Loggerhead Sea Turtle
(*Caretta caretta*)
Northwest Atlantic Ocean DPS
5-Year Review:
Summary and Evaluation

2023

National Marine Fisheries Service
Office of Protected Resources
Silver Spring, Maryland
and
U.S. Fish and Wildlife Service
Southeast Region
Florida Ecological Services Office
Jacksonville, Florida

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5-YEAR REVIEW

Loggerhead sea turtle, Northwest Atlantic Ocean DPS (*Caretta caretta*)

1.0 GENERAL INFORMATION

1.1 Reviewers

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1.2 Methodology

The purpose of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1531 *et seq*.) is to provide a means to conserve and recover endangered and threatened species and ecosystems upon which they depend. Under the ESA, the National Marine Fisheries Service (NMFS) and U.S. Fish and Wildlife Service (USFWS), together "we" or "the Services," share responsibility to conserve sea turtles (as described in the 2015 revision of the Memorandum of Understanding Defining the Roles of USFWS and NMFS in Joint Administration of the Endangered Species Act of 1973 as to Sea Turtles). NMFS has jurisdiction for sea turtles in the marine environment, and USFWS has jurisdiction for sea turtles in the terrestrial habitat. One of our responsibilities under the ESA is to conduct a review of each listed species at least every 5 years to determine whether its endangered or threatened status should be changed or removed (i.e., 5-year review, 16 U.S.C. 1533(c)(2)). The ESA requires us to make these determinations solely on the basis of the best scientific and commercial data available (16 U.S.C. 1533(b)(1)(A)). Under the ESA, the definition of species includes any subspecies of fish or wildlife or plants, and any distinct population segment (i.e., DPS) of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S.C. 1532). In 2011, after a status review of the species (the Status Review Report; Conant *et al*. 2009), the Services identified nine loggerhead sea turtle DPSs, including the Northwest Atlantic Ocean DPS (i.e., NW Atlantic DPS or the DPS; 76 FR 58868; September 22, 2011), in accordance with the Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the ESA (i.e., the DPS Policy; 61 FR 4722, February 7, 1996).

In 2019, we initiated the 5-year review for this DPS. To compile the best available scientific and commercial data on the DPS, we solicited relevant information from other Federal agencies, States, Territories, Tribes, foreign governments, academia, nonprofit organizations, industry groups, and individuals by publishing a request in the Federal Register (84 FR 70958; December 26, 2019). We received two comments, from the National Park Service and from the Southern Environmental Law Center; these comments are available to view at Regulations.gov (NOAA-NMFS-2019-0150). To provide the best available information, USFWS commissioned a literature review, which was completed by Dr. Joseph B. Pfaller of the Caretta Research Project (Savannah, Georgia). Relevant information from that report is incorporated into sections below; please see Pfaller (2021) for additional details. In addition, we reviewed and included information provided in the 2019 Assessment of Progress Toward Recovery (Bolten *et al.* 2019). We then searched for relevant new information on the DPS, its biology and habitat, and threats to its existence that have become available since the DPS listing in 2011.

We compiled, reviewed, and evaluated all information. We did not conduct new empirical studies because the ESA requires the use of the best *available* scientific and commercial information. Instead, we first reviewed newly available information relevant to the DPS determination, following the DPS Policy. Under this policy, a DPS must be discrete and significant relative to its species. We asked whether new data supported or refuted our previous determinations of discreteness and significance. Next, we considered the biology and habitat of the DPS. We identified information that has become available since the publication of the Status Review Report in 2009. We also reviewed the best available information on abundance and trends, genetics, spatial distribution, and habitat conditions. We then assessed threats to the DPS by identifying and evaluating the ESA section 4(a)(1) factors (i.e., the five factor analysis; 16 U.S.C. 1533(a)(1)):

1. Present or threatened destruction, modification, or curtailment of habitat or range
2. Overutilization for commercial, recreational, scientific, or educational purposes
3. Disease or predation
4. Inadequacy of existing regulatory mechanisms
5. Other natural or manmade factors affecting its continued existence

Because abundance and trends are influenced by past threats, we focused on present threats. For each factor, we evaluated the magnitude of the threat and how it would impact the DPS. We synthesized the above information to assess the DPS's status, identifying factors that weighed most heavily in our evaluation. Based on this information, we provide a recommendation on the status of the DPS.

1.3 Background

1.3.1 Federal Register Notice

FR notice: 84 FR 70958
Date listed: December 26, 2019
Purpose: NMFS gave notice of our initiation of 5-year reviews of the NW Atlantic DPS and the foreign loggerhead DPSs; we requested relevant information from the public.

1.3.2 Listing History

<u>Original Listing</u>
FR notice: 43 FR 32800
Date listed: July 28, 1978
Entity listed: Loggerhead sea turtle (*Caretta caretta*)
Classification: Threatened

<u>Revised Listing</u>
FR notice: 76 FR 58868
Date listed: September 22, 2011
Entity listed: Loggerhead sea turtle (*Caretta caretta*), NW Atlantic DPS
Classification: Threatened

1.3.3 Associated Rulemakings

<u>4(d) Rules</u>
FR notice: 64 FR 14069

Date: March 23, 1999
Purpose: Applied section 9 prohibitions (16 U.S.C. 1538) to threatened sea turtles

FR notice: 64 FR 14070
Date: March 23, 1999
Purpose: Identified exceptions to section 9 prohibitions

Critical Habitat Rules
FR notice: 79 FR 39855
Date: July 10, 2014
Conclusion: NMFS designated critical habitat for the NW Atlantic DPS: 38 occupied marine areas containing nearshore reproductive habitat, winter areas, breeding areas, constricted migratory corridors, and *Sargassum* spp. habitat.

FR notice: 79 FR 39756
Date: July 10, 2014
Conclusion: USFWS designated critical habitat for the NW Atlantic DPS: 1,102 km of coastal shoreline in the States of North Carolina, South Carolina, Georgia, Florida, Alabama, and Mississippi containing suitable nesting beach habitat.

1.3.4 Review History

- In 1985, NMFS conducted the first 5-year review of the species, concluding that of 52 nesting populations evaluated throughout the Atlantic, Pacific, and Indian Oceans, 33 were thought to be declining, 18 were unknown, and only one – the U.S. southeast (SE) Atlantic – was thought to be increasing. Although the United States had implemented protective regulations and commercial harvest of eggs had decreased, many threats continued both domestically and abroad. NMFS determined that information was insufficient to assess whether a change in status was warranted.

- In 1991, USFWS conducted a 5-year review of many species, including the loggerhead sea turtle (56 FR 56882, November 6, 1991). USFWS requested new or additional information on the species and indicated that it would propose a change in status if warranted by the data received. Following the review, USFWS did not recommend a change in status.

- In 1995, the Services conducted a joint 5-year review (Plotkin 1995). Although we identified a need for further study of U.S. loggerhead population structure, we did not recommend a change in the status of the species.

- In 2007, we conducted a joint 5-year review on the loggerhead sea turtle (NMFS and USFWS 2007). We identified new information on statistically significant genetic population structure within and among ocean basins, based on the analyses of tissue samples collected at nesting beaches and foraging grounds. In addition, new information was available on age at first reproduction, survival rates, and in-water turtles that suggested discreteness among populations. Although we did not recommend a change in status at that time, we recommended further analysis and review to apply the DPS Policy to the species (NMFS and USFWS 2007).

- On November 15, 2007, the Center for Biological Diversity and Oceana petitioned us to identify the western North Atlantic loggerhead population as a DPS, list it as endangered, and designate critical habitat. On March 5, 2008, we found that the petition presented substantial scientific information indicating that the petitioned actions may be warranted (73 FR 11849) and conducted the Status Review Report (Conant *et al*. 2009). Following that review, the NW Atlantic Ocean Loggerhead DPS was listed as threatened in 2011 (76 FR 58868).

- On December 26, 2019, NMFS gave notice of our initiation of a 5-year review of the NW Atlantic Ocean Loggerhead DPS and foreign loggerhead DPSs; we requested relevant information from the public (84 FR 70958).

1.3.5 Species' Recovery Priority Number

NMFS' Recovery priority number: 5C (NMFS 2022), defined as follows in the Recovery Priority Guidelines (84 FR 18243; April 30, 2019):

- low demographic risk (relative to other listed species);
- well understood major threats;
- low to high U.S. jurisdiction, authority, or influence to address major threats; and
- high certainty that management or protective actions will be effective.

USFWS' Recovery priority number: 9C (USFWS 2018), as defined in the Recovery Priority Guidelines (48 FR 43098; September 21, 1983), which reflects a Distinct Population Segment with:

- moderate degree of threat;
- high recovery potential; and
- when recovery may be in conflict with construction or other development projects or other forms of economic-activity.

1.3.6 Recovery Plan

Name of plan: Recovery Plan for the NW Atlantic Population of the Loggerhead Turtle (*Caretta caretta*) [Created prior to the listing of the DPS]
Date issued: Second Revision 2008
2019 Progress Report: Assessment of Progress Toward Recovery (Bolten *et al.* 2019)

2.0 REVIEW ANALYSIS

2.1 Application of the DPS Policy

2.1.1 Is the species under review a vertebrate?

__x__**Yes**
_____**No**

2.1.2 Is the species under review listed as a DPS?

__x__**Yes**
_____**No**

2.1.3 Was the DPS listed prior to 1996?

_____**Yes**
__x__**No**

2.1.4 Is there new information regarding the application of the DPS policy?

__x__**Yes**
_____**No**

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Cheloniidae
Genus: *Caretta*
Species: *caretta*
DPS: NW Atlantic Ocean
Common name: Loggerhead sea turtle

The taxonomy of the species has remained consistent and unchallenged since 1962 (Dodd 1988) and was summarized in the Status Review Report (Conant *et al*. 2009). After reviewing the Status Review Report and the genetic and satellite tracking data that have become available since its publication, we confirmed the NW Atlantic DPS (Figure 1) to be reproductively and geographically discrete from all other loggerhead DPSs and significant to the species. The following paragraphs summarize these newly available data.

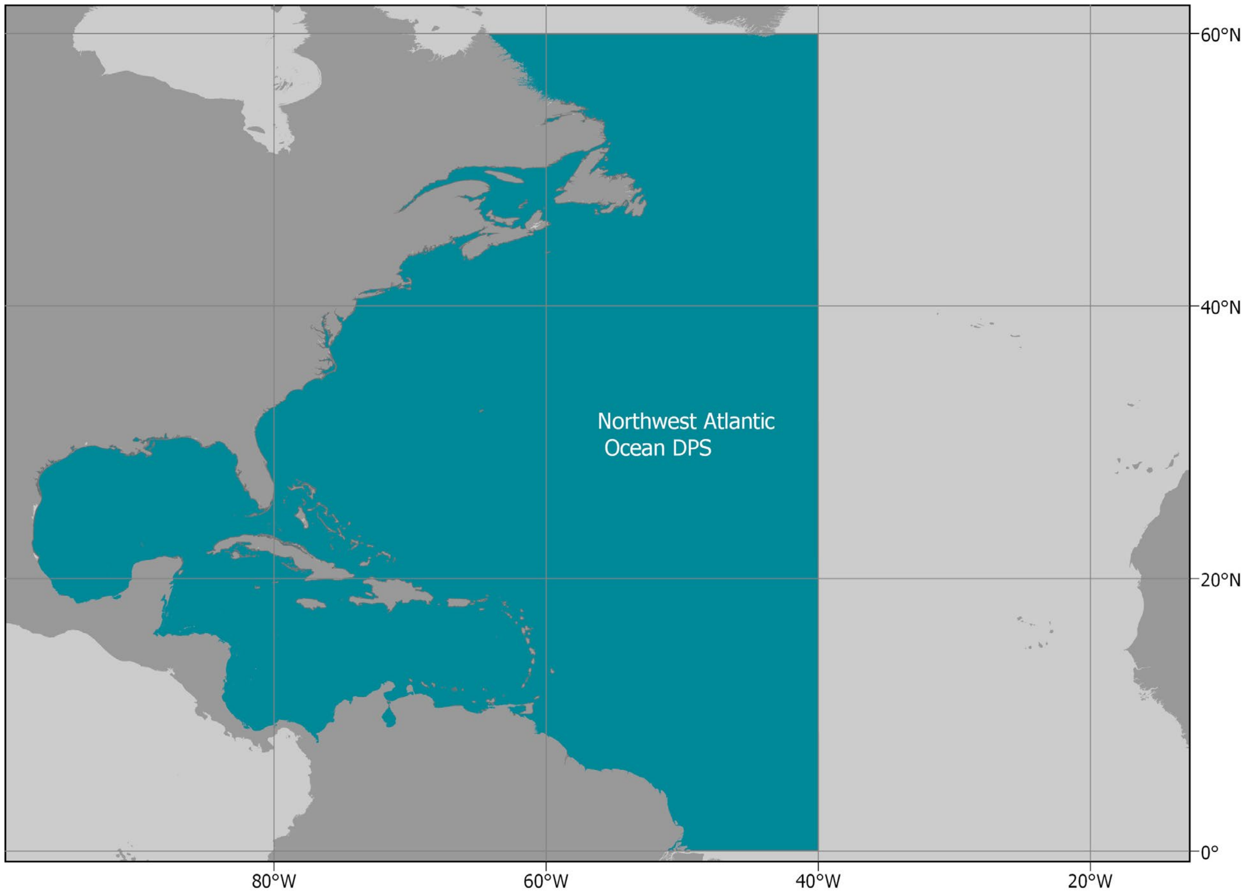

Figure 1. Boundaries of the NW Atlantic DPS
The DPS is defined as loggerhead sea turtles originating from the NW Atlantic Ocean north of the Equator, south of 60° N. latitude, and west of 40° W. longitude. The range of the DPS potentially includes the entire North Atlantic Ocean during pelagic developmental life history phases.

As described in the Status Review Report, loggerhead turtles nesting on beaches of the NW Atlantic Ocean are genetically isolated (i.e., discrete) from all other nesting populations, based on population comparisons of 380 base pair (bp) mitochondrial DNA (mtDNA) control region sequences. Since the publication of the Status Review Report, additional genetic data have become available. Shamblin *et al.* (2014) analyzed longer (760 to 817 bp) mtDNA sequences and sampled additional nesting beaches (Cape Verde, Brazil, South Africa, and Oman) to evaluate genetic differentiation among six DPSs in the Atlantic and Indian Oceans and the Mediterranean Sea. With this added resolution, Shamblin *et al*. (2014) identified high magnitude and statistically significantly population structure ($F_{ST} = 0.349$, $p < 0.0001$) among nesting beaches, with differences among the six DPSs accounting for the majority of genetic partitioning. Additional, finer-scale substructure was detected within the NW Atlantic DPS, see section 2.3.1.4 on spatial distribution. We conclude that these data provide further support for the genetic discreteness of the NW Atlantic DPS nesting population.

