat vessel traffic, effects from trash and debris, and OCS-related air emissions.

ENVIRONMENTAL BASELINE

Status of the species within the action area

The ANWR located in Aransas, Calhoun, and Refugio counties is the wintering home of the last remaining wild migratory flock of whooping cranes. Their winter range includes the Aransas and Matagorda Island refuges and surrounding areas and stretches over 35 miles along the Texas coast (Service-ANWR 2005).

In December 1993, the wild population of the endangered whooping crane was estimated at 160 individuals (Service 1994). In the spring of 2002 the size of the Aransas-Wood Buffalo population was estimated at 174 individuals (Canadian Wildlife Service and Service 2007). In 2006, the AWBP population had grown to approximately 220 (72 FR 29544, May 29, 2007). As of December 2014, through the implementation of the Act, aggressive conservation efforts, and preservation of habitat, the migratory AWBP whooping crane population consisted of approximately 308 birds total.

Designated critical habitat areas in Texas are found in Aransas, Calhoun and Matagorda Counties within the Aransas NWR and adjacent lands and waters. This area constitutes the entire wintering range of the whooping crane's reproducing wild population. In March, the whooping cranes leave their Texas wintering grounds and return to their nesting grounds in Canada.

Factors affecting species environment within the action area

The greatest threats to the wild population of whooping cranes occur at their wintering habitat in Aransas; those threats include the potential of a hurricane or contaminant spill destroying their wintering habitat on the Texas coast, and impacts to the inflow of fresh water into whooping crane critical habitat. Data indicate that, while on its wintering grounds in Texas, the health and survival of the entire whooping crane flock is directly related to freshwater inflows. Those inflows are strongly tied to blue crab populations; blue crabs are the whooping cranes' primary winter food source. Collisions with power lines and fences are known hazards to wild whooping cranes.

EFFECTS OF THE ACTION

Direct effects

Potential sources of direct impact to the whooping crane from the proposed oil and gas activities are habitat loss and fragmentation, disturbance from aircraft and boat vessel traffic, effects from trash and debris, and OCS-related air emissions. New pipeline construction should only have a minimal effect on whooping cranes and their critical habitat because the range of this species encompasses an area where new pipeline construction is not anticipated. In addition, new pipelines that go ashore require an environmental analysis before approval. Where pipeline landfall occurs, the permitting process encourages the development of measures to minimize and potentially eliminate impacts to federally listed species.

Low-altitude aircraft overflights related to OCS oil and gas operations could affect the whooping crane. Whooping cranes may be susceptible to disturbance by low-altitude aircraft during nesting, foraging and resting periods. Those birds may leave and cease using their preferred nesting and feeding areas and possibly seek less desirable ones, resulting in decreased nest success, increased energy expenditure via flight and alertness, and reduced energy intake via lower feeding rates. The Service, FAA, NPS, and BLM have an Interagency Agreement to reduce low-level flights over natural resource areas. The recommended minimum flight altitude is 2,000 feet above ground level. This limitation is included on aeronautical maps. The FAA (FAA Advisory Circular 91-36C) and corporate helicopter policy also states that helicopters must maintain a minimum altitude of 700 feet while in transit offshore and 500 feet while working between platforms. When flying over land, the specified minimum altitude is 1,000 feet over unpopulated areas or across coastlines and 2,000 feet over populated areas and biologically sensitive areas such as wildlife refuges and national parks. Within the WPA, BOEM/BSEE anticipates 594,500-1,112,500 helicopter trips annually. Service vessels, in support of OCS-related activities, are expected to use selected nearshore waters and existing coastal navigation waterways. Those activities are not anticipated to significantly increase the amount of existing routine helicopter and vessel traffic within the WPA. Impacts to the whooping crane from helicopter and vessel traffic should, therefore, be minimal.

There are numerous existing laws, regulations, and enforcement guidelines that prohibit and discourage the disposal of solid debris in Gulf waters that can impact listed species and their critical habitats. For example, BSEE prohibits the disposal of equipment, containers, and other materials into offshore waters by lessees (30 CFR 250.300). Also, BSEE NTL No. 2015-G03 requires annual awareness training and the posting of placards to minimize the unintentional loss of debris from industry structures or vessels. BSEE inspectors routinely conduct site visits and issue citations for noncompliance. In addition, MARPOL, Annex V. Public Law 100-220 (101 Statute 1458), which prohibits the disposal of any plastics, garbage, and other solid wastes at sea or in coastal waters, went into effect January 1, 1989, and is enforced by the USCG. The MDRPR (P .L 109-449) was enacted in December 2006. The purposes of the MDRPR are to help identify, determine sources of, assess, reduce and prevent marine debris and its adverse impacts on the marine environment and navigation safety; to reactivate the Interagency Marine Debris Coordinating Committee; and to develop a Federal marine debris information clearinghouse. The MDRPR established, within NOAA, a Marine Debris Prevention and Removal Program to reduce and prevent the occurrence and adverse impacts of marine debris on the marine environment and navigation safety. Greatly improved handling of waste and trash by industry, along with the annual awareness training required by the marine debris mitigations, is decreasing OCS-related debris in the ocean and impacts to the whooping crane are, therefore, expected to be negligible.

BOEM/BSEE anticipates minimal effects to air quality associated with OCS oil and gas emissions due to prevailing atmospheric conditions, emission heights and rates, and pollutant concentrations. Because emissions from OCS-related activities are not likely to impact ambient air quality effects to the whooping crane from decreased air quality are expected to be negligible.

Indirect effects

As stated above, freshwater inflows starting hundreds of miles inland primarily from the Guadalupe and San Antonio rivers that flow into whooping crane critical habitat at ANWR are needed to maintain the proper salinity gradients, nutrient loadings, and sediments that produce an ecologically healthy estuary (Texas Parks and Wildlife Department (TPWD) 1998). Inflows are essential to maintain the productivity of coastal waters and produce foods used by the whooping cranes. Should oiled waters extend into these important whooping crane feeding areas between November and late April, when the whooping cranes are on their wintering grounds, the likelihood of significant impacts would be increased. Ingestion of oil could occur during the feeding process. Some oiling may occur through direct contact with oiled sediments or waves in the splash zone.

According to OSRA, however, there is less than a 0.5 percent probability that an oil spill > 1,000 barrels would occur and contact whooping crane habitat (including critical habitat) within 10 days in the WPA (note again that those probabilities due not include clean-up activities and natural weathering of the spill). This wintering habitat is protected to some extent from oil spills in the open Gulf by barrier islands, but the loss of even a relatively small portion of the Aransas-Wood Buffalo population could cause serious delays in the recovery of the species.

The BOEM/BSEE, USEPA, and USCG have regulations, requirements, and recommendations that should prevent or reduce the likelihood of a spill occurring and prevent or reduce impacts to whooping cranes if a spill occurs. Those measures, and the weathering of oil in the environment, should significantly minimize potential impacts on whooping cranes if a spill occurs.

Effects summary

Activities resulting from the proposed action to lease OCS lands for oil and gas exploration, development, production and transportation may affect whooping cranes. No direct loss of whooping crane wintering habitat, however, is anticipated. It is expected that the majority of the effects from the major-impact producing factors are sublethal, causing discountable or insignificant effects. Some changes in local population numbers and distribution of wintering birds are expected as a result of all of these factors; however, it is unknown what significance those changes would have overall on wintering whooping cranes. The greatest concern is the threat of an oil spill reaching whooping crane wintering habitats. The probabilities, developed by BOEM/BSEE, of an oil spill occurring and contacting areas where whooping cranes or their critical habitat occur are low. They also over-estimate contact probability because they do not account for naturally occurring events such as weathering and activities included in the proposed action (e.g., clean up, containment, etc.). Although the reduction in those probabilities could not be quantified, it is the Service's belief that those reductions make the likelihood of contact extremely low, but not zero.

As discussed earlier, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to whooping cranes associated with a spill of this magnitude are not addressed in this BO.

STATUS OF THE SPECIES/CRITICAL HABITAT

Alabama beach mouse (*Peromyscus polionotus ammobates*)
Choctawhatchee beach mouse (*Peromyscus polionotus allophrys*)
Perdido Key beach mouse (*Peromyscus polionotus trissyllepsis*)
St. Andrew beach mouse (*Peromyscus polionotus peninsularis*)

This section of the biological opinion will address the above identified beach mice collectively because: 1) they are closely related systematically, morphologically similar, and have essentially identical habits and life histories; 2) they occur in close proximity to each other; and 3) they face nearly identical threats to their continued survival.

Species/critical habitat description

All beach mice are characterized by white feet, large ears, and large black eyes (Hall 1981). The Alabama beach mouse has a head and body length of 2.7 to 3.5 inches, a tail length of 1.7 to 2.4 inches with upper parts pale gray with an indistinct middorsal stripe, with its sides and underparts white while its tail is white with an incomplete dorsal stripe.

The Choctawhatchee beach mouse is distinctly more orange-brown to yellow-brown than the other Gulf coast beach mouse subspecies (Bowen 1968). Pigmentation on the head either extends along the dorsal surface of the nose to the tip, or ends posterior to the eyes leaving the cheeks white. A dorsal tail stripe is either present or absent. Head and body length ranges from 2.7 to 3.5 inches (Holler 1992).

The Perdido Key beach mouse upper parts are colored grayish fawn to wood brown with a very pale yellow hue and an indistinct middorsal stripe. The white of the underparts reaches to the lower border of the eyes and ears, and the tail is white to pale grayish brown with no dorsal stripe. Its head and body length is 2.7 to 3.3 inches; the tail length is 1.7 to 2.5 inches.

The St. Andrew beach mouse’s fur is a pale, buff/brown color on its head and back with extensive pure white coloration on its underparts, sides, feet, face, and tail (Howell 1939). They have two distinct rump color patterns, tapered or squared (Bowen 1968). Their average size is: head and body length, 2.95 inches; tail length, 2.05 inches; and hind foot length, 0.73 inches (James 1992).