Unlike mtDNA, which is maternally inherited, nuclear DNA markers (e.g., microsatellites) allow evaluation of male-mediated gene flow among DPSs. Using microsatellites, Bowen *et al.* (2005)

found statistically significant genetic differentiation (F_{ST} = 0.04-0.08; $p < 0.05$) between samples collected from North and South (e.g., Bahia, Brazil) Atlantic DPSs. Similarly, Carreras *et al.* (2011) found statistically significant genetic differentiation (F_{ST} = 0.03; $p < 0.001$) between samples collected from the Mediterranean and North Atlantic DPSs. Therefore, male-mediated gene flow among DPSs is unlikely because mating occurs while migrating toward or in the waters off natal nesting beaches (Lasala *et al.* 2018). Genetic assignment tests of 850 loggerhead turtles bycaught in western North Atlantic fisheries revealed that all but three likely originated from NW Atlantic nesting beaches; two had haplotypes common in the South Atlantic DPS, and one originated in Cape Verde, belonging the Northeast (NE) Atlantic DPS (Stewart *et al.* 2018). While Piovano *et al.* (2011) found 40 of 73 turtles foraging in the Mediterranean Sea to have Atlantic origins (mainly from the NW Atlantic DPS), all individuals measured less than 80 cm curved carapace length (CCL), and as juveniles or subadults, could not contribute to gene flow between the DPSs.

Satellite tracking studies provide further evidence for the lack of gene flow among the DPSs because large juvenile and adult loggerheads of this DPS appear to use coastal waters of the NW Atlantic Ocean for foraging. Winton *et al*. (2018) compiled loggerhead tracking data from 2004 to 2016 (N = 271 large juveniles over 51 cm straight carapace length (SCL) and adults), which demonstrated year-round residency in NW Atlantic waters, with the highest densities in shelf waters from Florida to North Carolina. Other studies confirmed this residency for mature females and males. Between 2008 and 2017, Evans *et al*. (2019) tracked post-nesting females (N = 45) from the Archie Carr National Wildlife Refuge and found that all remained within NW Atlantic waters. Between 1998 and 2008, Hawkes *et al.* (2011) tracked 68 post-nesting loggerheads from North Carolina, South Carolina, and Georgia beaches; they found that adult females generally do not leave U.S. waters and remain within the continental shelf (200 m depth). In 2006 and 2007, Pajuelo *et al.* (2012) tracked 29 males off a Florida nesting beach and found that they remained in NW Atlantic waters. Based on these data, we conclude that the DPS is genetically and geographically isolated from other DPSs at both nesting and adult foraging areas. Therefore, data published since the Status Review Report provide additional support for the discreteness of the NW Atlantic DPS.

2.2 Recovery Criteria

2.2.1 Does the species have a final, approved recovery plan?

__x__**Yes**
_____**No**

Prior to the listing of the DPS, the Services published the Recovery Plan for the NW Atlantic Population of the Loggerhead Sea Turtle (*Caretta caretta*) Second Revision (NMFS and USFWS 2008), which applies to the population that was later listed as a threatened DPS.

2.2.2 Do recovery criteria reflect the best available and most up-to-date information?

__x__**Yes**
_____**No**

In 2019, the NW Atlantic Loggerhead Recovery Team reconvened to review the Recovery Plan and to assess progress toward recovery. The Team concluded that the 2008 Recovery Plan continues to be the appropriate roadmap to recovery of the DPS (Bolten *et al*. 2019). We agree.

2.2.3 Recovery Units, demographic recovery criteria, and discussion of whether each demographic criterion has been met.

The 2008 Recovery Plan documents 13 recovery objectives, 3 demographic criteria, and 20 listing factor criteria for each of the 5 recovery units. Recovery Units include:

1. **Northern Recovery Unit:** loggerheads originating from nesting beaches from the Florida-Georgia border through southern Virginia (the northern extent of nesting range).
2. **Peninsular Florida Recovery Unit:** loggerheads originating from nesting beaches from the Florida-Georgia border through Pinellas County on the west coast of Florida, excluding the islands west of Key West, Florida.
3. **Dry Tortugas Recovery Unit:** loggerheads originating from nesting beaches throughout the islands located west of Key West, Florida, because these islands are geographically separated from other recovery units.
4. **Northern Gulf of Mexico (GoM) Recovery Unit:** loggerheads originating from nesting beaches from Franklin County on the NW Gulf coast of Florida through Texas (the western extent of U.S. nesting range).
5. **Greater Caribbean Recovery Unit:** loggerheads originating from all other nesting assemblages within the Greater Caribbean (Mexico through French Guiana, The Bahamas, Lesser Antilles, and Greater Antilles).

These RUs were based on genetic data available in 2008 and geographic and geopolitical boundaries that influence their exposure to threats and recovery efforts. In 2019, the NW Atlantic Loggerhead Recovery Team reviewed progress toward the criteria. They concluded that although progress has been made, the RUs have not yet met most demographic criteria and many of the listing factor criteria have not yet been addressed (Bolten *et al*. 2019). The 20 listing factor recovery criteria align with the five ESA 4(a)(1) listing factors or threats. For our full analysis of these threats, please see Section 2.3.2. Only three of the 20 listing factors criteria have been met: conservation of 1,581 km of loggerhead nesting beaches; ecologically sound predator control programs; and a strategy to recognize, respond to, and investigate mass/unusual mortality or disease events (Bolten *et al*. 2019). We agree with their assessment.

When considering the three demographic criteria, our analysis during this 5-year review has led to the determination that these also have not been met. The first demographic recovery criterion (Number of Nests and Number of Nesting Females) sets an annual rate of increase (specific to each RU, with 95% statistical confidence, $p = 0.05$) over one generation (50 years) and requires that this increase in nests results from corresponding increases in number of nesting females (estimated from nests, clutch frequency, and remigration interval, (i.e., the average number of years between consecutive nesting seasons)). This criterion has not been met because none of the RUs have been monitored for 50 years nor have any met the target annual rate of increase (Table 1). Nest trends of the Peninsular Florida RU, which hosts the majority of nesting within the DPS, have not increased over 30 years of monitoring. The much smaller Northern RU has demonstrated some progress toward its goal, with a statistically significant 1.7% rate of increase over 37 years, which is less than the 2% criteria; genetic analyses of all nests laid in the Northern RU indicated that the number of annual nests since 2010 significantly correlates ($p = 0.004$) to the number of annual nesting females (Shamblin *et al*. 2017; Georgia Department of Natural Resources and University of Georgia, unpublished data 2021). Nesting rates from the Northern GoM RU were not significantly different from zero. Data were insufficient or unavailable from the Dry Tortugas and Greater Caribbean RUs. Despite some encouraging data from the Northern RU, we conclude that the first demographic criterion has not been met.

Table 1. Recovery Plan demographic recovery criteria (i.e., nest trend criteria)
Comparison of the 2008 demographic recovery criteria (NMFS and USFWS 2008) and recent nesting data at index beaches (Bolten *et al*. 2019). The p-value indicates statistical significance, and only the Northern Recovery Unit data shows a significant positive trend.

Recovery Unit	Criteria: Annual Rate of Increase over 50 Years (target number of nests annually)	Estimates of Annual Rate of Increase (p-value) over X Years (year span measured)
Northern	≥ 2% (14,000 nests: 2,000 in NC, 9,200 in SC, and 2,800 in GA)	1.3% (p = 0.04) 37 (1983–2019)
Peninsular Florida	1% (106,100 nests)	No significant trend (p = 0.61) 30 (1989–2018)
Dry Tortugas	≥ 3% (≥1,100 nests)	Insufficient data
Northern GoM	≥ 3% (≥4,000 nests)	No significant trend (p = 0.17) 22 (1997–2018)
Greater Caribbean	Any % (≥100 nests)	Insufficient data

The second demographic recovery criterion (Trends in Abundance on Foraging Grounds) requires that "a network of in-water sites, both oceanic and neritic, distributed across the foraging range is established and monitoring is implemented to measure abundance. There is statistical confidence (95%) that a composite estimate of relative abundance from these sites is increasing for at least one generation." To address this criterion, Bolten *et al*. (2019) described the need for dedicated aerial surveys targeting sea turtles and covering large geographic regions. While two large-scale surveys are underway (Atlantic Marine Assessment Program for Protected Species (AMAPPS) and Gulf of Mexico Marine Assessment Program for Protected Species (GoMAPPS)), these surveys cover multiple taxa and have not been specifically evaluated for their appropriateness to generate long-term trends in abundance for sea turtles. Furthermore, establishment and evaluation of a coordinated network of index in-water sites are still needed. Therefore, Bolten *et al.* (2019) concluded that this criterion has not been met, and we agree.

The third demographic recovery criterion (Trends in Neritic Stranding Relative to In-Water Abundance) requires that "stranding trends are not increasing at a rate greater than the trends in in-water relative abundance for similar age classes for at least one generation." While stranding data are collected, trend analyses are not yet available. As described by Bolten *et al*. (2019), geographically broad, robust data analyses of strandings are still needed. We conclude that this criterion has not been met.

2.3 Updated Information and Current Species Status

2.3.1 Biology and Habitat

Like all sea turtles, loggerheads of the NW Atlantic DPS exhibit a complex life cycle that contains several life stages (i.e., hatchling, juvenile, and adult, for the purposes of this review), occurring across wide-spread and diverse habitats. Nesting occurs on beaches within the SE United States and the Wider Caribbean Region. Foraging occurs at different locations in waters of the North Atlantic Ocean, depending on life stage and foraging strategy. Here, we provide a brief description of habitat use, prey, and foraging strategies for each life stage.

After emerging *en masse* from their nests at night, hatchlings crawl seaward using visual cues, toward the brighter horizon of the open ocean and away from the dark, elevated silhouettes of dunes and vegetation (Pankaew and Milton 2018). After hatchlings enter the sea, they begin a 24 to 36-hour swim frenzy, orienting into waves to reach offshore currents (DuBois *et al.* 2021). They likely imprint on the magnetic signature of the coastal area so that they can return as adults to waters off their natal beaches for reproduction (Lohmann and Lohmann 2019).

Post-hatchling loggerheads are primarily associated with consolidated patches of floating material, especially *Sargassum* spp., which becomes concentrated in the GoM and especially along the western wall of the Gulf Stream (Witherington *et al.* 2012). At this stage, they are likely generalist and opportunist omnivores (Witherington *et al.* 2012), using visual and olfactory cues to actively search for food (Warraich *et al.* 2020). Post-hatchlings grow rapidly (Avens *et al.* 2013), exhibit allometric growth (width increases faster than their length), and develop prominent spines on scutes of their carapace, which help exceed the gape of predators to reduce predation risk as they age (Salmon and Scholl 2014; Marn *et al.* 2015; Pate and Salmon 2017).

Young juvenile loggerheads inhabit oceanic waters spanning the width of the North Atlantic Ocean and Mediterranean Sea. They are initially carried offshore by the Gulf Stream (Mansfield *et al.* 2012; Mansfield *et al.* 2014). Thereafter, some depart currents associated with the North Atlantic Subtropical Gyre to exploit favorable foraging areas and thermal niches within the Sargasso Sea (Mansfield *et al.* 2012; Mansfield *et al.* 2014). Others traverse large areas of the North Atlantic Ocean to inhabit oceanic waters of the Mediterranean Sea and off the coasts of the Canary Islands, Azores, Madeira, France, United Kingdom, Ireland, and Canada (Harris *et al.* 2010; McCarthy *et al.* 2010; Pajuelo *et al.* 2010; Monzon-Arguello *et al.* 2012; Clusa *et al.* 2014; Botterell *et al.* 2020). Within these waters, high-usage areas are associated with oceanographic features, such as weak surface currents, that support concentrated prey availability (Freitas *et al.* 2018). They use active and passive transport to search for odors or oceanographic features associated with increased ocean productivity and prey availability, such as high chlorophyll a and shallower bathymetry (Endres and Lohmann 2012; Freitas *et al.* 2018; Chambault *et al.* 2019). Their movements also correspond with broad seasonal changes and fine-scale differences in sea surface temperature that are likely associated with thermoregulation (Chambault *et al.* 2019; Vandeperre *et al.* 2019).

After several years inhabiting oceanic waters spanning the width of the North Atlantic Ocean and Mediterranean Sea, juveniles typically return to the waters of the NW Atlantic Ocean. These older juveniles may undergo an ontogenetic, oceanic-to-neritic habitat shift that is primarily driven by factors independent of growth rate, including hormone regulation, metabolic needs, or allometric relationships (Ramirez *et al.* 2017); however, this transition is not obligate, permanent (i.e., some return to oceanic habitats; Mansfield and Putman 2013), nor fixed to a certain body size or age class (Winton *et al.* 2018). At least two major ontogenetic shift patterns have been identified: discrete shifts, which are completed within one year; and facultative shifts which are completed over multiple (up to 5) years during which the juveniles forage in oceanic and neritic habitats (Ramirez *et al.* 2015). Generally, there is no difference in growth patterns between the shift patterns despite apparent energetic advantages associated with a neritic habitat (Ramirez *et al.* 2017). These transitions may occur where major oceanic currents approach or enter the neritic zone, such as along the continental shelf of the eastern United States (Ceriani *et al.* 2017). Returns to an oceanic habitat may be opportunistic or facilitated by prevailing oceanic currents. Some juveniles feed on nearshore benthic prey in the summer and offshore pelagic prey (mostly jellyfish) in the winter (McClellan *et al.* 2010; Ramirez *et al.* 2015; Smolowitz *et al.* 2015; Ramirez *et al.* 2017). Others forage on benthic and pelagic prey concurrently, even within the same dive (Smolowitz *et al.* 2015; Patel *et al.* 2016). Pelagic prey includes Lion's mane jellies (*Cyanea capillata*), comb jellies (Ctenophora), and salps (Salpidae). They eat many species of benthic prey including hermit crabs (Paguroidea), rock crabs (*Cancer irroratus*), and Atlantic sea scallops (*Placopecten magellanicus*). Within neritic habitats, juveniles commonly forage in nearshore coastal waters, coastal inlets, sounds, bays, estuaries, lagoons and along the continental shelf during the spring, summer, and fall months from Cape Cod, south to Florida, and into the Gulf of Mexico; during winter, they are found off the coast from North Carolina to Florida. Large juveniles may occur in the same foraging and resting habitats as adults. Resting involves wedging under or against reefs and ledges and has been documented for up to 144 minutes at twilight or night (Auster *et al.* 2020).