The Alabama, Choctawhatchee, and Perdido Key beach mice were federally listed as endangered on June 6, 1985. The St. Andrew beach mouse was federally listed as endangered on December 18, 1998. The mice are four of five subspecies of the old field mouse (*Peromyscus polionotus*) that occur on the Gulf coasts of Alabama and Florida (USFWS 1987, 1989a, 1989b). Critical habitat for the Alabama beach mouse has been designated on Fort Morgan, inholdings within Bon Secour NWR, and a portion of the Gulf State Park in Baldwin County, Alabama (Service 1987, 1989b). Critical habitat for the Choctawhatchee beach mouse occurs on four separate areas in Walton and Bay Counties, Florida. Those areas are: 1) Shell Island in Bay County, 2) St. Andrews State Recreation Area, mainland, west of the St. Andrew Bay inlet, in Bay County, 3) Grayton Beach State Recreation Area main unit, in Walton County, and 4) Topsail Hill State

Preserve in Walton County. Critical habitat for the Perdido Key beach mouse has been designated on five different areas/units in Perdido Key. Those areas are the: 1) Gulf State Unit in Baldwin County, Alabama, 2) West Perdido Key Unit in Baldwin County, Alabama and Escambia County, Florida, 3) Perdido Key State Park Unit in Escambia County, Florida, 4) Gulf Beach Unit in Escambia County, Florida, and 5) Gulf Islands National Seashore Unit in Escambia County, Florida (Service 1987, 1989b). Critical habitat for the St. Andrew beach mouse has been designated on three areas/units: 1) East Crooked Island Unit in Bay County, 2) Palm Point Unit in Gulf County, and St. Joseph Peninsula Unit in Gulf County, Florida.

The Alabama beach mouse presently survives on Bon Secour NWR and private lands on the Ft. Morgan peninsula west of Gulf Shores and Orange Beach, Alabama, but has been extirpated from most of its original range, including all of Ono Island (Holler and Rave 1991). As of May 2008, the Service estimated that the Alabama beach mouse current distribution is contained within 2,450 acres of frontal, tertiary and interior scrub habitat along an estimated 13 miles of Alabama coastline (Service 2009c).

The historic range of the Choctawhatchee beach mouse extended from the East Pass of Choctawhatchee Bay in Okaloosa County east through Walton County to Shell Island in Bay County, Florida. The Choctawhatchee beach mouse currently persists on three isolated areas along 12.1 miles of Gulf of Mexico beachfront. These areas are spread out along 53 miles of its historic range (60 miles). Additional available but unoccupied habitat exists on 2.9 miles.

The Perdido Key beach mouse is the most endangered of the five Gulf coast beach mouse subspecies and was once thought to inhabit the entire Perdido Key. Since 1987, all populations of the Perdido Key beach mouse at each of the three public lands within its range (Gulf State Park, Perdido Key State Park, and Gulf Islands National Seashore) have been extirpated at some time. Through translocation efforts, Perdido Key beach mice are currently present on all public land areas. Current distribution and densities of the Perdido Key beach mouse on adjacent private lands are unknown (Service 2014b).

The St. Andrew beach mouse is the eastern most beach mouse subspecies occurring along the northern Gulf coast (James 1992). Its historic range is defined as extending from the East Pass of St. Andrew Bay (Crooked Island) in Bay County, Florida, southward along the mainland coastline adjacent to St. Joseph Bay, to St. Joseph Peninsula and east to Money Bayou along the Gulf of Mexico in Gulf County, Florida (Bowen 1968, James 1992). Over the years, this subspecies' range decreased to only one known population (on St. Joseph Peninsula) by the early 1990s. This was approximately a 68% reduction in its historic distribution (63 FR 70053). Due to concerns for the subspecies, its range was expanded through reintroduction efforts during 1997 and 1998. This effort resulted in the establishment of an additional population on East Crooked Island, making a total of two known populations (Moyers et al. 1999, Lynn 2002, Moyers and Shea 2002). Surveys conducted on East Crooked Island between May 2005 and January 2007 found beach mice were also present on Tyndall Air Force Base property and also on adjacent private lands southeast of Tyndall Air Force Base property.

Life history

Beach mice habitats are restricted to the primary and secondary sand dunes and scrub dunes along the GOM coastline. Beach mice dig burrows mainly in the primary dunes and in other secondary and interior dunes where the vegetation provides suitable cover. Beach mice are nocturnal and forage for food throughout the dune system. They feed primarily on the seeds of beach grass and sea oats; however, recent food habit studies show that insects are an important component of their diet (Holler 1990, 1991; Service 1987, 1989b; J. Moyers 1996).

Based on studies of other subspecies of beach mice, it can be inferred that reproduction may occur throughout the year, but peaks during November, December, and January. Beach mice litters may range from two to seven, and the young may reach sexual maturity by 6 weeks of age. Results of a laboratory study (Blair 1948) show that female beach mice are capable of producing litters every 26 days, and they may produce 80 or more young in their lifetimes. Studies of other closely related beach mouse species indicate that life spans may be short, ranging from less than 5 months (in the wild) to more than 3 years (Blair 1948).

Population dynamics

There are few historical population estimates for the Alabama beach mouse. Holliman (1983) estimated that the entire population of Alabama beach mice numbered less than 900 individuals and occupied fewer than 350 acres of habitat. Since the late 1980s, more robust grid-based sampling has been conducted intermittently on various areas within the Bon Secour National Wildlife Refuge (BSNWR) by various researchers and the Service. Analysis of these long-term trapping data has shown that Alabama beach mice densities are cyclic and fluctuate by orders of magnitude on a seasonal and annual basis (Rave and Holler 1992, Holler et al. 1997, Swilling et al. 1998, Sneckenberger 2001). These population fluctuations can be a result of varying reproduction rates, food availability, habitat quality and quantity, catastrophic events, disease and predation pressures (Blair 1951, Bowen 1968, Smith 1971, Hill 1989, Rave and Holler 1992, Swilling et al. 1998, Swilling 2000, Sneckenberger 2001). Tropical cyclones and hurricanes have a profound impact on Alabama beach mice populations and habitat, as seen with Hurricanes Frederic (1979), Elena (1985), Opal (1995), Ivan (2004) and Katrina (2005) (Holliman 1983, Rave and Holler 1992, Swilling et al. 1998, Service 2004a and 2005, and Conroy and Runge 2008). Recently, there has been a slow expansion of occupied Alabama beach mouse habitat over the last three years as mice have begun to recolonize areas previously impacted by Hurricanes Ivan and Katrina (Danielson and Falcy 2008, Service 2008c). However, the estimated available Alabama beach mouse habitat has been declining due to habitat loss from coastal development and temporary habitat recovery delays from hurricane impacts since we began tracking in 2003 (Service 2003, 2005, 2008c). Trapping data suggest that Alabama beach mice were extirpated from Gulf State Park (GSP) and Laguna Key/West Beach after Hurricane Ivan (Volkert 2005), but in 2008 they began to recolonize Laguna Key/West Beach (Service 2009d). Natural recolonization at GSP is unlikely because it has become isolated from extant Alabama beach mouse populations to the west by high density development in Gulf Shores, and had not recolonized naturally following other extirpation events (Holliman 1983, Holler and Rave 1991, Service 2004a and 2005, Volkert 2005). Because of the dramatic fluctuations in local beach mouse populations (both seasonally and in response to tropical cyclones) and limited

Service access to privately held Alabama beach mouse habitat, generating robust population estimates with precise confidence levels is difficult at best.

Novak (1997) reported winter and early spring densities of 5.95 to 14.25 mice/acres for Choctawhatchee beach mice on Shell Island, Tyndall Air Force Base (AFB) in 1993 and 1994, prior to Hurricane Opal. Three years following the hurricane in 1998, based on Auburn unpublished data (1999), densities of Choctawhatchee beach mice were 0.72 mice/acre on Shell Island. Densities for Choctawhatchee beach mice on Grayton Beach SRA main unit from 1995 to 1999 (includes pre and post-Hurricane Opal) ranged from 0.29 to 1.36 mice/acre, based on Auburn unpublished data (1999), densities at Topsail Hill State Preserve from 1995 (post Hurricane Opal) to 1999, yield densities from 0.018 to 0.18 mice/acre.

Since its listing in 1985, Perdido Key beach mouse population estimates never reached more than 400 to 500 individuals until 2003. Before Hurricane Ivan (2004), trapping survey data led to a population estimate of 500 to 800 which was divided between two populations - the Johnson Beach Unit of Gulf Islands National Seashore and Perdido Key State Park (Service 2004b). The population of Perdido Key beach mouse at Gulf State Park-Florida Point was likely extirpated in 1999 (Moyers et al 1999). In October 2005, following the active hurricane seasons of 2004 and 2005, a trapping effort of less than one-third of the habitat available on public lands yielded captures of fewer than 30 individuals. Tracking data from June 2006 indicated that about 25 and 32 percent of the available habitat was occupied at Perdido Key State Park and Gulf Islands National Seashore, respectively (FWC 2007). Trapping at Perdido Key State Park in March 2007 was cancelled after two nights following the capture of only one mouse (a fatality) and very few sightings of beach mouse tracks or burrows (FWC 2007). Trapping conducted in April of 2008 was more encouraging with the capture of 35 mice at Gulf Islands National Seashore (Sneckenberger 2008 pers. comm.). However, no mice were captured on Perdido Key State Park (Loggins et al 2008). Tracking data from summer of 2009 suggested population abundance and distribution was increasing within Gulf Islands National Seashore and Perdido Key State Park (FWC 2010). Trapping at Gulf Islands National Seashore and Perdido Key State Park in spring 2010 generally confirmed this with Perdido Key beach mice widely distributed at both public lands. However, abundance at Gulf Islands National Seashore was lower than anticipated.

Extensive monitoring efforts at GSP during 2009 and early 2010 failed to show any presence of Perdido Key beach mice. In the spring of 2010, captive-born Perdido Key beach mice from Brevard and Palm Beach zoos were released at Gulf State Park. A total of 48 Perdido Key beach mice were released in the southwestern portion of Gulf State Park and 28 were fitted with radio transmitters. Within a few days, 15 of the transmitters were found in a red fox den. By the time two adults and five red fox pups were removed by USDA employees, only 13 mice remained. Monitoring continued daily for the life of the transmitters (3 weeks) and monthly tracking and periodic trapping continued over the summer and fall. A 3-day trapping effort at the end of September 2010 yielded 51 individual Perdido Key beach mice, including 8 of the originally released mice. Mice were found throughout habitat at Gulf State Park south of Highway 182 (FWC 2010). A 3-day trapping effort the week of May 7, 2012, continued to find Perdido Key beach mice distributed throughout habitat south of Highway 182. Two reproductively-active male Perdido Key beach mice were found north of Highway 182 (Gore pers. comm. 2012). According to current track tube data and recent limited trapping, the reintroduced population at

Gulf State Park is still present in 2014 and Perdido Key beach mice are occupying all three public lands for the first time since being listed as endangered in 1985.