Similar to later juvenile life stages, adult loggerheads demonstrate a variety of habitat use patterns in the NW Atlantic that, while mainly neritic, may include oceanic foraging (Mansfield and Putman 2013). Generally, adults are found in deeper, more offshore areas in the Mid-Atlantic Bight, South Atlantic Bight, subtropical NW Atlantic, Greater Caribbean, and Gulf of Mexico. Variation in foraging areas is primarily associated with RUs, with the Northern RU primarily using Mid-Atlantic Bight foraging areas and the Peninsular Florida, Dry Tortugas, and Northern GoM RUs primarily using subtropical NW Atlantic and eastern GoM foraging areas (Pfaller *et al.* 2020b). Within the Mid-Atlantic Bight, some adults and large juveniles forage on benthic prey in neritic habitats from New York to Virginia in the summer, then move south and to shelf waters from Florida to North Carolina in the winter (Winton *et al.* 2018; Patel *et al.* 2021). Others occupy year-round foraging sites off North Carolina, including estuarine and neritic waters (McNeill *et al.* 2020). Year-round residents and seasonal migrants in this area use a narrow migratory corridor within the South Atlantic Bight to access the waters off nesting beaches (Griffin *et al.* 2013). Post-nesting females and post-mating males (Pajuelo *et al.* 2012) from the three Florida RUs take up residence in discrete foraging areas, including the GoM (Hart *et al.* 2020; Phillips *et al.* 2021), east coast of Florida (Evans *et al.* 2019), and Bahamas Banks (Ceriani *et al.* 2017). They use migratory corridors in the eastern GoM, along the Florida Keys, or through the Florida Straits to the Bahamas (Iverson *et al.* 2020). Based on satellite telemetry and stable isotope analysis of 749 post-nesting females at the Archie Carr National Wildlife Refuge, females foraging in southern areas appear to produce more offspring than those foraging in northern areas (Ceriani *et al.* 2017).

Adults migrate to the waters off their natal beach to mate. Females nest every 1 to 7 years and exhibit relatively strong nest-site fidelity (Shamblin *et al.* 2017). In a genetic recapture study of the Northern RU, 65% of 1,770 females that nested in 2010 remigrated at least once over the 5-year recapture period, with a mean observed remigration interval of 2.67 (±0.89 SD) years and a median inter-seasonal displacement of 1.84 km (Shamblin *et al.* 2021). Nesting females of the Northwest Atlantic DPS prefer steeply sloped, coarse-grained beaches prone to high rates of erosion (Lamont and Houser 2014). Nesting begins in April, peaks in June or July, and ends in August or September (Hart *et al.* 2010; Tucker 2010; Monk *et al.* 2011; Hart *et al.* 2013; Pfaller *et al.* 2013; Phillips *et al.* 2014). Females remain in shallow, nearshore waters directly off nesting beaches during the internesting interval (Hart *et al.* 2010; Hart *et al.* 2013; Scott *et al.* 2013). They nest one to seven times in a season, with an internesting interval of approximately 14 days. Clutch sizes range from 95 to 130 eggs, with an incubation duration of 42 to 75 days. Temperature dependent sex determination occurs within the middle third of development, with nest temperatures over 29°C resulting in female hatchlings (Wyneken and Lolavar 2015; Wyneken and Salmon 2020).

2.3.1.1 Abundance

It is difficult to estimate overall abundance for sea turtle populations because individuals spend most of their time in water, where they are difficult to count, especially considering their large range and use of many different and distant habitats. Females, however, converge on their natal beaches to lay eggs, and nests are easily counted. As described by Ceriani *et al.* (2019), nest counts are used as an index of abundance and population trends; however, they do not provide a direct index of adult female population abundance because females typically lay more than one nest per year (measured as clutch frequency, the average annual number of clutches) and most do not reproduce every year (measured as the remigration interval, the average number of years between consecutive nesting seasons). Analyzing the Florida nest data over 30 years, Ceriani *et al.* (2019) recommended caution when using nest counts as a direct proxy for adult female population status due to the uncertainty in these reproductive parameters because we cannot distinguish between the abundance of nesting females and their cumulative reproductive effort. Due to these uncertainties, we provide available nest counts (Table 2) but do not convert these data into a total estimate of nesting females. Mississippi data include nests discovered either opportunistically, during bird surveys, or in association with monitoring during Army Corps of Engineers projects on beaches (USFWS unpublished data 2022). Surveys were not conducted in Louisiana from 2016 to 2021; however, the Coastal Protection and Resource Authority and Louisiana Department of Wildlife and Fisheries, in coordination with Breton National Wildlife Refuge, conducted aerial surveys for sea turtle nesting evidence on Chandeleur Islands, Louisiana between May 27 and August 29, 2022. Fifty-three crawls were detected, and preliminary review indicates some of these are Kemp's ridley and loggerhead sea turtle nests. Final determinations will be made at a later time. (Louisiana Department of Wildlife and Fisheries unpublished data 2022). The Quintana Roo data (Greater Caribbean RU) include nests on index and non-index beaches and were provided by the Committee to Protect Sea Turtles in Quintana Roo (CPTMQROO). Nest count data are not available for other beaches within the Greater Caribbean RU; however, Eckert and Eckert (2019; WIDECAST.org) reported binned average annual crawls between 2007 and 2018, based on expert opinion (Table 3). These data, which do not include U.S. and Quintana Roo nesting beaches, cannot be compared to the nest count data because they are based on total crawls (nesting emergences and non-nesting emergences); however, they provide a relative sense of nesting activity in the Greater Caribbean RU.

Table 2. Available nest count data by state

State	2016 Nests	2017 Nests	2018 Nests	2019 Nests	2020 Nests	Reference
North Carolina	1,622	1,195	765	2,293	1,331	North Carolina Wildlife Resources Commission, Seaturtle.org 2020
South Carolina	6,446	5,231	2,762	8,774	5,552	South Carolina Department of Natural Resources, Seaturtle.org 2020
Georgia	3,289	2,155	1,735	3,950	2,786	Georgia Dept. of Natural Resources, Seaturtle.org 2020
Florida	122,707	96,912	91,451	106,373	105,185	FWC 2020
Alabama	233	178	91	113	97	Share the Beach 2020
Mississippi	1	10	3	1	1	USFWS unpublished data 2022
Texas	6	8	6	8	3	D. Shaver, National Park Service, unpublished data 2020
Virginia	4	10	8	10	12	Virginia Department of Wildlife Resources, 2022
Quintana Roo	5,367	3,142	4,681	3,639	3,935	CPTMQROO unpublished data 2022
TOTAL	**139,675**	**108,841**	**101,502**	**125,161**	**118,902**	

Table 3. Wider Caribbean sea turtle crawl data
(Eckert and Eckert 2019; WIDECAST.org)

Binned Average Annual Crawls per Beach	Estimated Number of Nesting Beaches	Estimated Annual Crawls
Unknown	20	20+
<25	272	272 – 6,800
25-100	32	800 – 3,200
100-500	13	1,300 – 6,500
500-1000	4	2,000 – 4,000
TOTAL	**342**	**4,392 – 21,520**

Based on the estimates in Table 2, the NW Atlantic DPS hosts more than 110,000 nests annually. For the IUCN Red List assessment of the NW Atlantic loggerhead, Ceriani and Meylan (2017) estimated 83,717 annual nests at index beaches from 2009 to 2013. The difference in estimates is likely because we used all available data, including index and other nesting beaches for which data were available.

As described in the next section, there does not appear to be an increasing nesting trend; rather, recent estimates are more accurate than previous ones, which were based on less data.

An overall estimate of nesting females for the DPS is not available because of reproductive parameter uncertainty: remigration intervals and clutch frequencies vary spatially and temporally, and data are insufficient for some RUs. Adequate data are available from the Northern RU and the State of Florida, which represents 89% of nesting within the DPS (Ceriani and Meylan 2017). Ceriani *et al.* (2019) evaluated all known Florida nesting data from 1989 to 2018. Using the average annual number of loggerhead nests between 2014 and 2018, Ceriani *et al.* (2019) estimated the total number of adults females nesting in Florida to be 51,319 (95% CI=16,639–99,739). Their estimate is higher than the Richards *et al.* (2011) point estimate of 38,334 (range = 30,096 – 51,211) females nesting in Florida between 2001 and 2010 because Ceriani *et al.* (2019) included a longer time range and accounted for uncertainties in remigration interval and clutch frequency. Thus, the difference does not reflect an increase in nesting, but rather variance in nesting over time (Ceriani *et al.* 2019), which is why the range and confidence intervals of the two estimates overlap. To avoid pitfalls of estimating nesting females based on estimates of emigration interval and clutch frequency, Shamblin *et al*. (2021) used genetic analyses to estimate female abundance for the Northern RU: 8,074 total nesting females from 2010 to 2015 (Shamblin *et al.* 2021).

Additional abundance data are provided by Eckert and Eckert (2019), who summarize the NW Atlantic DPS nesting activities as follows: there are approximately 379 nesting sites, with an abundance estimate for all but 20 beaches; the majority of these beaches (74 percent) host less than 25 annual crawls; only 17 beaches host more than 1,000 annual crawls, which include 16 beaches in Florida, which host 90 percent of the nesting for the entire DPS, and one in Mexico (Quintana Roo). Based on these data, we conclude that the NW Atlantic DPS hosts a large abundance of annual nests, concentrated on Florida beaches.

In water estimates of abundance include juvenile and adult life stages of both sexes but are difficult to perform on a wide scale. In the summer of 2010, NMFS' NE and SE Fisheries Science Centers estimated the abundance of juvenile and adult loggerhead sea turtles along the continental shelf between Cape Canaveral, Florida and the mouth of the Gulf of St. Lawrence, Canada, based on AMAPPS aerial line-transect sighting survey and satellite tagged loggerheads (NMFS 2011). They provided a preliminary regional abundance estimate of 588,000 individuals (approximate inter-quartile range of 382,000-817,000) based on positively identified loggerhead sightings (NMFS 2011). A separate, smaller aerial survey, conducted in the southern portion of the Mid-Atlantic Bight and Chesapeake Bay in 2011 and 2012, demonstrated uncorrected loggerhead sea turtle abundance ranging from a spring high of 27,508 to a fall low of 3,005 loggerheads (Barco *et al.* 2018). We are not aware of any current range-wide in-water estimates for the DPS.

2.3.1.2 Trends

The overall nesting trend of NW Atlantic DPS appears to be stable, neither increasing nor decreasing, for over two decades. At index nesting beaches monitored for 30 years, the average annual growth rate of the largest recovery unit (Peninsular Florida, −0.2%; $p = 0.61$) is not significantly different from zero indicating a lack of trend in the data (Bolten *et al*. 2019; Table

2; Figure 2). While the Northern RU demonstrates a positive, statistically significant growth rate (1.3%; $p = 0.04$) over 37 years, it falls short of the 2% growth over 50 years recovery criteria (Bolten *et al*. 2019; Figure 2). The average annual growth rate of the GoM RU (1.7%; $p = 0.17$), measured over 22 years, is not significantly different from zero (Bolten *et al*. 2019; Figure 2). Trend estimates are not available for the Dry Tortugas and Greater Caribbean RUs due to insufficient data.

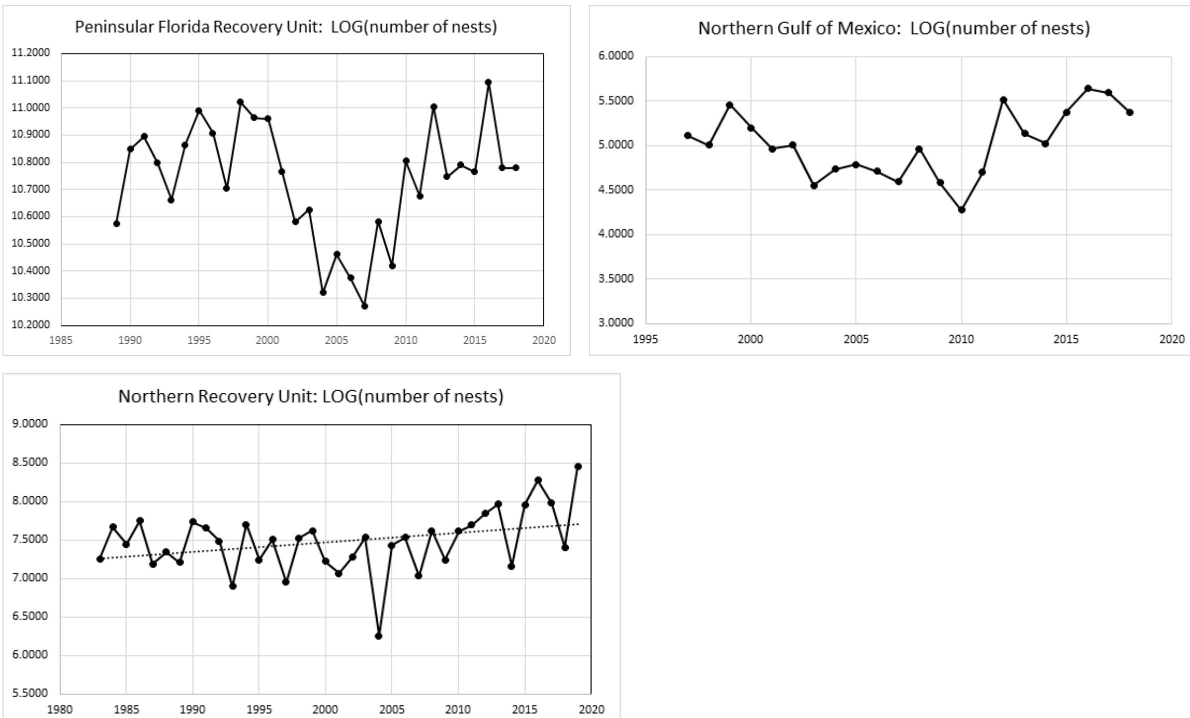

Figure 2. Average annual nesting trends at index nesting beachesThe natural log of annual nest counts at index beaches from the Peninsular Florida, Northern, and Northern Gulf of Mexico RUs; figures from Bolten *et al*. (2019).

This conclusion is similar to other published nesting trends for the DPS. As described above, Ceriani *et al.* (2019) analyzed 30 years (1989 to 2018) of loggerhead nesting data from all nesting beaches in Florida (not just index beaches, and including multiple RUs), incorporating uncertainty in clutch frequency and remigration interval into their analyses. They reported an exponential curve mean annual growth rate of 0.0093 (95% confidence interval = −0.029 to 0.056), which they interpreted as neither an increasing nor decreasing trend of nesting females in Florida (Ceriani *et al.* 2019). The IUCN Red List Assessment (Ceriani and Meylan 2017) reports a very small but positive overall trend for the DPS (a total of 2% growth between 1989 and 2013). Despite large increases in nesting trends at index beaches of the Northern RU (35%) and Greater Caribbean RU (53%) between 1989 and 2013, overall growth is hampered by nesting declines in the GoM RU (−1%) and Peninsular Florida RU (−2%), which hosts the vast majority of nesting within the DPS (Ceriani and Meylan 2017). The IUCN Red List Assessment requires three generations of abundance data to apply Criterion A, a reduction in population size (IUCN 2014); however, such historical data (covering approximately 135 years) are not available for this DPS, nor any sea turtle population. Therefore, the IUCN assessment assumed that the population abundance three generations ago was "similar to the first observed abundance" on nesting beaches. We do not agree with this assumption. Prior to European contact, there were likely millions of loggerhead turtles in the Caribbean (Bjorndal and Jackson 2003). By the 1960s and 1970s, the earliest years of "first observed abundance" estimates, nesting declines had already been reported in the United States (Florida), Mexico (Quintana Roo), Colombia, and Honduras (Ross 1995). Thus, it is likely that nesting has declined over historical time frames (i.e., hundreds of years).