Prior to the 1980s, there were two known populations of St. Andrew beach mice. One population was found on St. Joseph Peninsula, which included St. Joseph Peninsula State Park, and the other was located on the eastern portion of Crooked Island (Moyers et al. 1999). In 1975, Hurricane Eloise fragmented Crooked Island into two separate land bodies, forming eastern and western segments now known as East Crooked Island and West Crooked Island, respectively (James 1987, Moyers et al. 1999). Trapping efforts conducted by Tyndall Air Force Base (AFB) in 1985 and 1986 on West Crooked Island failed to capture any mice (Gore 1987). Moyers et al. (1999) reported that no St. Andrew beach mouse tracks had been found on West Crooked Island as recently as 1998. During the mid to late 1980s, trapping and track survey efforts conducted on East Crooked Island showed mice were still present on the eastern segment of the island (Gore 1987 and 1990, James 1987). By 1992-1993, trapping efforts were unsuccessful in producing captures of St. Andrew beach mice on East Crooked Island and the population was, therefore, thought to be extirpated (Gore 1994, Alabama Cooperative Fish and Wildlife Research Unit 1997). Plans to reintroduce St. Andrew beach mice, using individuals from the St. Joseph Peninsula State Park population, were initiated in 1994 (Moyers et al. 1996). Reintroduction of 43 individuals from St. Joseph Peninsula State Park took place in November 1997 (16 individuals) and January 1998 (27 individuals), and December 1998 (4 individuals). Subsequent monitoring efforts to assess the effectiveness of the reintroduction resulted in the capture of 38 individuals in February 1998 and 34 individuals in May 1998 (Moyers et al. 1999). Trapping efforts in 2000 and 2002 resulted in the capture of 132 individuals and 41 individuals, respectively (Lynn 2000 and 2002). Furthermore, in April 2001, 55 St. Andrew beach mice were captured on private lands south of Tyndall AFB property (Moyers and Shea 2002). Recent work by FWC have found St. Andrew beach mice on the Tyndall AFB property of East Crooked Island (Slaby 2005). Surveys conducted on East Crooked Island between May 2005 and January 2007 found beach mice were present on Tyndall AFB property and also on adjacent private lands southeast of Tyndall AFB property (Loggins et al. 2008). Loggins et al. (2008) estimated an average of 59.5 +/- 4% of East Crooked Island was occupied by St. Andrew beach mice. These results indicate that St. Andrew beach mice have become reestablished on East Crooked Island. Trapping and track surveys were also conducted on St. Joseph Peninsula from the mid-1980s through the early 2000s. These efforts showed a continued presence of St. Andrew beach mice on St. Joseph Peninsula State Park (James 1987, Gore 1990 and 1995, Bates 1992, Moyers et al. 1996 and 1999, Loggins et al. 2008). South of the Park and at Cape San Blas, Gore (1990) was unable to capture any St. Andrew beach mice during his trapping effort in 1989. In 1992 and 1993, St. Andrew beach mice were captured south of the Park and north of Cape San Blas (Gore 1994). Trapping in 1996, at Rish Park and neighboring private parcels, showed a continued presence of St. Andrew beach mice south of the St. Joseph Peninsula State Park, (Holler 1996, Loggins et al., 2008). In November 2004, track surveys south of the Park showed a presence of beach mice from the Park boundary south to approximately the “stump hole” (the area on the Peninsula, just north of Cape San Blas, where the peninsula constricts vehicle access point at the reinforced rock portion of Highway 30E). Surveys conducted again between 2005 and 2007 south of the Park showed a presence of mice (Loggins et al. 2008). Tracking surveys in 2005 by the FWC showed that mice were present at St. Joseph Peninsula State Park (Slaby 2005). Surveys conducted between May 2005 and April 2006 within the State Park showed the continued presence of beach mice (Loggins et al. 2008). Loggins et al. (2008) estimated an average of 61.0 +/- 9% of St. Joseph Peninsula State Park was occupied by St. Andrew beach mice.

Population persistence

Population viability analysis (PVA) is essentially a demographic modeling exercise to predict the likelihood a population will continue to persist over time (Groom and Pascual 1997). PVAs have many forms, but all types attempt to mimic or replicate the complex process of population growth or decline and the factors that regulate that process. The objective of a PVA for beach mice is to determine how much and what configuration of habitat is necessary to reasonably assure that the species will survive and recover. The question requires biological data and an understanding of beach mouse reproduction and survival, rates of population growth or decline, and how the environment (including hurricanes) will cause variation in these parameters over short and long periods of time.

The Holler et al. (1999) analyses indicated that certain populations of the beach mouse were at risk of extinction. At Fort Morgan, the Alabama beach mouse population had a 49.4 percent chance of becoming functionally extinct within 5 to 20 years. At Bon Secour NWR, the Alabama beach mouse had a 0.2 percent chance of becoming functionally extinct between 16 and 23 years. The occurrence of hurricanes would increase that risk depending on their frequency and severity and the ability of beach mouse populations to recover.

A specific PVA has not been conducted for the Choctawhatchee beach mouse because necessary life history data or adequate long term data is not available.

At Gulf Islands National Seashore-Perdido Key, the Perdido Key beach mouse had a 100 percent chance of reaching one individual (becoming functionally extinct) within 21 to 45 years. At Florida Point, the Perdido Key beach mouse had a 1.3 percent chance of becoming functionally extinct within 13 to 20 years.

The Service is cognizant of the potential for any of the beach mouse subspecies to go extinct. The Perdido Key beach mouse population at Florida Point was declared extirpated in 1999 although it had the lowest probability of going extinct compared to the other subspecies. Reasons for extinction include habitat loss, fragmentation, or degradation from natural (hurricanes) or human (development and recreation) causes, genetic viability, and native and non-native depredation. Holler et al. (1999) noted that the PVA presented further evidence that habitat fragmentation will continue to exacerbate the risk of extinction.

Status and distribution

Historically, the Alabama beach mouse ranged from the tip of the Fort Morgan Peninsula to Perdido Pass and Ono Island, all within Baldwin County, Alabama, utilizing approximately 33.5 miles of shoreline (Service 2009d). This species now only occurs in a reduced distribution in Baldwin County, Alabama along 13 miles of Alabama coastline (Service 2009d). Range-wide population sizes are difficult to estimate for the species due to large fluctuations in populations and limited access to privately held lands that provide Alabama beach mouse habitat. Grid sampling and Habitat Conservation Plan (HCP) monitoring have indicated recolonization by the Alabama beach mouse to areas affected by Hurricanes Ivan and Katrina; however, the amount of available habitat is on the decline due to coastal development. It inhabits coastal dune ecosystems, particularly secondary (i.e., frontal) dunes and scrub (i.e., tertiary and interior scrub) dunes (Service 2009d). The Alabama beach mouse is listed as endangered due to habitat loss and fragmentation associated with development, isolation of small local populations precluding gene flow, predation (particularly by cats), and losses of isolated populations following tropical storms and hurricanes (Service 2009d).

The Choctawhatchee beach mouse inhabits coastal sand dune systems characterized by moderate vegetation cover and interior scrub forest dunes. Historically, the species ranged from Destin Pass in Choctawhatchee Bay (Okaloosa County, Florida) to East Pass in St. Andrew Bay (Bay County, Florida) along approximately 53 miles of coastline (Service 2006a). Its current range is limited to an estimated 10-15 miles of coastline on the Gulf coast of the Florida panhandle between Choctawhatchee Bay and St. Andrew Bay (Bay, Okaloosa, and Walton counties) (Service 2006a). Range-wide population sizes are difficult to estimate for the species due to large fluctuations in populations and gaps in public land sampling data. There are five populations of Choctawhatchee beach mouse currently recognized, occurring at: 1) Topsail Hill Preserve State Park; 2) Shell Island/West Crooked Island; 3) Grayton Beach; 4) Deer Lake State Park; and 5) Henderson Beach. The Choctawhatchee beach mouse is listed as endangered due to population decline resulting from habitat disturbance, fragmentation, isolation, and loss resulting from coastal development. Subsequent effects include feral cat predation and losses of isolated populations following storm events (Service 2006a).

The Perdido Key beach mouse inhabits high quality coastal sand dune ecosystems with native vegetation (including scrub and frontal dunes) (Service 2014b). Historically, the species occurred on Perdido Key between Perdido Bay, Alabama and Pensacola, Florida. It now occurs only in Baldwin County, Alabama and Escambia County, Florida on a portion of its historic range. The Perdido Key beach mouse is listed as endangered due to habitat loss and fragmentation associated with development and suitable habitat being located on private lands, predation (particularly by cats), and population effects following tropical storms and hurricanes. Range-wide population sizes are difficult to estimate for the species due to large fluctuations in populations and limited access to privately held lands that provide Perdido Key beach mouse habitat. The current distribution of Perdido Key beach mice that occur on public lands are found at the following locations: 1) Gulf State Park; 2) Perdido Key State Park; and 3) Gulf Islands National Sanctuary. The populations of this species were extirpated from each locale at one time, but due to translocation efforts populations have been reestablished, yet not considered to be self-sustaining. The populations that occur on adjacent private lands are currently unknown.

The St. Andrew beach mouse inhabits primary and secondary coastal sand dunes. Historically, the species ranged from St. Joseph Peninsula northwest along the coastal mainland adjacent to St. Joseph Bay to Crooked Island at the East Pass of St. Andrews Bay in Florida (63 FR 70053). It now only occurs in Bay and Gulf counties, Florida on a portion of its historic range (in two core populations: East Crooked Island in Bay County and St. Joseph Peninsula in Gulf County). The species is listed as endangered due to habitat loss and fragmentation associated with development, dune encroachment by vehicles and pedestrians, shoreline erosion, predation (particularly by cats), and population effects following tropical storms and hurricanes. A study in 2008 indicated approximately 3,000 individuals at East Crooked Island and approximately 1,775 individuals in the front dunes of St. Joseph State Park. These numbers suggested an increase in population at East Crooked Island from earlier studies conducted in 1992 and 1998 and also at St. Joseph State Park compared to population estimates in that area from 1992 and 1999.