The Status Review Report expressed concern over declining trends for the DPS, citing a 41% decline in the Peninsular Florida RU from 1998 to 2008 (Conant *et al.* 2009). It is encouraging that the annual number of nests is no longer declining; however, the lack of continued population growth (not significantly different from 0) is somewhat surprising, given its estimated average annual maximum population growth rate of 0.024, with a mode of 0.017 and a 95% highest density interval of 0.006–0.047 (Hatch *et al.* 2019). Others have struggled to explain the lack of strong population recovery, especially considering extensive conservation efforts and protections for loggerheads in Florida and throughout the United States (Ceriani *et al.* 2019). To further evaluate the lack of recovery, we review demographic parameters in section 2.3.1.3 and analyze the impact of threats on the DPS in section 2.3.2 to provide possible explanations for our conclusion, that the DPS demonstrates a stable (neither increasing nor decreasing) trend.

2.3.1.3 Demographic Parameters

Female maturity is associated with a mean SCL of 90.5 cm (range 75.0–101.3 cm) and a mean age of 36 to 38 years (mean age predictions for minimum age are 22.5 to 25 years; Avens *et al.* 2015) with a 95% predictive interval of 29 to 49 years (Chasco *et al.* 2020). Male maturity is associated with a mean SCL of 96.8 cm (range 75.0–101.3 cm) and a mean age of 37 to 42 years (mean age predictions for minimum age are 26 to 28 years; Avens *et al*. 2015). On average, post-maturation longevity (i.e., adult-stage duration) is 19 years, ranging from 4 to 46 years (Avens *et al.* 2015). Mayne *et al.* (2020) estimated the average maximum lifespan of loggerhead turtles to be 62.8 years (± 3.7 years). Capture-mark-recapture survival estimates (88%) for NW Atlantic loggerheads are below the global average annual survival for adult marine turtles, but this is possibly a result of sampling bias (Pfaller *et al*. 2018). Tag loss, undetected nests, emigration, death, and senescence may account for this bias and for the "missing majority," i.e., the large portion of nesting females that are not recaptured (Shamblin *et al.* 2021). This missing majority may also impact other parameters, including clutch frequency and remigration interval. In regards to clutch frequency, Tucker (2010) found that satellite telemetry studies detected much higher clutch frequencies (5.4 nests per season) than mark-recapture approaches (2.2 nests per season). In regards to remigration intervals, Shamblin *et al.* (2021) found that genetic assignment of clutches to nesting females provides greater detection rates (65%) of inter-seasonal nesting compared to mark-recapture approaches (<25%). Unfortunately, satellite tracking and genetic assignment tests are less common than flipper tag and passive integrated transponder (PIT) tag data. We report estimated values for various life stage parameters (Table 4).

Table 4. Life history parameters
Estimates or ranges based on available data. Data included are not an exhaustive list.

Parameter	Location or RU	Value	Reference
Remigration interval: mean years	Northern RU Archie Carr NWR, FL Keewaydin Island, FL St. Joseph Peninsula, FL Quintana Roo, Mexico Cuba	2.67±0.89 3-5 3.2±1.82 4.4±2.8 1.99 2.77-4.08	Shamblin *et al*. 2021 Ceriani *et al*. 2015 Phillips *et al*. 2014 Lamont *et al*. 2014 González *et al*. 2020 Azanza-Ricardo *et al*. 2020
Clutch frequency: mean nests/year	Northern RU Keewaydin Island, FL St. Joseph Peninsula, FL Quintana Roo, Mexico Cuba	4.3−4.6 3.8 3.1 2.33 1-2	Shamblin *et al*. 2017 Phillips *et al*. 2014 Lamont *et al*. 2014 Cuevas *et al*. 2020 Azanza-Ricardo *et al*. 2020
Clutch size: range eggs/nest	Northern RU Peninsular Florida RU Northern GoM RU Greater Caribbean RU	102.4-114.7 95.4-125 98.6-108 85.9-129.9	Eskew 2012; Lasala *et al*. 2013 Perrault *et al*. 2016; Ceriani *et al*. 2015 Lamont *et al*. 2014; Lamont *et al*. 2012 Azanza-Ricardo *et al*. 2020; Garcia-Cruz *et al*. 2020
Nesting success: mean %	North Carolina Archie Carr NWR, FL Juno Beach, FL St. Joseph Peninsula, FL Quintana Roo, Mexico Cuba	53.9 68±19 42.4 40.6 75.2±23 67	Halls and Randall 2018 Witherington *et al*. 2011 Hirsch *et al*. 2019 Lamont and Fujisaki 2014 González *et al*. 2020 Azanza-Ricardo *et al*. 2020
Hatching success: mean %	Indian River County, FL Boca Raton, FL Keewaydin Island, FL St. Joseph Peninsula, FL Quintana Roo, Mexico	68.6±35.5 42.8, 53.6 55.5±39.7 87.3±17.3 87.2±16.9	Lindborg *et al*. 2016 Bladow and Milton 2019 Shaw 2013 Montero *et al*. 2018 González *et al*. 2020
Emergence success: mean %	Jekyll Island, GA Archie Carr NWR, FL Peninsular Florida RU Northern GoM RU Quintana Roo, Mexico Cuba	69.9 53.3±3.7 45.6 51.6 78.8±24.4 74-82	Holbrook *et al*. 2019 Ehrhart *et al*. 2014 Brost *et al*. 2015 Brost *et al*. 2015 González *et al*. 2020 Medina Cruz *et al*. 2012
Female maturity: mean age (years) and size (cm SCL)	NW Atlantic Ocean	36−38 90.5 (75−101.3)	Avens *et al*. 2015
Male maturity: mean age (years) and size (cm SCL)	NW Atlantic Ocean	37−42 95.8 (80.6−103.8)	Avens *et al*. 2015
Annual adult survival rate: mean % (95% CI)	Bald Head Island, NC Wassaw Island, GA Keewaydin Island, FL St. Joseph Peninsula, FL Juno Beach, FL	85 (78-93) 87 (84-89) 73 (69-76) 86 (75-93) 60 (40-78)	(Monk *et al*. 2011) (Pfaller *et al*. 2013; Pfaller *et al*. 2018) (Phillips *et al*. 2014) (Lamont *et al*. 2014) (Sasso *et al*. 2011)

2.3.1.4 Spatial Distribution and Structure

The NW Atlantic DPS occurs in the North Atlantic Ocean, from the Equator to 60º N latitude and west of 40º W longitude (Conant *et al*. 2009; Figure 1). The northern extent of their range is limited by their thermal tolerance: in surface temperatures lower than 10 ºC and bottom temperatures lower than 7 ºC, loggerheads may lose their ability to swim and dive, a phenomenon known as cold stunning (Witherington and Ehrhart 1989; Morreale *et al.* 1992; Smolowitz *et al.* 2015; Patel *et al.* 2018). The overall range of the DPS includes nesting, foraging, breeding, and migratory areas. Breeding likely occurs off nesting beaches, described below. Adult foraging is mainly concentrated along the North American continental shelf (Hawkes *et al*. 2011; Winton *et al*. 2018; Evans *et al*. 2019) but also occurs throughout the Wider Caribbean (Table 5). Stewart *et al*. (2019) conducted a mixed stock analysis of loggerheads (N = 850) bycaught in NW Atlantic fisheries between 2000 and 2014; they found that fisheries' bycatch in northern fishing areas tend to be smaller turtles (<63 cm SCL), while those bycaught in southern areas tend to be larger (>63 cm SCL) and are closer to their putative nesting beaches. Juveniles from this DPS also forage in the NE Atlantic Ocean and the Mediterranean Sea (Piovano *et al*. 2011), where mixed stock analyses of stranded and bycaught juveniles indicate western Atlantic origin (i.e., at least 50 percent originated from one or more of the NW Atlantic RUs). For example, the area around the Azores is known to host many small juveniles associated with *Sargassum* spp. in the North Atlantic Gyre (Chambault *et al.* 2019).

Table 5. Nesting and foraging occurrence in the Wider Caribbean
Nations that host regular or infrequent nesting and/or foraging loggerheads of the NW Atlantic DPS (Eckert and Eckert 2019; WIDECAST.org).

Wider Caribbean Region	Nation	Nesting	Foraging
Bahamian	The Bahamas	Regular	Regular
	Turks & Caicos Islands (United Kingdom)	Infrequent	Infrequent
Greater Antilles	Cuba	Regular	Regular
	Cayman Islands (United Kingdom)	Regular	Infrequent
	Jamaica		Infrequent
	Haiti	Regular	Regular
	Puerto Rico (United States)		Infrequent
Lesser Antilles (Eastern Caribbean)	U.S. Virgin Islands		Infrequent
	British Virgin Islands (United Kingdom)		Infrequent
	Saint Kitts & Nevis		Infrequent
	Montserrat (United Kingdom)	Infrequent	Unknown
	Guadeloupe (France)		Regular
	Saint Martin (France)		Unknown
	Saint Barthelemy (France)		Unknown
	Martinique (France)		Unknown
	Grenada		Regular
Guiana	Suriname		Infrequent
Southern Caribbean	Trinidad & Tobago	Infrequent	Infrequent
	Venezuela	Regular	Regular
	Bonaire (Netherlands)	Regular	Infrequent
	Curacao (Netherlands)	Regular	
	Aruba (Netherlands)	Regular	Infrequent
Southwestern (SW) Caribbean	Colombia	Regular	Regular
	Panama	Infrequent	Regular
	Costa Rica	Regular	Regular
	Nicaragua		Regular
Western Caribbean, Gulf of Mexico, and Florida	Honduras	Infrequent	Infrequent
	Guatemala		Regular
	Belize	Regular	Regular
	Mexico	Regular	Regular
	US: Texas and Florida	Regular	Regular
Bermuda	Bermuda (United Kingdom)	Infrequent	Infrequent

The NW Atlantic DPS hosts the largest nesting assemblage of loggerhead turtles worldwide. Nesting occurs on beaches along the coasts of the SE United States, Yucatan Peninsula (Mexico), Central America, northern South America, and throughout the Caribbean (Figure 3;

Eckert and Eckert 2019). The vast majority of nesting (approximately 90%) occurs in Florida (Figure 4; Ceriani *et al*. 2019).

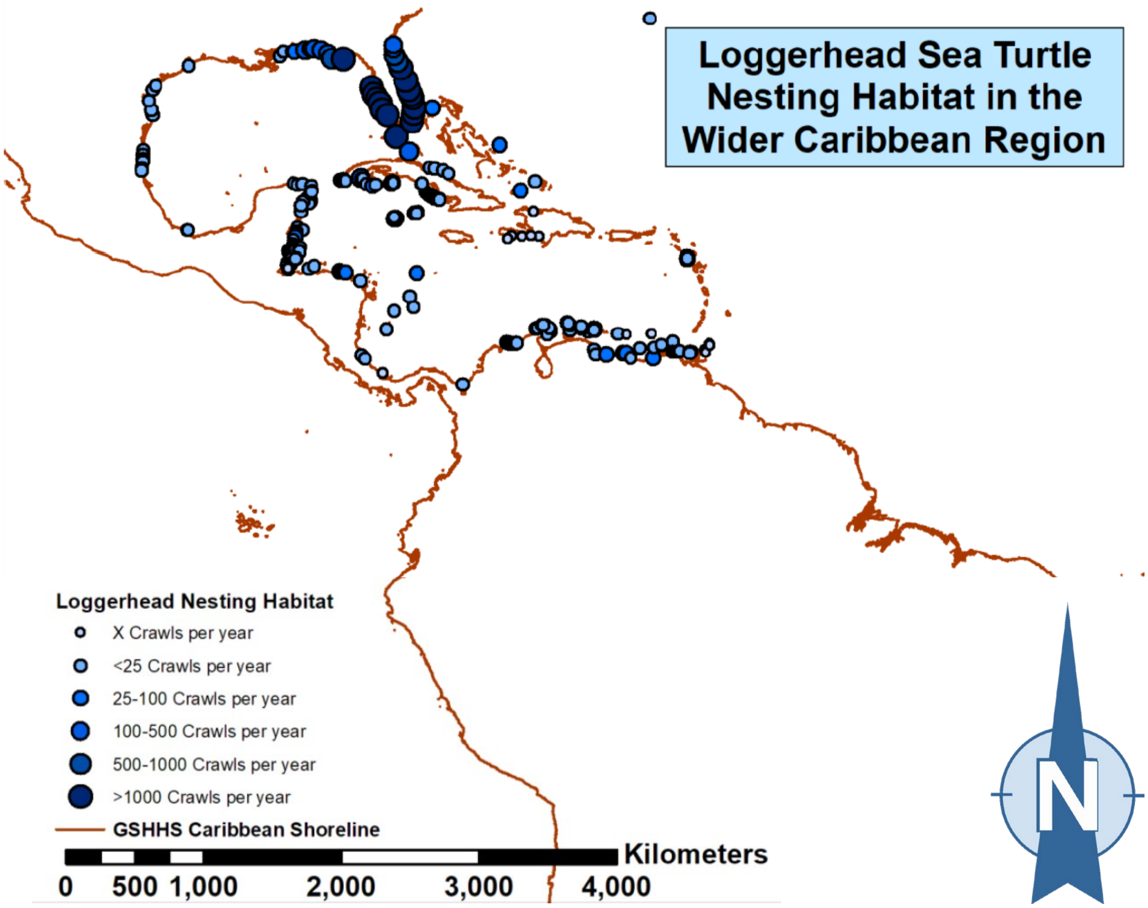

Figure 3. Known nesting sites in the Wider Caribbean
Estimated number of annual crawls at NW Atlantic DPS nesting beaches (N = 379); figure from Eckert and Eckert (2019) WIDECAST.org; SW Atlantic DPS nesting sites in Brazil have been excluded from this version to focus on the NW Atlantic DPS.

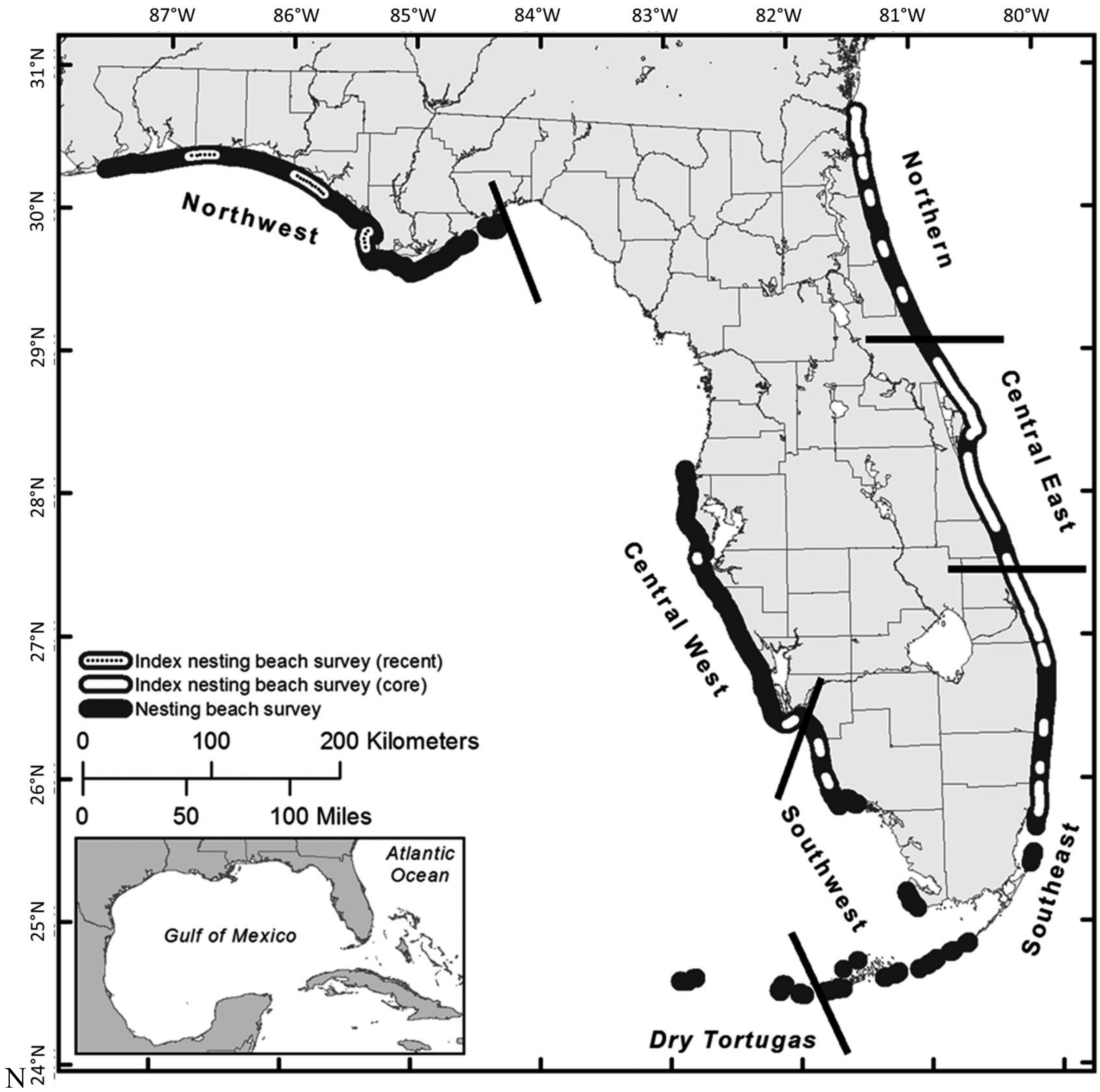

Figure 4. Nesting beach surveys within Florida
Location of annual statewide nesting surveys in Florida, including index and other monitored beaches; figure from Ceriani *et al*. (2019).

Population structure (or subdivision) occurs within the DPS. Early genetic studies of this subdivision were described in the 2008 Recovery Plan. Subsequent genetic analyses have revealed additional fine-scale subdivision, especially within the Peninsular Florida RU (Shamblin *et al.* 2014). Analyzing 800 bp of control region mtDNA, Shamblin *et al*. (2014) found statistically significant pairwise differentiation ($F_{ST} = 0.01 – 0.93$; $P < 0.05$) among the following eight clusters:

1. Northern RU
2. Dry Tortugas and Cay Sal, Bahamas
3. Quintana Roo, Mexico and SW Cuba
4. Central eastern Florida

5. SE Florida
6. SW Florida
7. Central western Florida
8. NW Florida

Shamblin *et al*. (2014) found that the Northern RU extends southward into NE Florida, and the Peninsular Florida RU likely comprises four genetically differentiated nesting clusters: central eastern Florida (Canaveral and Melbourne), SE Florida (Juno and Fort Lauderdale), SW Florida (Keewaydin), and central western FL (Casey Key). Loggerheads nesting in the Dry Tortugas were not genetically distinct from those nesting in Cay Sal (The Bahamas); however, Shamblin *et al*. (2014) suggest separate management based on geographic and likely demographic differences. Shamblin *et al*. (2014) did not evaluate samples from Texas in their study; however, the Division of Sea Turtle Science and Recovery of Padre Island National Seashore (National Park Service) reports haplotype frequencies of turtles nesting in Texas are similar to those within the Northern GoM RU (public comment submitted by D. Shaver, Division of Sea Turtle Science and Recovery of Padre Island National Seashore, National Park Service, 2020).

The Greater Caribbean RU, which includes the most diverse and possibly oldest NW Atlantic rookery in Mexico, remains distinct from all other RUs and from Cay Sal, although Nielsen (2010) detected evidence of male-mediated gene flow between loggerheads from NW Florida and Mexico. Mexico and Cuba were the only two nesting aggregations sampled from the Greater Caribbean RU, and they were not genetically differentiated. Again, Shamblin *et al*. (2014) suggest separate management based on geographic and likely demographic differences.

We conclude that the DPS is not a panmictic population with a large nesting female population (i.e., females do not randomly mate within the DPS). The DPS is a subdivided population of at least eight genetic subpopulations, each with small to moderate nesting female abundance. While having multiple subpopulations provides some resilience to the DPS (e.g., it is less likely to be extirpated due to a local catastrophic event), the DPS cannot be managed as a single unit, as the recovery plan demonstrates. The different RUs face different threats or demographic challenges that must be addressed with conservation actions focused on each RU, and on nesting clusters within the Peninsular Florida RU. The DPS has a broad distribution, with many diverse nesting and foraging areas throughout the North Atlantic Ocean. All areas appear to sufficiently provide for necessary life history requirements.

2.3.1.5 Genetic Diversity

In the Mediterranean Sea, Carreras *et al*. (2011) found that genetic diversity was similar between individuals originating from NW Atlantic and Mediterranean DPSs, which is surprising given the smaller population size, exploitation history, and population decline of the Mediterranean DPS.

Chow *et al*. (2019) conducted the first genome-wide genetic assessment of loggerheads using 45 neritic-stage loggerheads captured from the South Atlantic Bight and Florida Bay. They found relatively low genome-wide genetic diversity, which may indicate a past bottleneck, though additional studies are needed. We conclude that the DPS demonstrates relatively low genetic diversity, especially for its population size, which may reduce its resilience to environmental changes.

2.3.2 Threats Analysis

Section 4(a)(1) of the ESA requires the Services to determine whether a species is endangered or threatened because of any of the following factors (or threats) alone or in combination:

1) The present or threatened destruction, modification, or curtailment of its habitat or range;
2) Overutilization for commercial, recreational, scientific, or educational purposes;
3) Disease or predation;
4) Inadequacy of existing regulatory mechanisms to address identified threats; or
5) Other natural or human factors.

2.3.2.1 Present/Threatened Destruction/Modification/Curtailment of Habitat/Range

As described below, destruction and modification of terrestrial and marine habitats threaten the NW Atlantic DPS. On beaches, threats that interfere with successful nesting, egg incubation, hatchling emergence, and transit to the sea include erosion, erosion control, coastal development, artificial lighting, beach use, and beach debris. In the marine environment threats that interfere with foraging and movement include marine debris, oil spills and other pollutants, harmful algal blooms, and noise pollution.

Terrestrial Habitat Modification

Erosion of beaches is a result of anthropogenic and natural processes. Erosion is often worsened when man-made coastal and inwater structures interfere with natural coastal processes (Von Holle *et al.* 2019). Ultimately, erosion and efforts to mitigate erosion, lead to the loss of suitable nesting habitat. For example, between 1982 and 2006, beach erosion narrowed most sections of the Archie Carr National Wildlife Refuge (Melbourne, Florida), which hosts the largest nesting assemblage in the DPS, with an average loss of 3.22 m (0.16 m/yr) of beach (Reece *et al*. 2013). Beach erosion is also one of the most significant threats to loggerheads at the egg life stage (Bolten *et al.* 2011). High tides and tidal washouts can flood and erode nesting beaches, washing eggs into the sea or lethally inundating developing embryos (Brost *et al.* 2015; Butler *et al.* 2020).

Beach renourishment is often used to manage beach erosion by adding or redistributing sand; however, renourishment often results in diminished nesting success (Long *et al.* 2011; Hays 2012). Designed to stabilize shorelines and prevent erosion, beach armoring structures (e.g., revetments and seawalls) decrease nesting activity by preventing females from accessing suitable nesting sites (Rizkalla and Savage 2011). Lamont and Houser (2014) found that alterations to the nearshore environment (e.g., jetties, dredging, or installation of pilings) also alter sea turtle nest distribution. Thus, beach renourishment may result in reduced nesting or force turtles to nest at suboptimal locations.

Coastal development alters nesting habitat, making it less suitable for nesting females, egg incubation, and hatchling emergence. Sella and Fuentes (2019) found that 100% of very high (i.e., top three) and high (i.e., top 25%) loggerhead density nesting beaches were exposed to cumulative coastal modification and construction. Fuentes *et al*. (2016) found that 100% of very high and high density nesting areas in the Peninsular Florida RU are exposed to coastal development and light pollution and therefore need intervention (Fuentes *et al.* 2016). Artificial lighting deters females from nesting (Witherington *et al.* 2014), resulting in reduced nest densities (Bonner 2015; Weishampel *et al.* 2016; Hu *et al.* 2018; Linz 2018; Price *et al.* 2018; Windle 2018). Artificial lighting can disrupt or delay hatchlings' sea-orienting ability, which increases nest-to-sea mortality as a result of dehydration, exhaustion, or predation (Witherington *et al.* 2014; Erb and Wyneken 2019; Vindiola 2019; Stanley *et al.* 2020).

Nesting habitat can also be degraded by the presence of humans, human activities (such as beach driving), and recreational equipment (e.g., boats, cabanas, and furniture). Equipment left on beaches and other beach debris can also deter, impede, and/or entrap nesting females and hatchlings, reducing nesting, and interfering with hatchling emergence and transit to the sea (Martin *et al.* 2019). Microplastic beach debris alters the temperature and permeability of sand (Andrady 2011), disturbing the incubating environment for marine turtles (Beckwith and Fuentes 2018). In Mexico, extreme *Sargassum* events can create obstacles for hatchlings. Decomposing *Sargassum* that is washed up into nesting habitat and on beaches results in lethal temperatures for underlying nests, and the removal of *Sargassum* can cause beach compaction, deter nesting activity, and interfere with hatchling emergence (Chávez *et al.* 2020). In Cuba, large amounts of *Sargassum* on beaches led to an increase in the frequency of failed nesting attempts and caused a displacement in nesting activities (Azanza-Ricardo and Perez Martin 2016).

Marine Habitat Modification

Marine pollution, which includes marine debris and chemical pollutants, is one of the main anthropogenic threats to sea turtles and a critical environmental concern (Schuyler *et al*. 2016). Loggerhead turtles of all life stages are prone to ingesting marine debris (Gall and Thompson 2015). Foraging loggerheads respond similarly to the odors of prey items and biofouled plastic, the scent of which stimulates foraging behavior and contributes to detrimental (and sometimes fatal) interactions with marine debris (Pfaller *et al.* 2020a). Marine debris ingestion results in death when the debris blocks or injures turtles' digestive tracts (Wilcox *et al.* 2018). It can also cause sub-lethal effects (e.g., dietary dilution, malnutrition, and assimilation of contaminants) that reduce energy intake, lower overall condition, and diminish reproductive output (Nelms *et al.* 2016a; White *et al.* 2018; Eastman *et al.* 2020; Marn *et al.* 2020). Marine debris also causes entanglement, which may lead to injury or death from drowning, starvation, or predation due to increased drag (Nelms *et al.* 2016a). In a survey of sea turtle experts, entanglement in marine debris was ranked the third greatest threat to sea turtles, less than fisheries bycatch and plastic ingestion but greater than direct exploitation, climate change, and oil pollution (Duncan *et al.* 2017). Chemical pollutants are another potential concern, having been identified in blood, tissue, and eggs of marine turtles, with concentrations varying greatly depending on geographic location, seasonal changes, trophic level, lipid content (of different tissues), body condition, sex, and age class (Keller 2013); however, very little is known about toxicological effects of chemical contaminants in marine turtles, and how this threat might impact turtle development, health, growth, survivorship, reproduction, and, ultimately, population stability (Hamann *et al.* 2010; Finlayson *et al.* 2016).

Oil spills can affect sea turtles at all life stages (NOAA 2016), as demonstrated by the Deepwater Horizon oil spill in the Gulf of Mexico, which contaminated vital foraging, migratory, and breeding habitats at the surface, in the water column, and on the ocean bottom (McDonald *et al.* 2017; Mitchelmore *et al.* 2017; Wallace *et al.* 2017). The Natural Resources Damage Assessment conducted following the spill estimated that approximately 2,100 to 10,000 small juveniles and 2,200 to 3,600 large juvenile and adult turtles were killed by the spill; an additional 34,000 loggerhead hatchlings were estimated to have been killed by oil spill response activities (Deepwater Horizon Natural Resource Damage Assessment Trustees 2016). Tracking data demonstrated that Northern GoM RU nesting females were likely exposed to the oil spill at internesting areas, foraging sites, and migratory corridors (Hart *et al.* 2013; Hart *et al.* 2014; Hart *et al.* 2018). Scute layer biomarkers confirmed that females continued to forage in oiled areas after the spill (Vander Zanden *et al.* 2016). Lauritsen *et al.* (2017) estimated a 44% reduction in loggerhead nest densities on NW Florida beaches in 2010, a significant portion of annual nesting in the Northern GoM RU, as a result of direct (e.g., mortality) and indirect (e.g., deterrence of nesting) effects of the oil spill. Hatchlings from *in-situ* nests were affected by oil contamination and/or the dispersant. Bembenek-Baily *et al*. (2019) found that skeletal muscle lactate, creatines, and taurine concentrations were significantly lower in hatchlings exposed to crude oil than in control hatchlings, suggesting that oil-exposed animals suffered energy depletion; those exposed to both oil and Corexit (an oil dispersant used in spill clean-up) likely experienced oxidative stress. Harms *et al.* (2019) found that hatchlings exposed to the dispersant failed to gain weight as expected with normal hatchling hydration in seawater. Nesting beaches of the northern GoM RU were also threatened with oil. Nearly 275 loggerhead turtle nests (approximately 28,000 sea turtle eggs) were relocated from oiled beaches to an incubation facility at the Kennedy Space Center in Cape Canaveral, Florida, where the eggs were held until the hatchlings emerged and were released on unoiled Atlantic coast beaches (MacPherson *et al.* 2012).

Harmful algal blooms (HABs), also called “red tides,” are a significant, nearly-annual threat to the DPS, especially to turtles inhabiting the waters off SW Florida (Hart *et al.* 2018). Turtles exposed to elevated concentrations ($\geq 10^5$ cells l^{-1}) of the algae *Karenia brevis* inhale and/or ingest neurotoxins that can cause brevetoxicosis, which may result in death or lethargy, lack of coordination, and unusual muscular activity. Depending on the location, HABs can lead to an increase in loggerhead strandings, which are usually adults and large juveniles (CCL mean ± SD = 88.3 ± 12.9 cm), consistent with the life stages typically found in neritic habitats of Florida (Foley *et al.* 2019b). Based on Florida loggerhead stranding data between 1986 and 2013, HABs likely caused 260 to 520 loggerhead deaths annually and accounted for 7.1% of strandings during that time period (Foley *et al.* 2019b). In addition, nesting females can transfer brevetoxins to eggs which can reduce hatching and emergence success (Perrault *et al.* 2016).