Analysis of the species/critical habitat likely to be affected

Alabama, Choctawhatchee, Perdido Key, and St. Andrew beach mice and their critical habitats that occur along the Gulf coast may be affected by the proposed action. The effects of the proposed action on beach mice and their habitat will be considered further in the remaining sections of this opinion.

ENVIRONMENTAL BASELINE

Status of the species within the action area

The entire range of the Alabama, Choctawhatchee, Perdido Key, and St. Andrew beach mice are within the action area. All of the designated critical habitat for the Alabama beach mouse (72 FR 4330) and Choctawhatchee, Perdido Key, and St. Andrew beach mouse (71 FR 60238) occur within the action area, as described above.

Factors affecting species environment within the action area

Habitat loss and fragmentation associated with residential and commercial real estate development are the primary threats contributing to the endangered status of beach mice (Holler 1992, Humphrey 1992, 71 FR 5515, 71 FR 44976). Isolation of small local populations of beach mice reduces or precludes gene flow between these populations and can result in the loss of genetic diversity. Demographic factors, such as predation (particularly by cats), disease and competition, are intensified in small, isolated local populations which may be rapidly extirpated by these pressures. Especially when coupled with events, such as tropical storms, reduced food availability and/or reduced reproductive success, isolated local populations may experience severe declines or extirpation (Caughley and Gunn 1996, 71 FR 5515).

Tropical storms are also a significant threat to beach mouse subspecies. Because of fragmented habitat, loss of habitat from hurricanes, and low numbers of beach mice surviving hurricanes, the subspecies ability to quickly repopulate has been compromised. Hurricanes can impact beach mice either directly (e.g., drowning) or indirectly (loss of habitat). Additionally, hurricanes can affect beach mice on either a short-term basis (temporary loss of habitat) or long term (loss of food, which in turn may lead to increased juvenile mortality, which can lead to a depressed breeding season). How a hurricane affects beach mice depends primarily on its characteristics (winds, storm surge, rainfall, etc.), the time of year, and where the eye crosses land (side of hurricane-clockwise or counterclockwise). The frequency between severe weather events could compromise the ability of beach mice to survive and recover.

Beach mice have a number of natural predators including, but not limited to, the coachwhip, corn snake, pygmy rattlesnake, Eastern diamondback rattlesnake, short-eared and great-horned owl, great blue heron, northern harrier, red fox, gray fox, skunk, weasel, and raccoon (Blair 1951, Bowen 1968, Holler 1992, Novak 1997, Moyers et al. 1999, Van Zant and Wooten 2003). Natural predation of beach mouse populations that have sufficient recruitment and habitat availability is generally not a concern. However, excessive predation pressure from natural and non-native predators may result in the extirpation of small, isolated local populations of beach mice, especially after hurricanes when both predators and prey are more concentrated in smaller and often isolated habitat patches.

A significant predation concern for beach mice is free-roaming and feral domestic cats. The damage inflicted on birds, small mammals, reptiles, and amphibians from cats is in the hundreds of millions each year (American Bird Conservancy 1999). Cat tracks have been observed in areas of low trapping success for beach mice.

EFFECTS OF THE ACTION

Direct effects

Potential sources of impact to the beach mice and their critical habitat from existing and proposed oil and gas activities are habitat loss and fragmentation, disturbance from aircraft and boat vessel traffic, effects from trash and debris, and OCS-related air emissions. New pipeline construction should only have a minimal effect on beach mice and their critical habitat because the range of this species encompasses an area where new pipeline construction is not anticipated. In addition, new pipelines that go ashore require an environmental analysis before approval. Where pipeline landfall occurs, the permitting process encourages the development of measures to minimize and potentially eliminate impacts to federally listed species.

Aircraft and boat vessel traffic should have minimal impact on those beach mice due to the mice's nocturnal behavior and because the majority of helicopter traffic is expected to occur west of any beach mouse habitat. In addition, those activities are not anticipated to significantly increase the amount of existing routine helicopter and vessel traffic within the action area. Impacts to the beach mice from helicopter and vessel traffic should, therefore, be minimal.

Although accumulations of marine trash and debris do not appear to be a severe problem in beach mouse habitats, beach mice may ingest trash and debris mistakenly for food. Ingestion could interfere with their digestive process or produce a fatal response. BSEE, however, prohibits the disposal of equipment, containers, and other materials into offshore water by lessees (30 CFR 250.300; also BSEE NTL No. 2015-G03 Marine Trash and Debris Awareness) and requires Marine Trash and Debris Awareness Training.

Coastal discharges are not expected to be a concern because the beach mouse subspecies rely on fresh rather than saline drinking water. BOEM/BSEE anticipates minimal effects to air quality associated with OCS oil and gas emissions due to prevailing atmospheric conditions, emission heights and rates, and pollutant concentrations. Because emissions from OCS-related activities are not likely to impact ambient air quality effects to beach mice from decreased air quality are expected to be negligible.

Indirect effects

Although the primary habitat of the beach mice is in coastal dunes and their food sources are located above the high tide line, there is a potential for impacts from an oil spill. The National Park Service described the following occurrence during a small oil spill on Horn Island, Mississippi, in September 1989.

> Several days after landfall of the Horn Island spill, strong surf action and winds combined to remobilize and distribute significant amounts of oil from the beach face up into the adjacent primary dunes. The spray generated by the wind and surf action was sufficiently oily to completely coat most of the dune vegetation, and resulted in leaf browning which persisted until the next growing season (Zimmerman 1990).

On Gulf coast areas with relatively narrow beaches, an oil spill occurring during an episode of high winds and seas (a relatively common occurrence) could result in severe mortality of plant and insect species associated with coastal beach/dune ecosystems.

Direct contact with spilled oil can cause dermatitis. Fur will mat and, therefore, lose its insulating properties against heat and cold. Other direct toxic effects may result from oil ingestion or asphyxiation or from inhalation of fumes. Indirect effects may include contamination and depletion of food supply, destruction of habitat, and fouling of burrows.

Impacts can also occur from spill-response activities. Vehicular traffic and other activities associated with oil-spill cleanup degrade preferred habitat and cause displacement of mice from these areas without thorough training of all personnel with in an emergency would need to happen on short notice.

There is no definitive information on the persistence of oil in the event that a spill was to contact beach mouse habitat. In Prince William Sound, Alaska, after he Exxon Valdez spill in 1989, buried oil has been measured in the intertidal zone of beaches, but no effort has been made to search for residual buried oil above high tide. Similarly, NRC (2003) makes no mention of studies of oil left above high tide after a spill. Regardless of the potential for persistence of oil in beach mouse habitat, a slick cannot wash over the foredunes unless carried by a heavy storm swell.

According to the OSRA, there is approximately less than a 0.5 percent probability that an oil spill >1,000 barrels would occur and contact beach mouse habitat within 10 or 30 days of a spill (note again that those probabilities do not include clean-up activities and natural weathering of the spill). This represents a minimal threat to beach mouse critical habitat if a spill were to occur. The BOEM/BSEE, USEPA, and USCG have regulations, requirements, and recommendations that should prevent or reduce the likelihood of a spill and prevent or reduce impacts to beach mice and their critical habitat if a spill occurs. This and the weathering of oil in the environment should significantly minimize potential impacts if a spill occurs.

Effects summary

Activities occurring as a result of the proposed action may affect beach mice; however, no direct loss of habitat is anticipated. It is expected that the majority of the effects from the major-impact producing factors would be sublethal, causing discountable or insignificant effects. Changes in local population numbers and distribution are assumed as a result of any of these factors, and could have significant impacts to beach mice. Although the probability of an oil spill reaching beach mouse habitat is very small, it is a concern. The probabilities, developed by BOEM/BSEE, of an oil spill occurring and contacting habitat (including critical habitat) where beach mice occur overestimate contact probability because they do not account for naturally occurring events such as weathering and activities included in the proposed action (e.g., clean up, containment, etc.). Although the reduction in those probabilities could not be quantified, it is the Service's belief that those reductions make the likelihood of contact extremely low. In addition, because contact with habitat does not necessarily mean contact with the individual, the likelihood of incidental take is not reasonably certain to occur. If a spill were to occur, the adverse effects that might occur to that critical habitat would be temporary in nature and are of low probability.

As discussed earlier, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to beach mice associated with a spill of this magnitude are not addressed in this BO.

STATUS OF THE SPECIES/CRITICAL HABITAT

West Indian Manatee (*Trichechus manatus latirostris*)

Species/critical habitat description

Manatees are large fusiform-shaped mammals with skin that is uniformly dark grey, wrinkled, sparsely haired, and rubber-like. Manatees possess paddle-like forelimbs, no hind limbs, and a spatulate, horizontally flattened tail. Females have two axillary mammae, one at the posterior base of each forelimb. Their bones are massive and heavy with no marrow cavities in the ribs or long bones of the forearms (Odell 1982). Adults average about 10 feet long and 2,200 pounds in weight, but may reach lengths of up to 15 feet (Gunter 1941) and weigh as much as 3,570 pounds (Rathbun et al. 1990). Newborns average 4 to 4.5 feet long and weigh about 66 pounds (Odell 1981). The nostrils located on the upper snout, open and close by means of muscular valves as the animal surfaces and dives (Husar 1977, Hartman 1979). Manatees use a muscular flexible upper lip with the forelimbs to manipulate food into the mouth (Odell 1982). Bristles are located on the upper and lower lip pads. Molars designed to crush vegetation form continuously at the back of the jaw and move forward as older ones wear down (Domning and Hayek 1986). The eyes are very small, close with sphincter action, and are equipped with inner membranes to protect the eyeball. The ears are external, minute, with no pinnae. The anatomy of the internal ear structure indicates they can hear sounds within a relatively narrow low frequency range that their hearing is not acute, and they have difficulty in localizing sound (Ketten et al. 1992). However, Gerstein (1995) suggested manatees might have greater low-frequency sensitivity than other marine mammal species.

In 1967, both the Florida and Antillean subspecies of manatees (*T. manatus latirostris and T. inanatus manatus*) were listed as endangered throughout their respective ranges (32 FR 4061) and received Federal protection with the passage of the Act in 1973. Because the manatee was designated as an endangered species prior to enactment of the Act, there was no formal listing package identifying threats to the species, as required by section 4(a)(l) of the Act.