In the marine environment, noise pollution is created by shipping, drilling, seismic surveys, pile driving, and Navy sonar and airguns. Sea turtles can detect frequencies between 50 Hz and 1600 Hz (Dow Piniak *et al.* 2012; Lavender *et al.* 2012; Martin *et al.* 2012; Lavender *et al.* 2014), which includes the peak amplitude, low frequency sound emitted by seismic airguns (10-500 Hz; DeRuiter and Larbi Doukara 2012) as well as other pervasive low-frequency and high-intensity anthropogenic noise in the ocean including engine noise, pile driving, offshore drilling, and low-frequency sonar (Nelms *et al.* 2016b). An ESA section 7 consultation on Phase III Navy Atlantic Fleet Training and Testing anticipated incidental take (mostly acoustic harassment) of 47,047 loggerheads over 7 years (see FPR-2018-9259 in the Environmental Consultation Organizer). Noise pollution affects turtles’ ability to avoid predators, navigate, and identify nesting beaches (Martin *et al*. 2012; Dow Piniak *et al*. 2012). Hearing damage may reduce a turtle’s ability to avoid natural and anthropogenic threats. Turtles may react by avoiding noisy areas, which would otherwise be used for breeding, foraging or thermoregulation (basking); or they may compromise their energy budgets by changing their foraging duration, swim speed, dive depth/duration, and time spent at the surface to rest/breathe (DeRuiter and Larbi Doukara 2012). Noise pollution impacts on individual fitness and population demographics remain unknown.

Summary

Based on the best available data, we find that the DPS faces present and threatened destruction and modification of its habitat. Habitat threats directly reduce abundance by removing nesting females from the population; they also reduce productivity by interfering with nesting and hatching and emergence success. Loss and modification of beach habitat are also likely to reduce the diversity and distribution of nesting beaches, thus impairing representation, resilience, and redundancy of the DPS.

Beach erosion, erosion control methods, and coastal development reduce availability of suitable nesting habitat. Artificial lighting, beach use, beach debris, shoreline structures, and coastal development deter nesting females and prevent females and hatchlings from reaching the sea. Important foraging habitats are modified by pollution, contaminants, and oil spills. Individuals are killed and injured by derelict fishing gear and other marine debris, either through ingestion or entanglement. The threat clearly affects many individuals; however, the magnitude of the threat to the population is unknown because most affected turtles are never observed.

Furthermore, most habitat-related recovery plan objectives, listed here, have not been met (Bolten *et al*. 2019):

- Beach armoring, shoreline stabilization structures, and all other barriers to nesting are categorized and inventoried for areas under U.S. jurisdiction: not met. Florida has a representative index of all barriers (Witherington *et al*. 2011) and a draft inventory of beach armoring that covers armoring and shoreline stabilization structures (Florida Beaches Habitat Conservation Plan (HCP), in progress); however, other States do not.
- A peer-reviewed strategy is developed and implemented to ensure that the percentage of nesting beach free of barriers to nesting is stable or increasing relative to baseline levels: not met. The percentage of nesting beach free of barriers is decreasing.
- Beach sand placement projects conducted in areas under U.S. jurisdiction are in compliance with state and USFWS criteria and are conducted in a manner that accommodates loggerhead needs and does not degrade or eliminate nesting habitat: not met but in progress. While most sand placement projects on nesting beaches are conducted in compliance with state and USFWS criteria, projects should not be conducted during the nesting season.
- At least 1,581 km of loggerhead nesting beaches and adjacent uplands under U.S. jurisdiction are maintained within conservation lands in public (Federal, state, or local) or private (NGO and private conservation lands) ownership that are managed in a manner compatible with sea turtle nesting: met. Additional lands have been acquired in Alabama and Georgia.
- A peer-reviewed model is developed that describes effects of sea level rise on loggerhead nesting beaches, and steps have been taken to mitigate such effects: not met. Multiple models predict sea level rise and can be used to infer effects on nesting beaches; however, no steps have been taken to mitigate such effects.
- Nesting beaches outside U.S. jurisdiction are managed for compatibility with loggerhead nesting: not met. Data not available.
- A peer-reviewed, comprehensive strategy is developed and implemented to identify, prioritize, and protect marine habitats (e.g., feeding, migratory, internesting) important to

loggerheads: not met. A comprehensive strategy has not been developed; however, critical habitat has been designated for the DPS, and European initiatives protect seamounts, which are important developmental areas for oceanic stage loggerhead turtles.

- A peer-reviewed strategy is developed and fully implemented to quantify, monitor, and minimize the effects of marine debris ingestion and entanglement in U.S. territorial waters, the U.S. Exclusive Economic Zone (EEZ), the band of water that extends outward 200 miles from shore, foreign EEZs, and the high seas: not met. No progress has been made on this criterion.

These recovery plan objectives, once met, are likely to reduce the threat of habitat destruction and modification. Given the current magnitude of habitat destruction and modification and its impact on abundance and productivity, we conclude that it is a major threat to the DPS that is further exacerbated by climate change (see 2.3.2.6 Climate Change).

2.3.2.2 Overutilization: Commercial/Recreational/Scientific/Educational Purposes

Overutilization includes the killing of turtles and eggs for commercial, recreational, scientific, and education purposes. Within the United States, the harvest of turtles and eggs is illegal; however, poaching of eggs appears to be a negligible threat. We were unable to estimate poaching as a percentage of total egg production; however, data indicate that it accounts for a small percentage of egg loss. In Florida, poaching accounted for less than 1% of egg and hatchling loss (all sea turtle species) between 2002 and 2012 (Brost *et al.* 2015). In Georgia, 38 nests with 3,428 eggs (1.8% of total egg loss) were poached between 2009 and 2018 (Butler *et al.* 2020).

The harvest of turtles and eggs is illegal in most nations throughout the range of the Greater Caribbean RU (Eckert and Eckert 2019); however, legal harvest occurs in some nations, and illegal harvest occurs in others. Humber *et al*. (2014) estimated the legal harvest of loggerhead turtles as of January 1, 2013: Atlantic Colombia (N = 646); Haiti (N = 328); Grenada (N = 24); and St. Vincent (N = 8). Since then, Colombia and St. Vincent have enacted national policies to protect sea turtles (Eckert and Eckert 2019). A report from 19 of the 48 parties of the North Atlantic and Wider Caribbean Region indicated that illegal harvest of turtles occurs in Cuba and Mexico, despite full protection of the species in these nations, and in Belize, Colombia, and Venezuela, despite protection with the exception of subsistence/indigenous use (Eckert and Eckert 2019; Nalovic *et al.* 2020). Caderno Peña and Moncada Gavilán (2019) found that poaching occurs at 33% of protected beaches in NW Cuba, where fewer than eight loggerheads were poached annually between 2010 and 2016 (Caderno Peña and Moncada Gavilán 2019). In Venezuela, large juvenile (58.5-83.2 cm CCL) loggerhead carcasses provide evidence of poaching (Rojas-Cañizales *et al.* 2020). A report from 19 of the 48 parties of the North Atlantic and Wider Caribbean Region indicates that illegal egg harvest occurs in Belize, Colombia, Curaçao, Mexico, and Venezuela (Nalovic *et al.* 2020). Egg poaching likely occurs in other nations but is difficult to quantify.

Research activities involve handling turtles for purposes of tagging, measuring, and biological sampling. Within the United States, researchers are required to obtain an ESA permit, which includes specific protocols to avoid, minimize, and mitigate unintended adverse effects that may result from their activities. In 2020, NMFS had 40 active permits that allow direct take of Atlantic loggerheads for research purposes (see NMFS' [Authorizations and Permits for Protected]

Species (APPS) for Atlantic loggerheads). Under an ESA section 7 programmatic review of the issuance of permits for research activities on sea turtles, the maximum allowed number of mortalities is 5 NW Atlantic loggerheads over a 10-year period (NMFS 2019). In 2020, permitted direct take included 158 loggerheads, with no mortalities (NMFS unpublished data 2021). We conclude that the impact of research on turtles is low.

Eggs are also taken for scientific studies, which requires an ESA permit within the United States. For the Northern RU, one egg from each nest was sacrificed for a series of genetic studies, for a total of 41,576 eggs between 2010 and 2015 (Shamblin *et al.* 2014; Shamblin *et al.* 2017; Shamblin *et al.* 2021). In Georgia, research egg loss included eggs taken for ongoing genetic research as well as eggs damaged through nest probing and excavation, representing 10% of all egg loss in Georgia between 2009 and 2018 (Butler *et al.* 2020).

Legal and illegal harvest of adults and large juveniles reduces abundance and productivity. Egg harvest and poaching reduces productivity. Reductions in abundance and productivity reduce DPS resilience. Furthermore, none of the overutilization recovery plan objectives, listed here, have been met (Bolten *et al*. 2019):

- Legal harvest (both commercial and subsistence) in the Caribbean, Atlantic, and Mediterranean is identified and quantified: not met. The Recovery Plan identifies, but does not quantify, legal harvest.
- A strategy is developed and implemented to eliminate legal harvest through international agreements: not met. The Inter-American Convention (IAC) for the Protection and Conservation of Sea Turtles prohibits direct harvest with potential exception for traditional economic subsistence; however, some nations within the NW Atlantic DPS range are not party to the IAC.
- A scientifically based nest management plan outlining strategies for protecting nests (under U.S. jurisdiction) from natural and manmade impacts is developed and implemented: not met. Most States have developed management plans for nest protection; however, additional efforts are needed to ensure consistency across States and minimize impacts to nests.

Turtles are protected from harvest in most nations throughout the range of the DPS, however, even with these protections some level of poaching of both adults and eggs still occur. Loss of large juveniles and adults impacts abundance and productivity, but the magnitude of harvest is low. The impact of research on turtles is also low. Therefore, we conclude overutilization poses a low-level threat to the DPS.

2.3.2.3 Predation and Disease

Predation and disease affect all life stages of the DPS, which has evolved to exist with these natural phenomena. Anthropogenic impacts, such as introduced species, coastal development, and climate change, exacerbate the natural threats of predation and disease.

On nesting beaches, native and introduced species prey on loggerhead eggs and hatchlings. Most common native predators include: red and gray foxes (*Vulpes vulpes* and *Urocyon cinereoargenteus*), coyotes (*Canis latrans*), raccoons (*Procyon lotor*), armadillos (*Dasypus novemcinctus*), ghost crabs (*Ocypode quadrata*), and yellow-crowned night herons (*Nyctanassa* *violacea*). Occasional or less common native predators include: crows (*Corvus spp*.), American minks (*Mustela vison*), Virginia opossums (*Didelphis virginiana*), river otters (*Lontra canadensis*), eastern kingsnakes (*Lampropeltis getula*), and ants (Formicidae). Introduced predators include feral hogs (*Sus scrofa*), dogs (*Canis familiaris*), and red imported fire ants (*Solenopsis invicta*). Generally, predation affects less than 10 percent of nests (Brost *et al.* 2015; Butler *et al.* 2020). There has been a significant decline in nest predation in recent decades, likely as a result of nest protection (e.g., screening and caging) and targeted predator control (Kurz *et al.* 2012; Welicky *et al.* 2012; Lamarre-DeJesus and Griffin 2013; Brost *et al.* 2015; Urbanek and Sutton 2019; Butler *et al.* 2020). Predation of hatchlings varies by location, but ghost crabs and mammals (mainly raccoons) are often the predominant predators (Peterson *et al.* 2013; Erb and Wyneken 2019). In Peninsular Florida, mammalian and ghost crab depredation reduced hatching success from 75.4% to 26.1% and emergence success from 73.0% to 25.4% (Brost *et al.* 2015). At Florida nesting beaches, two studies of nest-to-sea mortality found that approximately 7% of hatchlings fail to reach the sea (Erb and Wyneken 2019; Vindiola 2019), with predation accounting for 39% of hatchling mortality (Vindiola 2019). Predation is likely exacerbated by artificial lighting that may attract predators and disorient hatchlings, delaying their sea-orienting ability (Erb and Wyneken 2019). Overall, we find that predation on eggs and hatchling reduces productivity at low levels.

In the marine habitat, sharks prey on post-hatchling, juvenile, and adult loggerhead turtles. The presence of tiger sharks does not appear to alter surfacing behaviors of adult loggerheads (Hammerschlag *et al.* 2015); however, turtles may use predator avoidance behavior in the presence of sharks at depth (Smolowitz *et al.* 2015).

It is difficult to quantify impacts of disease on the DPS (Bolten *et al.* 2011). The diversity and prevalence of internal parasites (e.g., helminths, protozoa, arthropods, and annelids) is greater in loggerheads stranded in Florida, relative to those in the Mediterranean (Greiner 2013). Spirorchiid trematodes (blood flukes) caused or contributed to deaths of some loggerheads stranded in Florida but was incidental to the cause of death in others (Stacy *et al.* 2010). George (1997) describes at least two bacterial diseases in wild loggerhead populations (bacterial encephalitis and ulcerative stomatitis/obstructive rhinitis/pneumonia), but the prevalence is unknown. Viral diseases have not been documented in free-ranging loggerheads, with possible exception of sea turtle fibropapillomatosis, which may have a viral etiology (George 1997). Between 2001 and 2014, fibropapillomatosis affected less than 1% of loggerhead turtles in rehabilitation facilities from North Carolina to Florida, with turtles displaying only mild or moderate tumors (Page-Karjian *et al.* 2015).

Recovery plan objectives pertaining to predation and disease have been met (Bolten *et al*. 2019):

- Ecologically sound predator control programs are implemented to ensure that the annual rate of mammalian predation on nests (under U.S. jurisdiction) is 10% or below within each recovery unit based on standardized surveys: met. Throughout the U.S. loggerhead nesting range, programs control predation at or below 10% within each recovery unit.
- A peer-reviewed strategy is developed to recognize, respond to, and investigate mass/unusual mortality or disease events: met. NMFS' veterinary pathologist leads investigations of unusual, mass mortality, or disease events, working in conjunction with the States.

Disease has the potential to reduce abundance by removing individuals at all life stages. Predation of eggs and hatchlings reduces productivity. Such reductions likely have a small impact on the DPS, which has evolved in the presence of predators, parasites, and diseases. We conclude that disease and predation are low-level threats to the DPS.

2.3.2.4 Inadequacy of Existing Regulatory Mechanisms

The NW Atlantic DPS has a large range, extending throughout the North Atlantic Ocean. As such, it is protected by numerous international, national, regional, and local regulations. Notable progress has been made since the DPS listing, but the regulatory landscape remains fragmented: 37 (82%) nations and territories within the range of the DPS prohibit sea turtle exploitation year-around, although four of these (Colombia, Honduras, Nicaragua, Suriname) provide for legal exceptions related to "traditional" or "subsistence" exploitation (Eckert and Eckert 2019). Where exploitation is allowed, minimum size limits (by weight or shell length) are still the norm, targeting large juveniles and adults, which reduces both abundance and productivity (Eckert and Eckert 2019). None of the regulatory recovery plan objectives, listed here, have been met (Bolten *et al*. 2019):

- Light management plans, which meet minimum standards identified in the Florida Model Lighting Ordinance (Florida Administrative Code Rule 62B-55), are developed, fully implemented, and effectively enforced on nesting beaches under U.S. jurisdiction: not met. All nesting beaches in Georgia and most nesting beaches in Florida are subject to lighting ordinances; however, few nesting beaches in South Carolina, North Carolina, and Alabama have local lighting ordinances. In Virginia, lighting protections are under Virginia Administrative Code: 9VAC15-60-60: Mitigation plan which includes guidance to avoid construction and associated lighting impacts during the nesting and hatching season (Commonwealth of Virginia, 2022). Implementation and enforcement are highly variable, due to different government entities involved, availability of funding and staff, and varying levels of prioritizing light management on nesting beaches.
- Annual percentage of total nests with hatchlings disoriented or misoriented by artificial lighting does not exceed 10% based on standardized surveys: not met. To assess this criterion, consistent reporting is needed to quantify the annual percentage of nesting females and hatchlings that are disoriented or misoriented by artificial lighting.
- Specific and comprehensive Federal legislation is developed, promulgated, implemented, and enforced to ensure long-term (including post-delisting) protection of loggerheads and their terrestrial and marine habitats, including protection from fishery interactions: not met. Bolten *et al*. (2019) report that progress has not been made on this critical issue.
- State and local legislation is developed and/or maintained, promulgated, implemented, and enforced to ensure long-term (including post-delisting) protection of loggerheads and their terrestrial and marine habitats, including protection from fishery interactions: not met. State and local laws vary in their scope, strength, and level of enforcement, with little protection from State-managed fisheries interactions.
- Foreign nations with significant loggerhead foraging or migratory habitat have implemented national legislation and have acceded to international and multilateral agreements to ensure long-term protection of loggerheads and their habitats: not met. Nations that have important foraging or migratory habitat include Canada, Mexico, Cuba, The Bahamas, Turks and Caicos Islands, Nicaragua, Panama, Colombia, Spain, Portugal,

Morocco, and Cape Verde Islands. Canada and The Bahamas are not parties to the IAC. Protection of the Sargasso Sea is critical to long-term loggerhead habitat protection.

- Nations that conduct activities affecting loggerheads in foraging or migratory habitats in the North Atlantic Basin and the western Mediterranean have implemented national legislation and have acceded to international and multilateral agreements to ensure long-term protection of loggerheads and their habitats throughout the high seas and in foreign EEZs: not met. Progress is limited. European nations protect seamounts, important developmental and foraging areas for oceanic loggerheads. However, marine pollution and other habitat threats remain problematic.

Most regulatory mechanisms within the range of the DPS have remained the same since its listing and are detailed in the listing rule (76 FR 58868; September 22, 2011). Below, we describe updates that are relevant to the DPS.

U. S. Magnuson-Stevens Fishery Conservation and Management Act (MSA)
NMFS implements the MSA, which is the primary law governing marine fisheries management in United States (16 U.S.C. 1801). Passed in 1976 and amended in 2007, the MSA fosters long-term biological and economic sustainability of marine fisheries. Section 301 (16 U.S.C. 1851) requires fishery management plans to include conservation and management measures to the extent practicable to minimize bycatch; to the extent bycatch cannot be avoided, fisheries are required to minimize mortality of bycatch. Section 316 (16 U.S.C. 1865) requires a bycatch reduction program to develop technological devices and design other engineering changes to minimize bycatch. MSA revisions to Section 610 (16 U.S.C. 1826(k)) of the High Seas Driftnet Moratorium Protection Act requires identification of nations that: 1) are engaged in fishing activities that result in bycatch of protected living marine resources; 2) fail to implement effective measures to reduce bycatch; and 3) have not adopted a regulatory bycatch reduction program comparable to that of the United States.
https://www.fisheries.noaa.gov/topic/laws-policies

U.S. Public Law 101-162, Section 609
Section 609 provides that wild-caught shrimp or products from wild-caught shrimp harvested with commercial fishing technology that may adversely affect protected sea turtle species may not be imported into the United States unless the U.S. Department of State, acting on authority delegated by the President, certifies to Congress that the exporting nation harvests shrimp under conditions that minimize the impact on endangered sea turtle populations. As of 2021, the U.S. Department of State's Bureau of Oceans and International Environmental and Scientific Affairs has determined that the following nations, within the range of the DPS, have adopted sea turtle conservation programs comparable to the U.S. program and recommends section 609 certification: Colombia, Guatemala, Guyana, Honduras, Nicaragua, Panama, and Suriname. The 2019 Improving International Fisheries Management Report to Congress reported illegal, unreported, and unregulated (IUU) fishing activities (occurring from 2016 to 2018) in Mexico, and the U.S. Department of State recommended suspending the certification of Mexico because its sea turtle protection program is no longer comparable to that of the United States.
https://www.fisheries.noaa.gov/national/endangered-species-conservation/shrimp-import-legislation-sea-turtle-conservation

Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)
This Convention is the only international treaty dedicated exclusively to sea turtles, setting standards for their conservation and habitats with a large emphasis on bycatch reduction. It is the only binding multi-national agreement for sea turtles and is open to all countries in North, Central, and South America, and the Caribbean. It currently has 16 Contracting Parties, with the United States becoming a signatory in 1999. Of note, Canada and The Bahamas are not parties to the IAC. In 2015, IAC passed the loggerhead resolution (CIT-COP7-2015_R3), which calls on Mexico and the United States to work together with other countries of the North Atlantic to share information and identify conservation actions. Additional information is available at *http://www.iacseaturtle.org*.

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)
This Convention was designed to regulate international trade in a wide range of wild animals and plants. CITES came into force in 1975 and currently includes 183 Parties. Although CITES has been effective at minimizing international trade of sea turtle products, it does not limit harvest within countries, nor does it regulate intra-country commerce of sea turtle products (Hykle 2002). A U.S. proposal – co-sponsored by Brazil, Colombia, Costa Rica, and Peru – to create a CITES resolution aimed at addressing the illegal trade in marine turtles was adopted at the November 2022 meeting. The resolution calls on Parties to scale up efforts to address illegal harvest and markets associated with illegal trade of marine turtles; take action to decrease consumer demand, improve monitoring, detection and enforcement activities; and address marine turtle bycatch in fisheries by effectively addressing IUU fishing that is a threat to marine turtles. Additional information is available at *http://www.cites.org*.

IUCN World Conservation Congress. Motion 097
The Congress has adopted a motion entitled: "Reducing marine turtle bycatch: the important role of regulatory mechanisms in the global roll-out of Turtle Excluder Devices." The motion requests voluntary measures for industry and calls on the European Union to work with exporting countries to support the use of TEDs. More information is available at *https://www.iucncongress2020.org/motion/097*

Ramsar Convention on Wetlands
The Convention on Wetlands, signed in Ramsar, Iran, in 1971, is an intergovernmental treaty that provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources and enhances the conservation of sea turtle habitat. Currently, there are 170 parties to the Convention, with 2,200 wetland sites. In 2018, a resolution was passed to enhance conservation of coastal marine turtle habitats and the designation of key areas as Ramsar Sites (/https://www.ramsar.org/sites/default/files/documents/library/xiii.24_sea_turtles_e.pdf).

Based on the regulatory mechanisms described here and in other sections of this review, we conclude that fisheries bycatch remains a major threat to the DPS, in-part due to lack of effective regulations or inadequate implementation, monitoring, and enforcement of existing regulations. Threats to nesting and marine habitats also continue to increase, in part due to inadequate regulation of coastal development near nesting beaches and marine pollutants. Overall, we conclude that the DPS is negatively impacted by the inadequacy of existing regulatory mechanisms.

2.3.2.5 Fisheries Bycatch

Fisheries bycatch impacts juvenile and adult loggerheads in pelagic and coastal waters throughout the range of the DPS (Bolten *et al.* 2011; Finkbeiner *et al.* 2011). Bycatch of NW Atlantic loggerheads occurs in numerous types of commercial and artisanal fishing gear including: pelagic and demersal longlines; drift and set nets (e.g., gillnets, trammel nets); bottom and mid-water trawling; fishing dredges; pound nets and weirs; haul and purse seines; pots and traps; and hook and line gear. While some fisheries have significantly reduced their bycatch of NW Atlantic loggerheads, bycatch continues to be the greatest threat to the DPS, reducing overall abundance (i.e., loss of individuals) and productivity (i.e., loss of reproductive potential). In the following sections, we summarize bycatch impacts on the DPS.

Fisheries within the U.S. Exclusive Economic Zone

The DPS range encompasses many fishing areas within the U.S. EEZ. Because of high density of loggerhead nesting on U.S. beaches and numerous foraging areas in coastal and continental shelf waters of the United States, this overlap often results in bycatch. As with all fisheries, bycatch data are limited. Therefore, estimates of bycatch and mortality rates are often dependent on observer data and modeling studies.

In one such study, NMFS and academic researchers used a spatial matrix population projection model to examine the potential future impact of U.S. federal fisheries' bycatch mortality on the DPS (Warden *et al.* 2015). The model predicted that if mortality continued at levels authorized by incidental take statements in NMFS' ESA section 7 biological opinions from 2001 to 2013, the DPS would decline by approximately 80% in 40 years. However, the authors cautioned that their deterministic model did not consider stochasticity in demographic parameters (which were based on 2009 values) and may not represent current and future conditions (Warden *et al.* 2015). Furthermore, mortality has not continued at levels used in the model (a maximum of 8,456 deaths annually; Warden *et al*. 2015). As of 2020, anticipated annual mortality in these fisheries was reduced to a maximum of 3,829 loggerheads (NMFS unpublished data 2021). Given this reduction in mortality and stabilization of the nesting trend, it is unlikely that the DPS will decline by 80% in 40 years. Additional modeling is needed to understand future impacts of current authorized mortality because Warden *et al*. (2015) found that removals have little impact in the near future but large impacts in later years and cautioned that such risk may not be readily apparent.

Among U.S. fisheries, longline and trawl fisheries represent the most significant threats to the DPS (Finkbeiner *et al.* 2011; Lewison *et al.* 2014; Savoca *et al.* 2020). Compiling available bycatch data of U.S. fisheries from 1990 to 2007, Finkbeiner *et al.* (2011) estimated the mean annual bycatch to be 26,500 Atlantic loggerheads, including 1,400 deaths. This is likely an underestimate because unobserved fisheries are not included in these estimates, observer coverage is low, interactions are difficult to observe if gear modifications are in place, and methods used are conservative (Finkbeiner *et al.* 2011). Savoca *et al.* (2020) evaluated bycatch in U.S. fisheries from 2010 to 2015, focusing on bycatch data from the U.S. National Bycatch Report, which estimates bycatch in major fisheries managed under the MSA. They found that longline fisheries are responsible for the majority of reported bycatch; however, shrimp trawls also capture NW Atlantic loggerhead turtles at a high rate. The U.S. National Bycatch Report, Update 3 (Benaka *et al.* 2019) summarized 2014 and 2015 annual bycatch estimates (Table 6) and average annual mortality data for the Greater Atlantic sink gillnet fishery from 2012 to 2016 (112 loggerheads killed per year). Evaluating observer, monitoring, and reported interaction data on sea turtle bycatch (all species) from 2013 to 2017, Upite *et al*. (2019) estimated mortality rate by gear type: trawl (48%), gillnet (73%), dredge (40%), vertical line 55% (or 61% including turtles that were not disentangled and assumed dead), and fish trap (57%). While all turtle species were combined to estimate these mortality rates, loggerheads comprised the vast majority of records for trawls and gillnets (Upite *et al.* 2019).

Table 6. Fisheries bycatch estimates from the U.S. National Bycatch Report, Update 3 (Benaka *et al*. 2019)

Fishery	Year	Bycatch
Atlantic/GoM highly migratory species pelagic longline	2015	242
Mid-Atlantic sink gillnet	2012-2016	141*
Mid-Atlantic otter trawl	2009-2013	231*
Mid-Atlantic scallop dredge	2009-2014	22*
GoM reef fish vertical line	2015	189
GoM coastal migratory pelagic gillnet	2015	1
GoM shrimp trawl	2015	46
SE Atlantic shrimp trawl	2015	111
SE Atlantic snapper-grouper bottom longline	2015	2
SE Atlantic/GoM shark bottom longline	2015	4
SE Atlantic coastal migratory pelagic troll	2015	1

*Annual average over years indicated

U.S. pelagic longline fisheries operate in the western and central North Atlantic, especially in the NE Distant (specifically Grand Banks off Newfoundland) and NE Coastal (specifically Georges Bank of Cape Cod) fishing areas (LaCasella *et al.* 2013; Swimmer *et al.* 2017; Stewart *et al.* 2018). Loggerheads off the western Florida shelf overlap with areas of high bottom longline fishing effort (Hardy *et al.* 2014). Bottom longline fisheries mortality rates may be high, depending on soak times, because turtles are unable to surface and breathe (Carlson *et al.* 2016). Bycatch in U.S. fisheries has declined significantly in recent decades; these trends indicate the success of previous regulatory requirements (Savoca *et al.* 2020). Analyzing 20+ years of U.S. pelagic longline observer data collected before and after implementation of extensive fisheries regulations (in 2004), Swimmer *et al.* (2017) found a 61% decline in Atlantic loggerhead bycatch rates. Longline modifications contributing to reduced bycatch include using fish baits rather than squid, wider circle hooks, and deeper hook depth (Swimmer *et al.* 2017; Swimmer *et al.* 2020). Safe handling and release procedures may reduce injury and mortality for bycaught turtles (Zollett and Swimmer 2019).

Trawl fisheries may represent a greater threat than longlines to the DPS because they mainly capture adults and large juveniles, individuals with the greatest reproductive value (Savoca *et al.* 2020). This is especially a concern when nesting females are killed. For example, loggerheads nesting in Georgia remain in shallow, nearshore habitats that overlap considerably with shrimp trawl fisheries (Scott *et al.* 2013). Between 2007 and 2015, the total annual estimated bycatch mortality in SE U.S. otter trawl fisheries was 277 loggerheads (NMFS 2021). Between 2014 and 2018, bottom trawl fisheries captured an estimated 571 loggerheads in the U.S. mid-Atlantic and 12 loggerheads in Georges Bank, with an estimated 272 total mortalities (Murray 2020). Turtle Excluder Device (TED) usage has reduced loggerhead bycatch in Mid-Atlantic bottom trawl gear from 353 annually (2005 to 2008) to 114 annually (2014 to 2018) (Warden 2011; Murray 2020). Skimmer trawl vessels 40 feet (12.2 meters) and greater in length are required to use TEDs in their nets (84 FR 70048; December 20, 2019). On April 20, 2021, NMFS published an advance notice of proposed rulemaking and solicited public comments on the possibility of modifying TED related requirements for skimmer trawl vessels less than 40 feet in length operating in the SE U.S. shrimp fisheries (86 FR 20475). Bottom trawl fisheries may result in high mortality rates due to decompression sickness; for example, using ultrasound, Parga *et al.* (2020) detected gas embolisms in 28 bycaught turtles, of which 12 (42%) died on-board, and another three (11%) died within 6 days of release. Large juvenile and adult loggerheads also overlap with U.S. commercial bottom trawl and scallop dredge fisheries throughout the mid and South Atlantic Bight (Arendt *et al.* 2012; Murray and Orphanides 2013; Murray 2015). Between 2015 and 2019, the Atlantic sea scallop dredge fishery had an annual average of 155 interactions and 53 mortalities of loggerhead turtles (Murray 2021).