Critical habitat for the Florida manatee was designated in 1976 (50 CFR 17.95). This was one of the first designations of critical habitat for an endangered species and the first for an endangered marine mammal. Critical habitat for any species is described as the specific area within the geographic area occupied by the species (at the time it is listed under the provisions of section 4 of the Act) on which are found those physical or biological features (i. e., constituent elements) essential to the conservation of the species and which may require special management considerations or protection. No specific primary or secondary constituent elements were included in the critical habitat designation. However, essential habitat features for the manatee could include seagrasses for foraging, shallow areas for resting and calving, channels for travel and migration, warm-water refuges during cold weather, and fresh water for drinking.

Designated critical habitat on the west coast of Florida includes Crystal River in Citrus County, portions of the Little Manatee River in Hillsborough County, the Manatee River in Manatee County, the Myakka River in Sarasota and Charlotte counties, the Peace River in DeSoto and Charlotte counties, Charlotte Harbor in Charlotte County, and the Caloosahatchee River in Lee County. The designation includes all the coastal waters in Lee, Collier, and Monroe counties between Gordon's Pass (Collier County) and Whitewater Bay (Monroe County). While critical habitat has been designated for the Caloosahatchee River, the tributaries that connect to the river, including the Orange River, are not designated as critical habitat.

Life history

Like many large mammals, manatees have a potentially long life span (60 years), mature at 4 to 7 years, have a low reproductive rate (one calf every 3 years, 11 - 13 month gestation period), and the calf will stay with the parent for 2 years (O'Shea and Hartley 1995, Marmontel 1995, Ode1l et al. 1995, Rathbun et al. 1995, Reid et al. 1995). For species with this life-history strategy to persist, adult survival rates need to be high and stable. Adult survival rate is the critical determinant of population growth rate. Long-term photo identification studies show that adult manatees have an annual survival rate of about 96 percent in certain regional "subpopulations" that have relatively low human-related mortality (Langtimm et al. 2004). Accordingly, manatee populations are vulnerable to elevated mortality rates. Florida manatees have a low level of genetic diversity, possibly resulting from a founder effect or a population bottleneck (Garcia-Rodriguez et al. 1998). This means that individual manatees are genetically very similar to one another. This similarity can result if only a few individuals started the population or if there was a time when the population decreased to only a few individuals. Lack of genetic diversity within a population can result in inbreeding and a decrease in reproductive fitness.

Breeding takes place when one or more males (ranging from 5 to 22 individuals) are attracted to an estrous female to form a temporary mating herd (Rathbun et al. 1995). Mating herds can last up to 4 weeks, with different males joining and leaving the herd daily (Hartman 1979, Bengston 1981, Rathbun et al. 1995, Rathbun 1999). Permanent bonds between males and females do not form. During peak activity, the males in mating herds compete intensely for access to the female (Hartman 1979). Successive copulations involving different males have been reported. Some observations suggest larger, presumably older, males dominate access to females early in the formation of mating herds and are responsible for most pregnancies (Rathbun et al. 1995).

Although breeding has been reported in all seasons, Hernandez et al. (1995) reported histological studies of reproductive organs from carcasses of males found evidence that sperm production in 94 percent of adult males occurred between March and November. Females appear to reach sexual maturity by about age 5, but have given birth as early as 4 (Marmontel 1995, Odell et al. 1995, O'Shea and Hartley 1995, Rathbun et al. 1995). Males may reach sexual maturity at 3 to 4 years of age (Hernandez et al. 1995). Manatees may live in excess of 50 years (Marmontel 1995), and evidence for reproductive aging is unclear (Marmontel 1995, Rathbun et al. 1995).

Calf dependency usually lasts 1 to 2 years after birth (Hartman 1979, O'Shea and Hartley 1995, Rathbun et al. 1995, Reid et al. 1995). Calving intervals vary greatly among females, with an average birth cycle of 2 to 2.5 years, but may be considerably longer depending on age and perhaps other factors (Marmontel 1995, Odell et al. 1995, Rathbun et al. 1995, Reid et al. 1995). Females that abort or lose a calf due to perinatal death (small manatees, less than 60 inches in length) (O'Shea and Hartley 1995), may become pregnant again within a few months (Odell et al. 1995) or even weeks (Hartman 1979).

Manatee distribution and dispersal patterns as well as numbers of individuals within an area can vary considerably from year-to-year and season-to-season. This variability in dispersal patterns is dependent on a variety of biotic and abiotic factors (e.g., mating season, foraging areas, warm-water discharges, and freshwater sources). Manatees often use secluded canals, creeks, embayments, and lagoons for feeding, resting, playing, mating, and calving (Marine Mammal Commission 1986 and 1988).

Manatees frequent coastal, estuarine, and riverine habitats and are capable of extensive north-south migrations. These north-south migrations are largely determined by water temperatures below 68° F. When ambient water temperatures drop below 68° F in autumn and winter, manatees aggregate within the confines of natural or artificial warm-water refuges or move to the southern tip of Florida (Snow 1991). Large groups of manatees, from 300 to more than 500, have been observed in Florida when the ambient temperature drops below 66ºF (Craig and Reynolds 2004, UNEP 2010). These aggregations tend to be less social and more resource-based occurring in natural springs and around power or industrial plants that are discharging warm water (Reynolds and Wilcox 1985, Laist and Reynolds 2005, UNEP 2010). Most warm-water artificial refuges are created by outfalls from power plants or paper mills. As ambient water temperatures rise, manatees disperse from these winter aggregation areas. While some remain near their winter refuges, others undertake extensive migrations along the coasts of Florida and far up rivers and canals. Many manatees return to the same warm-water refuges each year. However, some manatees use different refuges in different years, and others use two or more refuges in the same winter (Reid and Rathbun 1984, Rathbun et al. 1990, Reid et al. 1991). There are numerous lesser known, minor aggregation areas used as temporary thermal refuges. Many of these areas are canals or boat basins where warm-water temperatures persist as temperatures in adjacent bays and rivers decline. At the end of winter, manatees leave warm-water aggregation sites and head for warm weather use areas. There appears to be no significant spring aggregation areas on the east coast. During the summer, manatees can be found throughout Florida where water depths are greater than 3.3 to 6.6 feet (O'Shea 1988).

Manatees depend on natural springs, manmade warm-water refugia, areas with vascular plants, and freshwater sources. Manatees normally migrate along shorelines and use deeper corridors to access shallow water feeding and resting areas. Manatees are herbivores that feed opportunistically on a wide variety of aquatic vegetation. Feeding rates and food preferences depend, in part, on the season and on available plant species. Manatees frequently feed in water depths of 3 to 9 feet where aquatic vegetation is abundant. Seagrasses appear to be a staple of the manatee diet in coastal areas (Ledder 1986, Provancha and Hall 1991, Kadel and Patton 1992, Koelsch 1997, Lefebvre et al. 2000).

Extensive foraging resources generally typify summer use areas. Seagrasses and other food sources occur throughout coastal Florida. There are an estimated 3.73 million acres of open water habitat in coastal and interior areas, of which an estimated 1.1 million acres are designated manatee critical habitat. Almost 57,000 acres of known manatee aggregation habitat exist in the state, 85 percent of which is located along the Atlantic coast and in southwest Florida.

Manatees have low metabolic rates indicating a possible adaptation to their large size and low nutrient food sources, or to permit long dives, since manatees have less advanced diving abilities than other marine mammals. Manatees can remain submerged for several minutes with the longest submergence record lasting 24 minutes. Manatees increase submergence times while feeding and resting. Female manatees coordinate their breathing and submergence times with their calves. Manatees do not appear to be fast swimmers, but they usually swim 4 to 10 km an hour and may attain faster speeds in short bursts (Husar 1977).

Manatees have no known predators, except for humans. Although reactions may be different, manatees are susceptible to the same natural and human disturbances other aquatic organisms experience (e.g., changes in water quality, loss of habitat, and susceptibility to diseases and natural catastrophes). Manatees have robust immune systems that have, through the present time, provided disease resistance.

Population dynamics

One to three times each winter, a coordinated series of statewide aerial surveys and ground counts, known as the synoptic surveys, are conducted by FWC to count the number of manatees in Florida. The best, current, minimum population estimate of the statewide manatee population is approximately 6,250 animals based on a single statewide count at warm-water refuges and adjacent areas in February 2016 (FWC-FWRI Manatee aerial surveys, unpublished data, 2016).

A January 2010 statewide survey counted approximately 5,000 manatees prior to a die-off event between 2010 and 2014 in which at least 2,822 manatees died. As a result of the counts before and after the die-off event suggests a resiliency in the Florida population (FWC-FWRI Manatee aerial surveys, unpublished data, 2016; Runge et al. 2015, 82 FR 16668). Franklin and Frankham (1998) suggested that an effective population size of 500-1,000 animals is needed to retain genetic variation for future evolutionary change.

Status and distribution

The Federal government recognized the threats to the continued existence of the Florida manatee for over 30 years. The West Indian manatee was first listed as an endangered species in 1967 under the Endangered Species Preservation Act of 1966 (16 U.S.C. 668aa(c)) (32 FR 48:4001). The Endangered Species Conservation Act of 1969 (16 U.S.C. 668aa(c)) continued to recognize the West Indian manatee as an endangered species (35 FR 16047), and the West Indian manatee was also among the original species listed as endangered pursuant to the Endangered Species Act of 1973. Critical habitat was designated for the manatee in 1976. The justification for listing as endangered included impacts to the population from harvesting for flesh, oil, and skins as well as for sport, loss of coastal feeding grounds from siltation, and the volume of injuries and deaths resulting from collisions with the keels and propellers of powerboats. In 2016, the West Indian manatee was proposed for reclassification from endangered to threatened under the Act and in 2017 the reclassification to threatened was finalized (82 FR 16668).

Florida manatees can be found throughout the southeastern United States; however, within this region, they are at the northern limit of their range (Lefebvre et al. 2000). Because they are a subtropical species with little tolerance for cold, they remain near warm-water sites in peninsular Florida during the winter. During periods of intense cold, manatees will remain at these sites. During warm interludes, they move from the warm-water areas to feed and return once again when the water temperature is too cold (Hartman 1979). During warmer months, manatees may disperse great distances. They have been sighted as far north as Massachusetts and as far west as Texas and in all states in between (Rathbun et al. 1982, Fertl et al. 2005). Warm weather sightings are most common in Florida and coastal Georgia.

Threats

Threats encompass anthropogenic factors and catastrophic natural events that could cause declines in reproductive and survival rates or loss and degradation of habitat. The primary threats to manatee populations are collisions with watercraft and loss of warm-water refuges. The largest known cause of human-related mortality of manatees in Florida is watercraft collisions. Watercraft strikes result in numerous injuries and deaths each year. The future of the Florida manatee is jeopardized by the predicted loss and deterioration of warm-water habitat, including retirement or deregulation of aging power plants and reduction in natural spring flows.

About half of adult mortality rangewide is attributable to human-related causes, primarily watercraft collisions (Deutsch et al. 2002). This is significant because the manatee population growth rate is highly sensitive to changes in adult survival rate (Eberhardt and O'Shea 1995, Marmontel et al. 1997, Runge et al. 2004). The immature age class most common "cause of death" category is perinatal mortality and watercraft collisions is the next highest known cause of death.

Warm-water habitat is essential for manatee survival during cold weather. Prolonged exposure to cold water temperatures can result in debilitation and/or death due to "cold stress syndrome" (Bossart et al. 2002). However, when compared to all other threats, including the loss of warm-water habitat, watercraft-related mortality poses the most serious long-term risk to the growth and resilience of the manatee population.

Other threats to manatees include crushing or entrapment in gates and locks, entanglement in ropes, lines, and nets, ingestion of fishing gear or debris, vandalism, poaching, and exposure to red tide brevetoxin (Bossart et al. 1998). Red tide represents a major natural source of mortality for manatees in the Southwest region. Hurricanes are another type of phenomenon that can potentially affect manatee populations.

Population Viability Analysis

Runge et al. (2015) conducted a population viability analysis to forecast the Florida manatee population under different scenarios regarding the presence of the following threats: watercraft-related mortality, loss of warm-water habitat in winter, mortality in water control structures, entanglement, and red tide. Runge et al. (2015) estimated the probability of the manatee population falling to less than 500 adults on either the Atlantic or Gulf coasts (from a 2012 statewide population size of approximately 4,000) within 150 years is 0.92 percent. Complete removal of the watercraft threat alone would reduce this risk to 0.06 percent; complete removal of the warm-water threat alone would reduce this risk to 0.10 percent; removal of both threats would reduce the risk to 0.0 percent.

Protection Measures

Watercraft speed zones have been established in some coastal Florida counties with high manatee-watercraft collision rates to slow watercraft to reduce collisions. Anecdotal information indicates that, when manatees detect the presence of an oncoming boat, they often, but not always, dive and swim rapidly out of its path. Their ability to elude the oncoming boat is largely determined by the speed of the approaching boat. Given ample time, manatees should be able to avoid lethal and injurious encounters with boats; thus, slow-moving boats are less of a threat to manatees. The Service has determined increased manatee speed zone enforcement is the primary conservation measure through which proposed projects could reduce the incidental take associated with watercraft collisions to an unlikely to occur level.

A number of significant power plants utilized by manatees in the winter have been repowered for a projected time of 40 years (Laist et al. 2013). To protect warm-water sites, a network of power plants discharge areas have been designated as warm-water sanctuaries/no-entry areas and manatee protection areas. Alternative warm-water sites are being constructed to compensate for the loss of upstream passive warm-water sites lost due to restoration activities (82 FR 16678; April 5, 2017).

Counties in Florida are required to develop manatee protection plans (MPPs) for the development of boat facilities. Based on manatee protection needs, an evaluation of natural resources, and economic and recreation demands, these MPPs specify locations for boat facility development. The MPPs are reviewed by the Service and the FWC to evaluate boat access projects. Proposed projects that are consistent with MPPs can then be permitted by permitting agencies to be constructed in waters used by manatees.

Issuance and renewals of National Pollution Discharge Elimination System (NPDES) permits for power plants, wastewater treatment plants, desalination plants and other discharges, are reviewed by the Service, the FWC, and others to insure that no new attractant discharges are created and that existing discharges do not adversely impact manatees.

Water control structures can cause entrapment and crushing of manatees. To prevent manatee deaths from these structures, manatee protection devices have been installed on all but one structure in Florida although efforts to have that remaining structure retrofitted are ongoing. The implementation of these devices has significantly reduced impacts to manatees from water control structures. Entrapment in storm water pipes and other structures large enough for manatees to enter has also been addressed through designs including features prohibiting manatee access such as bars or grates.

Fishery and marine debris is a continued threat to manatees. Multiple programs such as the derelict crab trap program, the monofilament recycling program, a program to rescue entangled manatees, as well as extensive outreach and education efforts have created awareness for this issue. To minimize fishing gear related entanglements, guidelines have also been developed.

Various efforts have been implemented to minimize manatee harassment. These efforts include outreach encouraging proper viewing practices and having designated manatee sanctuaries as well as no entry areas where waterborne activities are prohibited. Commercial manatee viewing businesses are also required to obtain permits restricting activities to prevent harassment of manatees from occurring while on National Wildlife Refuges.

Analysis of the species/critical habitat likely to be affected

The Florida manatee and its critical habitats occur within the action area and may be affected by the proposed action. The effects of the proposed action on manatees and their habitat will be considered further in the remaining sections of this opinion.

ENVIRONMENTAL BASELINE

Status of the species within the action area

Manatee distribution and dispersal patterns, and numbers of individuals within an area, can vary considerably from year-to-year and season-to-season. This variability in dispersal patterns is dependent on a variety of biotic and abiotic factors, such as warm-water discharges, freshwater supplies, high quality feeding areas, and mating season. During January 2003, there were three aerial surveys covering Florida. A total of 1,695; 1,814; and 1,705 manatees were observed along the east coast of Florida.

Factors affecting the species' environment within the action area

Manatee habitat is being fragmented due to development and ensuing seawalls and boat docks. Development reduces water quality and increases turbidity. Seawalls increase lateral scouring of the sea bottom where seagrasses grow. Boats often scar seagrass beds in shallow water also.

A significant factor affecting manatees within the action area is sublethal injuries due to boat interactions. On a continued basis, this type of injury could have an impact on maintaining a healthy and viable population. In that regard, most manatee carcasses examined bear scars from previous strikes with watercraft (Wright et al. 1995), and a significant number of living, but scarred, manatees exist.

EFFECTS OF THE ACTION

Direct effects

Potential sources of direct impact to the West Indian manatee from the proposed oil and gas activities are the degradation of water quality from operational discharges, noise disturbance from aircraft, collisions with vessel traffic, and effects from trash and debris.

The primary operational waste discharges generated during offshore oil and gas exploration and development are drilling fluids, drill cuttings, produced water, deck drainage, sanitary wastes, and domestic wastes. During production activities, additional waste streams include produced sand and well treatment, workover, and completion fluids. Minor additional discharges occur from numerous sources; these discharges may include desalination unit discharges, blowout preventer fluids, boiler blowdown discharges, excess cement slurry, and uncontaminated freshwater and saltwater. Discharges are regulated by the USEPA's NPDES permits. Pollutants discharged into navigable waters of the U.S. are regulated by USEPA under the Clean Water Act of 1972 and subsequent provisions (33 U.S.C. §1251 *et seq*.). Specifically, an NPDES permit must be obtained from USEPA under Sections 301(h) and 403 (45 FR 65953, October 3, 1980) of the Clean Water Act. Most operational discharges are diluted and dispersed when released in offshore areas, and they are not expected to significantly impact West Indian manatees.

Low-altitude aircraft overflights related to OCS oil and gas operations could affect the West Indian manatee. Aircraft overflights (either helicopter or fixed-wing) in close proximity to marine mammals may elicit a startle response due to either the increasing noise as the aircraft approaches or due to the physical presence of the aircraft in the air. Marine mammals often react to aircraft overflights by hasty dives, turns, or other abrupt changes in behavior. Responsiveness varies widely depending on factors such as species, the activity the animals are engaged in, and water depth (Richardson et al. 1995). Marine mammals engaged in feeding or social behavior are often insensitive to overflights, while those in confined waters or those with calves may be more responsive. The effects appear to be transient, and there is no indication that long-term displacement of marine mammals occurs. However, the absence of conspicuous response does not show that the animals are unaffected; it is not known whether these subtle effects are biologically significant (Richardson and Würsig, 1997). Aircraft noise is generally short in duration and transient in nature. Aircraft operations will continue to be numerous in the western and central planning areas, with much more limited activity in the EPA. FAA Advisory Circular AC 91-36D at 2000 feet encourages pilots to maintain higher than minimum altitudes (noted below) over noise-sensitive areas. FAA corporate helicopter policy states that helicopters should maintain a minimum altitude of 700 feet while in transit offshore and 500 feet while working between platforms. In addition, guidelines and regulations issued by NMFS under the authority of the MMPA do include provisions specifying helicopter pilots to maintain an altitude of 1,000 feet within 100 yards of marine mammals. It is unlikely that manatees would be affected by routine OCS helicopter traffic operating at these altitudes. Occasional overflights likely will have no significant impact on manatees.

Vessel strikes are the most common cause of human-induced mortality for manatees (FWC 2015b). While manatees are less common in the western Gulf, they are being seen more frequently and increased sightings indicate that there is a potential for risks to this species from OCS vessel traffic. The BOEM NTL 2016-G01, "Vessel Strike Avoidance and Injured/Dead Protected Species Reporting," provides minimization measures for vessel strike avoidance and reporting. Those measures include, but are not limited to: (1) maintain a vigilant watch for marine mammals and sea turtles and slow down or stop vessel to avoid striking protected species, (2) when sea turtles, small cetaceans, or manatees are sighted, attempt to maintain a distance of 50 yards or greater whenever possible, and (3) reduce vessel speed to 10 knots or less when mother/calf pairs, pods, or large assemblages of cetaceans and manatees are observed near an underway vessel when safety permits. An individual at the surface may indicate the presence of submerged animals in the vicinity of the vessel; therefore, precautionary measures should always be exercised. Adherence to the NTL protocols should significantly reduce the potential for impacts to that species from vessel strikes. In addition, manatees are rare within the CPA and WPA which is the area where most vessel traffic associated with the proposed action occurs, thereby further limiting the potential for adverse impacts to that species.

There are numerous existing laws, regulations, and enforcement guidelines that prohibit and discourage the disposal of solid debris in Gulf waters that can impact listed species and their critical habitats. For example, BSEE prohibits the disposal of equipment, containers, and other materials into offshore waters by lessees (30 CFR 250.300). Also, BSEE NTL No. 2015-G03 requires annual awareness training and the posting of placards to minimize the unintentional loss of debris from industry structures or vessels. BOEM/BSEE inspectors routinely conduct site visits and issue citations for noncompliance. In addition, MARPOL, Annex V. Public Law 100-220 (101 Statute 1458), which prohibits the disposal of any plastics, garbage, and other solid wastes at sea or in coastal waters, went into effect January 1, 1989, and is enforced by the USCG. The MDRPR (P .L 109-449) was enacted in December 2006. The purposes of the MDRPR are to help identify, determine sources of, assess, reduce and prevent marine debris and its adverse impacts on the marine environment and navigation safety; to reactivate the

Interagency Marine Debris Coordinating Committee; and to develop a Federal marine debris information clearinghouse. The MDRPR established, within the NOAA, a Marine Debris Prevention and Removal Program to reduce and prevent the occurrence and adverse impacts of marine debris on the marine environment and navigation safety. Greatly improved handling of waste and trash by industry, along with the annual awareness training required by the marine debris mitigations, is decreasing. OCS-related debris in the ocean and impacts to the West Indian manatee are, therefore, expected to be negligible.

Indirect effects

A potential source of indirect effects to the West Indian manatee would be caused by oil spills. Those species may be among the more vulnerable species because they forage in intertidal areas. Ingestion of oil could occur during the feeding process. Some oiling may occur through direct contact with oiled sediments or waves in the splash zone.

Manatees concentrate their activities in coastal waters often resulting at or just below the surface, which may bring them in contact with spilled oil (St. Aubin and Lounsbury 1990). Types of impacts to manatees from contact with oil include: asphyxiation due to inhalation of hydrocarbons; acute poisoning due to contact with fresh oil; lowering of tolerance to other stressors due to the incorporation of sublethal amounts of petroleum components into body tissue; nutritional stress through damage to food sources; and inflammation or infection and difficulty eating due to oil sticking to the sensory hairs around their mouths (Preen 1989, Sadiq and McCain 1993, Australian Maritime Safety Authority 2003). Direct contact with oil likely does not impact adult manatees' thermoregulation abilities because they use blubber for insulation. Also, they exhibit no grooming behavior that would contribute to ingestion (Service 2006b). Manatees are nonselective, generalized feeders that might consume tarballs along with their normal food, although such occurrences have been rarely reported (review of St. Aubin and Lounsbury 1990). A manatee might also ingest fresh petroleum, which some researchers have suggested might interfere with the manatee's secretory activity of their unique gastric glands or harm intestinal flora vital to digestion (Geraci and St. Aubin 1980; Reynolds 1980). Spilled oil may also affect the quality or availability of aquatic vegetation, including seagrasses, upon which manatees feed.

There have been no experimental studies and only a handful of observations suggesting that oil has harmed any manatees (St. Aubin and Lounsbury 1990), although for a population under pressure from other mortality factors (e.g., vessel strikes), even a localized incident could be significant (St. Aubin and Lounsbury 1990). Oil spills that may occur from OCS energy activities that reach the coast or the confines of preferred river systems and canals, particularly during winter (when the animals are most vulnerable physiologically), could further endanger local populations. The physiological costs of animals moving to colder waters to escape oiled areas may result in thermal stress that would exacerbate the effects of even brief exposure to oil (St. Aubin and Lounsbury 1990).

Spill-response activities that may impact manatees include increased vessel traffic, use of dispersants, and other remediation activities (e.g., controlled burns, skimmers, booms, etc.). The increased human presence after an oil spill (e.g., vessels) would likely cause changes in behavior and/or distribution, thereby potentially stressing manatees further and perhaps making them more vulnerable to various physiologic and toxic effects of spilled oil. In addition, the large number of response vessels could place manatees at a greater risk of vessel collisions, which could cause fatal injuries. Vessel noise would also increase as a result of increased vessel activity and could result in behavioral changes in some individuals.

Remediation activities that could impact manatees include the use of skimmers, booms, and controlled burns. Impacts from skimmers could be through capture and/or entrainment. Booming operations could potentially impact marine mammals, particularly manatees, as they are known to explore and interact with objects in their environment (Hartman 1979). Lines used to anchor booms are more likely that the boom itself to impact manatees if the booms are deployed in manatee habitat. Controlled burns could impact manatees if they were in the burning oil; however, it is expected that animals would avoid the area once it is ignited. In both skimming and controlled burning activities, the use of trained observers is common and reduces the likelihood of impacts to marine life.

Spill-response activities may include the application of dispersant chemicals to the affected area. Dispersant chemicals are designed to break oil on the water's surface into droplets, which breakdown in seawater. Virtually nothing is known directly about the effects of oil dispersants on manatees, except that presumably removing oil from the surface would reduce the risk of contact and render it less likely to adhere to the skin, baleen plates, or other body surfaces (Neff 1990). Impacts from dispersants are unknown but may be irritants to manatees.

According to the OSRA, there is a less than a 0.5 percent probability that an oil spill >1,000 barrels would occur and contact manatees and their habitat within 10 days in the EPA (note again that those probabilities do not include clean-up activities and natural weathering of the spill). The occurrence of manatees in the coastal central and western Gulf of Mexico waters is rare, therefore, significant impacts to that species in the CPA and WPA are not anticipated. The BOEM/BSEE, USEPA, and USCG have regulations, requirements, and recommendations to prevent or reduce the likelihood of a spill occurring and prevent or reduce impacts to West Indian manatees if a spill occurs. Those measures, and the weathering of oil in the environment, should significantly minimize potential impacts on West Indian manatees if a spill occurs. Indirect impacts to critical habitat could result from contact by spilled oil if a spill were to occur. As discussed above, there is a low probability of oil, spilled as a result of the proposed action, contacting areas where critical habitat has been designated.

Effects summary

For reasons discussed above, it is expected that the majority of the effects from the major-impact producing factors (i.e., degradation of water quality from operational discharges, noise disturbance, collisions with vessel traffic, and effects from trash and debris) would be sublethal (causing discountable or insignificant effects) or would be unlikely to occur. The greatest threat to manatees associated with OCS oil and gas development is oil spills. This species may be among the more vulnerable species because they forage in intertidal areas. Although the probability of an oil spill reaching manatee habitat is very small, it is a concern. The probabilities, developed by BOEM/BSEE, of an oil spill occurring and contacting habitat (including critical habitat) where manatees occur overestimate contact probability because they do not account for naturally occurring events such as weathering and activities included in the proposed action (e.g., clean up, containment, etc.). Although the reduction in those probabilities could not be quantified, it is the Service's belief that those reductions make the likelihood of contact extremely low. In addition, because contact with habitat does not necessarily mean contact with the individual, the likelihood of incidental take is not reasonably certain to occur. If a spill were to occur, the adverse effects that might occur to that critical habitat would be temporary in nature and are of low probability.

As discussed earlier, BOEM and BSEE continue to maintain that a low-probability catastrophic spill is not reasonably certain to occur and, therefore, is neither a direct nor an indirect effect of the proposed action. Accordingly, potential impacts to manatees associated with a spill of this magnitude are not addressed in this BO.

CUMULATIVE EFFECTS

Cumulative effects include the effects of future state, tribal, local or private actions that are reasonably certain to occur in the action area considered in this biological opinion. Future federal actions that are unrelated to the proposed action are not considered in this section because they require separate consultation pursuant to section 7 of the Act. Four major activities are reasonably certain to occur within the action area: 1) commercial and recreational fishing, 2) release of marine debris and trash from commercial and recreational vessels related/unrelated to the offshore oil and gas industry, 3) transportation of imported oil and other petroleum products, and 4) coastal development for human communities and recreation.

The GOM provides nearly 21 percent (in weight and value) of the commercial fish landings in the continental U.S. Most commercial species harvested from federal waters of the GOM are considered to be at or near overfished conditions. Continued fishing at the present levels may result in rapid declines in commercial landings and eventual failure of certain fisheries. Recreational fishing is a major recreational activity occurring in Federal waters and associated with oil and gas platforms. The northern GOM coastal zone is one of the major recreational regions of the U.S. for marine fishing. Nearly all species significant to the GOM's commercial and recreational catches are estuarine-dependent. Impacts to the estuarine ecosystem include inshore water quality degradation, loss of wetlands, and natural catastrophes (hurricanes). Fishery Management Plans are developed by the GOM Fishery Management Council to assess and manage commercial and recreational species of fish harvested from federal waters and in need of conservation (MMS 1997). NMFS is the Federal agency responsible for regulating the commercial fishing industry and recreational fishing within these waters, and uses the management plans to implement those regulations.

Over the last several years, companies have employed waste reduction and improved waste-handling practices to reduce the amount of trash offshore that could potentially be lost into the marine environment. Improved waste management practices have resulted in a marked decline in accidental loss of trash and debris (MMS 2002). In accordance with MARPOL Annex V, all ships and watercraft (including all commercial and recreational vessels) are limited in the type and location of dumping of vessel-generated garbage and solid waste items, both at sea and in U.S. navigable waters (USCG 2001).

As previously mentioned, the NPS conducted a marine debris 5-year monitoring program on National Park beaches on the Atlantic, Gulf, and Pacific coasts after Annex V of MARPOL became effective. Generally, the rates of debris accumulation for the park beaches in 4 years of monitoring did not show an increase or decrease (Cole *et al*. 1995). However, results of the fifth year survey at Gulf Islands National Seashore in Florida and Mississippi showed the lowest number of accumulated debris items since inception of the monitoring. Neither survey distinguished the amount of debris attributed to oil and gas activities. Thus, it is anticipated that impacts to listed species caused by ingestion, contact or entanglement of trash and debris should remain the same or decrease. In addition, educating the commercial and recreational vessel/ boating sectors about enforcement of the MARPOL regulations should reduce, or at least stabilize, the amount of illegal disposal of trash in the OCS.

Besides being the leader in domestic offshore production, between 60 and 65 percent of the crude oil being imported into the United States comes through the GOM (MMS 2002). Most offshore spills from non-OCS related activities are the result of vessel accidents involving import/export tankers, and barge and tank vessels carrying foreign or state-produced crude oil. Most of the large coastal spills are terminal-related events involving coastal barging operations. The smaller non-OCS spill events involving offshore and coastal spills are the result of cargo transfer mishaps, which include lightering of oil in the GOM. Of these potential non-OCS related spill events, large tanker spills resulting from collisions and groundings have the greatest chance of reaching and impacting sensitive coastal habitat at beaches and islands along the Texas and Louisiana coast.

Tanker imports and exports of crude and petroleum products into the GOM are projected to increase (USDOE, EIA, 2001). In 2000, approximately 2.08 billion barrels of crude oil and 1.09 billion barrels of petroleum products moved through analysis area ports. These volumes are projected to grow to 2.79 billion barrels of crude oil and 1.77 billion barrels of petroleum products by the year 2020 (MMS 2002). Projected spill rates from oil imported or tankered from outside waters into the Gulf of Mexico is 0.36 spills per billion barrels of oil at sea and 0.43 spills per billion barrels of oil in ports and harbors. From 1974 through July 1990, 82 tanker and barge spills greater than 1,000 barrels occurred in the Gulf region (Rainey 1992). The volume of oil spilled in the U.S. from tankers and barges appears to be decreasing and may be a result of more restrictive conditions placed on tanker operation and better response and containment by the Oil Pollution Act of 1990. The responsibility for cleanup of an "import" transportation-caused oil spill lies with the transportation vessel. Cost for the cleanup is only “federalized” (i.e., assumed by the federal government) if the responsible party refuses to take responsibility, requests assistance, is unable to meet its liability responsibilities, or the federal government deems the cleanup inadequate. For activities in OCS waters, the USCG is the federal agency responsible for the cleanup. The Service, NPS, and various state resource and regulatory agencies were involved in preparation of the sensitive-area maps for hazardous spill contingency plans produced by the USCG. Species-specific information has been incorporated into the USCG contingency plans.

Inshore spill events have the greatest likelihood of impacting coastal estuary and bay shoreline habitats used by plovers or nesting brown pelicans. Large inshore spills would occur primarily from tankers and barges while at dock or during intracoastal transport of crude oil and petroleum products in barges and pipelines.

Extensive refinery capacity, easy port access, and a well-developed transportation system have contributed to the development of the Gulf Coast region as an important center for handling oil to meet the world's energy needs. Both petroleum products and crude oil are imported and exported. To handle the large quantities of crude and petroleum products that enter and exit the GOM and to meet the Gulf States' energy needs, a tremendous amount of coastwide transport of oil occurs. The greater chance and larger spill volumes of spills occurring from imports, and intracoastal transportation of crude and refined petroleum products, present a concern for impacts to listed species.

CONCLUSION

For spill sizes ranging from 900,000 to 7.2 million barrels (estimated by BOEM/BSEE to qualify as a catastrophic spill for analytical purposes) the frequency range is between 1 in every 10,000 wells drilled to 1 in every 100,000 wells drilled. The Deepwater Horizon spill of 4.9 million barrels is closer in order of magnitude to the 1 in 100,000 wells drilled frequency. Ji et al. (2014) estimates the return period of a catastrophic oil spill in the OCS area to be approximately 165 years, with a 95% confidence interval between 41 and more than 500 years. Accordingly, BOEM/BSEE continue to maintain that a catastrophic spill is neither a direct nor indirect effect of the proposed action because a spill of that magnitude is not reasonably certain to occur over the project life. BOEM/BSEE anticipate that the most frequent spills associated with OCS oil and gas activities are generally less than 1 barrel in size. These spills are so small and of short duration that impacts to federally listed species are expected to be insignificant. As the more reasonably expected spill size increases (to the largest category of greater than 10,000 barrels in size), BOEM/BSEE has determined that up to one spill of this size is likely to occur over the 40-year period analyzed.

After reviewing the current status of the above species and critical habitat, the environmental baseline for the action area, the effects of the proposed oil and gas lease sale, and the cumulative effects, it is the Service's BO that the proposed leasing, exploration, development, production, and abandonment activities for existing and proposed WPA, CPA, and EPA Gulf of Mexico Planning Areas under the leasing and regulatory authorities of BOEM/BSEE are not likely to jeopardize the continued existence of the listed species under the Service's jurisdiction and are not likely to destroy or adversely modify their designated critical habitat, if any. In evaluating the potential that this action constitutes destruction or adverse modification of critical habitat, the Service has evaluated whether the action will appreciably diminish the value of the designated critical habitat for the recovery of the listed species. For each species with designated critical habitat, the adverse effects that may occur to that critical habitat would be temporary in nature and are of low probability.

INCIDENTAL TAKE STATEMENT

Section 9 of the Act and Federal regulation pursuant to section 4(d) of the Act prohibit the take of endangered and threatened species, respectively, without special exemption. Take is defined as to harass, harm, pursue, hunt, shoot, wound, kill, trap, capture or collect, or attempt to engage in any such conduct. Harm is further defined by the Service to include significant habitat modification or degradation that results in death or injury to listed species by significantly impairing essential behavioral patterns, including breeding, feeding, or sheltering. Harass is defined by the Service as intentional or negligent actions that create the likelihood of injury to listed species to such an extent as to significantly disrupt normal behavior patterns which include, but are not limited to, breeding, feeding, or sheltering. Incidental take is defined as take that is incidental to, and not the purpose of, the carrying out an otherwise lawful activity. Because oil spills are considered an unlawful activity, take is not authorized under this BO for those events. Impacts to federally listed species as a result of an oil spill would be addressed in a separate consultation or in a subsequent NRDAR case. Under the terms of section 7(b)(4) and section 7(o)(2), taking that is incidental to and not intended as part of the agency action is not considered to be prohibited taking under the Act provided that such taking is in compliance with the terms and conditions of the Incidental Take Statement.

AMOUNT OR EXTENT OF TAKE ANTICIPATED

The Service does not anticipate the proposed action will incidentally take any listed species under our jurisdiction.

CONSERVATION RECOMMENDATIONS

Section 7(a)(1) of the Act directs federal agencies to utilize their authorities to further the purposes of the Act by carrying out conservation programs for the benefit of endangered and threatened species. Conservation recommendations are discretionary agency activities to minimize or avoid adverse effects of a proposed action on listed species or critical habitat, to help implement recovery plans, or to develop information.

1. <u>Oil Spill and Other Hazardous Emergency Contingency Plans</u> – In order to reduce impacts on threatened and endangered species, we recommend that the BOEM/BSEE continue to require the petroleum industry to prepare adequate hazardous spill contingency plans for all activities. This should include the strategic placement of appropriate spill cleanup equipment, personnel training in non-intrusive cleanup techniques, and demonstration of response commitment, capabilities, and implementation. The response plans should include, but not be limited to, identification of the identified species' habitat (including designated critical habitat), oil spill trajectory modeling to ascertain the time for oil to reach the habitats, implementation plans for protection of the species and their habitats in case of an oil spill, cleanup methods, best management practices, and state and Federal resource manager contacts.

BOEM/BSEE should ensure that oil spill contingency plans be prepared to meet the requirements of the agency, identify specific locations on nesting beaches utilized by sea turtles, and include provisions for removal of sea turtle eggs from beaches that are imminently expected to receive spilled oil. The eggs should be relocated and incubated and the hatchlings released in an uncontaminated area. The plan should also name qualified and permitted rehabilitators to handle oiled and/or stranded sea turtles.

Further, the Service recommends that the BOEM/BSEE coordinate annually with the USCG to assure that the spill contingency and response plans contain current and up-to-date sensitive areas information regarding Kemp's ridley and loggerhead sea turtles; Cape Sable seaside sparrow; Mississippi sandhill crane; piping plover; red knot; roseate tern; whooping crane; wood stork; Alabama, Choctawhatchee, Perdido Key, and St. Andrew beach mice; and West Indian manatee.

2. Aircraft Impacts - To minimize impacts to endangered and threatened species, we recommend that the BOEM/BSEE advise the lessees' aircraft operators to adhere to the above-specified altitude restrictions over NWRs and parks (including national seashores) and other ecologically sensitive areas (i.e., designated critical habitats, etc.).

3. Marine Debris and Trash - To further minimize impacts to endangered and threatened species, we recommend the BOEM/BSEE continue to enforce their regulations regarding marine debris disposal from offshore oil and gas operations.

4. Information Needs - Additional study is needed regarding the actual sea turtle nesting that occurs on the Chandeleur Islands and the genetics of nesting sea turtles throughout the EPA, CPA and WPA. Information is also still needed regarding chemical and physical impacts to sea turtles from oil dispersants and/or oil as well as sea turtle behavior in regard to oil spill slick avoidance or ingestion of weathered oil products.

   Aerial and/or ground surveys are needed for the threatened piping plover. Those surveys would be designed and conducted to obtain better information on the occurrence and distribution of those species within the project area. We also encourage the BOEM/BSEE to participate in and/or fund long-term shorebird migration studies along the Gulf Coast. Information is needed regarding the migratory pathways of red knots and migratory behaviors of both piping plovers and red knots, especially when migration and wintering grounds are disturbed. That information would benefit our knowledge regarding piping plover and red knot migration biology and aid in future section 7 consultations.

5. Coordination - For the Service to be kept informed of actions minimizing or avoiding adverse effects or benefitting listed species or their habitats, the Service requests notification of the implementation of any conservation recommendations.

REINITIATION NOTICE - CLOSING STATEMENT

This concludes formal consultation on the actions outlined in the request. As provided in 50 CFR §402.16, reinitiation of formal consultation is required where discretionary BOEM/BSEE involvement or control over the action has been retained (or is authorized by law) and if: (1) the amount or extent of incidental take is exceeded (in this consultation no incidental take is authorized); (2) new information reveals effects of the action that may affect listed species or critical habitat in a manner or to an extent not considered in this opinion; (3) the action is subsequently modified in a manner that causes an effect on the listed species or critical habitat not considered in this opinion; or (4) a new species is listed or critical habitat designated that may be affected by the action. In instances where the amount or extent of incidental take is exceeded, any operations causing such take must cease pending reinitiation.

We appreciate the cooperation exhibited by your agency, especially the Gulf Coast Regional Office in New Orleans, during this consultation. We look forward to future coordination with BOEM/BSEE in the conservation of endangered and threatened species in the Gulf of Mexico and adjacent coastal habitats. If your staff has questions regarding this consultation or other endangered and threatened species issues please contact Karen Soileau at 337/291-3132.

Sincerely,

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