Net fisheries cause significant mortality as well. Between 2012 and 2016, mid-Atlantic and Georges Bank sink gillnet fisheries captured an estimated total of 705 loggerheads annually, with 557 mortalities (Murray 2018). Between 1998 and 2017, the U.S. SE gillnet fishery bycaught 17 loggerheads, including four (24%) at-vessel deaths (Kroetz *et al.* 2020). While the number of pound nets in the Pamlico-Albemarle fishery declined by more than 60% between 1995 and 2009, bycatch increased from 447 to 946 loggerheads (including three deaths in 15 years), likely reflecting a change in turtle abundance within the area (McNeill *et al.* 2018).

State fisheries are also a threat to loggerheads, but bycatch data are limited. From 2010 to 2019, observers documented three loggerheads in gillnet gear and 19 loggerheads in bottom otter trawl gear fisheries occurring in state waters (NMFS unpublished data 2021). Between 2010 and 2019, the Sea Turtle Disentanglement Network in the NMFS Greater Atlantic Region reported 14 loggerhead entanglements in vertical fishing lines in state waters, specifically entanglements in blue crab and conch gear (Sea Turtle Disentanglement Network unpublished data 2021). Additionally, the Virginia state observer program documented 3 loggerheads in gillnet gear in 2019 (Virginia Marine Resources Commission unpublished data 2021), and Maryland has reported sea turtle interactions in state water conch pots and trawls.

Injuries due to fisheries interactions are often observed on stranded turtles. Of 9,950 loggerheads stranded in Florida between 1997 and 2009, 418 (4%) exhibited evidence of fishery gear interactions, with 73% of these attributed to hook and line (Adimey *et al.* 2014). Of 74 turtles stranded within or along the mouth of the Chesapeake Bay, 16 (22%) exhibited evidence of acute fishery interactions, though only one had compromised health as a result of the fishery interaction (Barco *et al.* 2016).

Strandings in Texas provide evidence for illegal fishing in Texas waters (public comment submitted by D. Shaver, Division of Sea Turtle Science and Recovery of Padre Island National Seashore, National Park Service, 2020). The number of adult and juvenile loggerhead sea turtles (over 10 cm straight carapace length) that have stranded on the Texas coast averaged 110 individuals/year over the past 10 years (2010–2019; less than 19% of these individuals were found alive) and have increased during the most recent eight years, with 131 in 2019 alone. Illegal fisheries may be contributing to these impacts since many intact dead loggerheads are found with evidence of forced submergence apparent at necropsy, including 20 documented in Texas during 2019. Small vessels from Mexico, called *lanchas*, illegally fish in GoM waters off the south Texas coast. The U.S. Coast Guard interdicted a total of 287 *lanchas* for suspicion of illegal fishing in 2018, 2019, and 2020, and based on the facts of each interdiction, they prepared 248 cases that had evidence of illegal fishing (NOAA 2021). This threat appears to be increasing (public comment submitted by D. Shaver, Division of Sea Turtle Science and Recovery of Padre Island National Seashore, National Park Service, 2020).

Fisheries outside of the U.S. EEZ

Loggerheads comprise a large proportion of bycatch in North Atlantic fisheries. Canadian swordfish and tuna longline fishery bycatch averaged 1,200 loggerheads annually from 2002 to 2008 (Paul *et al.* 2010). On the high seas of the Atlantic Ocean (only some of which overlaps with the range of the NW Atlantic DPS), through analysis of fisheries observer collected data from the Taiwanese deep-set longline fishery, loggerhead bycatch rates of 0.0128 to 0.0239 per thousand hooks and 34.3% mortality rates were determined, which reflect long soak times; the author suggests gear and bait modification to reduce sea turtle bycatch and increase survival rates (Huang 2015).

Juvenile loggerheads also comprise a large proportion of bycatch in Mediterranean fisheries. Since the 1980s, western Mediterranean fisheries have exhibited some of the highest levels of loggerhead bycatch globally; drifting longlines, bottom trawls, and trammel nets regularly take juveniles of the NW Atlantic DPS (Clusa *et al.* 2016). Lewison *et al.* (2014) found high intensity longline bycatch in the Mediterranean Sea. In the SW Mediterranean, the Spanish longline fleet has been estimated to take 10,656 loggerhead turtles annually, with mortality (including post-release mortality) estimated at 32% to 38% (de Quevedo *et al.* 2013). As a result of high bycatch mortality, de Quevedo *et al.* (2013) concludes that the Mediterranean Sea is a “dead end” for juvenile loggerheads of the NW Atlantic DPS. Such impacts would be reflected in future reductions in abundance and productivity.

Bycatch also occurs in fisheries throughout the Greater Caribbean RU. Juvenile loggerheads are incidentally captured by industrial shrimp trawlers off eastern Venezuela, Trinidad and Tobago, and NW Guyana (Alio *et al.* 2010). Cuevas *et al.* (2018) describe high artisanal longline and gillnet fishing effort along the Yucatan Peninsula, with sea turtle (all species) bycatch rates up to 0.72 individuals/1000 fishing hooks/season.

Illegal, unreported, and unregulated fishing is likely a large, though unquantified, threat to the DPS. Due to lack of reporting, we have no estimates of bycatch rates in illegal fisheries. Because turtle bycatch in these fisheries is not regulated or reported, there is no incentive to use turtle-friendly gear, check gear frequently, or release turtles; therefore, mortality rates are likely high.

Summary

Although gear requirements have reduced the impacts of trawl and longline fisheries, domestic and foreign fisheries continue to kill thousands of loggerhead turtles annually and are likely to limit recovery in the foreseeable future. Unquantified IUU fisheries may pose an even greater threat to the DPS because they are not required to use any mitigation measures that would reduce bycatch or mortality. Bolten *et al*. (2019) conclude, and we agree, that bycatch in commercial fisheries remains a significant threat to the DPS, and the following bycatch-focused recovery objectives have not been met:

- A peer-reviewed strategy is developed and fully implemented to minimize fishery interactions and mortality for each domestic commercial fishing gear type that has loggerhead bycatch: not met. The strategy has not yet been developed, but progress has been made in some fisheries to reduce loggerhead bycatch.
- A peer-reviewed strategy is developed and fully implemented in cooperation with relevant nations to minimize fishery interactions and mortality of loggerheads in foreign EEZs and on the high seas: not met. There are no binding requirements to reduce bycatch of NW Atlantic loggerheads in foreign EEZs or the high seas.
- A peer-reviewed strategy is developed and fully implemented to quantify, monitor, and minimize effects of trophic changes on loggerheads (e.g., diet, growth rate, fecundity) from fishery harvests and habitat alterations: not met. No progress has been made on this criterion.

Fisheries bycatch reduces abundance; it also likely reduces productivity by removing those individuals (i.e., adults and large juveniles) that survived decades of development and have the greatest potential to contribute to future generations. Reductions in abundance and productivity reduce DPS resilience. Therefore, we conclude that fisheries bycatch is the greatest threat to the DPS.

2.3.2.6 Climate Change

To evaluate the impact of climate change on the DPS, we used the best available data, which includes the Intergovernmental Panel on Climate Change (IPCC) Special Report on Oceans and Cryosphere (IPCC 2019). The Revised Guidance for Treatment of Climate Change in NMFS' ESA Decisions (NMFS 2016) requires us to use climate indicator values projected under the IPCC Representative Concentration Pathway (RCP) 8.5, which reflects a continued increase of greenhouse gas emissions and assumes that few mitigation measures will be implemented.

The IPCC (2019) reports the following consequences of climate change on sea turtles with high confidence, which is an evaluation of the underlying evidence and agreement in the conclusion. Loss of sandy beaches, due to sea level rise and storm events, reduces available nesting habitat (Fish *et al*. 2005; Fuentes *et al*. 2010; Reece *et al*. 2013; Katselidis *et al*. 2014; Patino-Martinez *et al*. 2014; Pike *et al*. 2015; Marshall *et al*. 2017). Storms, waves, and sea level rise are likely to increase erosion and sediment loss. Changes in beach morphology, dune scarping, vegetation loss, and reduction in beach area are likely to reduce availability of sea turtle nesting sites, and potential for landward migration of the beach profile is limited due to human development. Temperature directly affects important sea turtle life history traits, including: hatchling size, sex, viability, and performance (Hays *et al*. 2003; Pike 2014; Dudley *et al*. 2016; Santos *et al*. 2017). One of the greatest concerns is the effect of temperature on hatchling emergence rates and sex ratios (Santidrián Tomillo *et al*. 2014; Patrício *et al*. 2017). Changes in ocean temperature indirectly impact sea turtles by altering the abundance and distribution of their prey (Polovina

2005; Polovina *et al*. 2011; Doney *et al*. 2012; Sydeman *et al*. 2015; Briscoe *et al*. 2017). Additionally, sea turtles require habitat associated with bathymetric and mesoscale features that aggregate their prey, and the persistence and location of these features are linked to variations in climate (Baez *et al*. 2011; Bjorndal *et al*. 2017; Santora *et al*. 2017). The IPCC (2019) states with high confidence that climate change is likely to alter foraging success, juvenile recruitment, breeding phenology, growth rates, and population stability. The following sections discuss these changes as they apply to the DPS.

Sea Level Rise and Storm Events

Melting of glaciers and ice sheets is the primary driver of sea level rise, which has accelerated in recent years (very high confidence; IPCC 2019) as indicated in a study of ice loss from the Greenland Ice Sheet (Shepherd *et al*. 2019). By 2100 (relative to 2005), the IPCC projects global mean sea level rise of 0.84 m with a likely range of 0.61 to 1.1 m, where likely refers to 66 to 100 percent probability (IPCC 2019). A NOAA-led interagency report projects a range of 0.3 to 2.5 m sea level rise globally by 2100 (Sweet *et al.* 2017). Within the DPS nesting range, sea level rise is likely to inundate 81% of current beach nesting habitat, particularly in SE Florida (Fuentes *et al.* 2020). Sea level rise will make nesting beaches more susceptible to erosion (Von Holle *et al.* 2019; Lyons *et al.* 2020). In addition, hurricanes, storm surge, high tides, waves, and changes in shoreline geology appear to be increasing in frequency and magnitude due to climate change (Dewald and Pike 2014; Fuentes *et al.* 2019), further reducing suitable nesting habitat (Reece *et al.* 2013). Beach armoring is likely to become an even greater threat as sea level rises because it can limit the capacity for shorelines to shift naturally in the face of climate change (Von Holle *et al*. 2019). Biddiscombe *et al.* (2020) found that 24% of North American nesting beaches (all sea turtle species) were highly developed, and 16% of Florida beaches were backed by hard anthropogenic coastal development. Subsidence of land and sea level rise are also emerging threats to loggerhead nesting habitat in Texas (submitted as public comment by D. Shaver, Division of Sea Turtle Science and Recovery of Padre Island National Seashore, National Park Service, 2020).

Temperature Increases

Temperature increases are likely to change the timing and location of nesting. Azanza-Ricardo *et al.* (2017) found that the peak in loggerhead nesting on Guanahacabibes Peninsula, Cuba, has shifted from mid-June (1998-2008) to early June (2008-2014) and occurred during May in 2015; the end of the nesting season has shifted 36 days earlier between 1995 and 2015. Modeling climate variables, Fuentes *et al.* (2020) found that climatically suitable nesting habitat for the DPS is predicted to decrease by 4% by 2050 and shift northward.

Increases in sand temperature are likely to substantially reduce hatching success (to less than 20% in some areas) and induce feminization (i.e., increasingly female-biased sex ratios up to 100% at some beaches) across NW Atlantic loggerhead nesting beaches (Monsinjon *et al.* 2019a). As a result of nest temperatures near or above the lethal maximum in Boca Raton, Florida, hatching success has decreased from above 70% in 2014 to only 42.7% in 2016 and 55.5% in 2017 (Bladow and Milton 2019). With expected increases in temperatures, this increase in mortality is likely to continue to increase. Eggs incubated at higher temperatures may also produce smaller and less fit hatchlings (Fisher *et al.* 2014; Erb *et al.* 2018).

High nest temperatures are already producing strongly female-biased hatchling sex ratios in Palm Beach County, Florida, where the majority of hatchlings sampled between 2010 and 2013 were female; however, sex ratio estimates based on proxies should be interpreted with caution, as they do not account for developmental variability, variable temperatures, and the impact of moisture (Wyneken and Lolavar 2015). Long-term feminization of hatchlings could result in highly skewed breeding sex ratios, making it difficult for females to find mates. One study, however, found that females nesting on beaches in the Gulf of Mexico still have access to large numbers of males (Lasala *et al.* 2018). Another study found that 67% of 2,217 loggerheads captured in research trawls from St. Augustine, Florida to Winyah Bay, South Carolina between 2000 and 2019 were female, and that percentage declined slightly over time (Arendt *et al.* 2021). Therefore, there is uncertainty regarding sex ratio plasticity and population-level impacts.

Warming oceans have also led to shifts in behavior. The weakening of the Atlantic meridional overturning circulation, which includes the Florida Current and Gulf Stream (Caesar *et al.* 2021), is likely to impact the survival and offshore dispersal of NW Atlantic loggerheads (Putman *et al.* 2010; Lamont *et al.* 2015). Bjorndal *et al.* (2013) attributes declines in nesting and juvenile somatic growth rates between 1997 and 2007 to a large scale regime shift, resulting in reduced prey availability. It is possible that this shift has resulted in additional time spent foraging prior to remigration or first nesting, resulting in stalled population growth. However, Patel *et al.* (2021) projects an increase in loggerhead thermal habitat and seasonal duration in northern regions of the NW Atlantic shelf foraging area over the next 80 years. More research is needed in this area.

Finally, modeling based on 189 cold-stunned loggerheads in North Carolina between 2010 and 2015 indicates that cold stunning (or hypothermia) events may increase as a result of climate change (Niemuth *et al.* 2020). Large cold-stunning events occur when turtles do not migrate south before water temperatures drop during autumn or during extreme cold weather snaps (Pirhalla *et al.* 2015; Griffin *et al.* 2019; Niemuth *et al.* 2020). Cold-stunned turtles gradually cease swimming and may become stranded on beaches when washed ashore by tidal activity and wind. While cold-stunned turtles likely have little chance of surviving under natural conditions, those found alive can be rehabilitated and released (Robinson *et al.* 2020). Between 2008 and 2016, 194 loggerheads were admitted to the New England Aquarium following cold-stunning events in the NE United States (Innis *et al.* 2019; McNally and Innis 2020), with survival rates of approximately 87% (Innis *et al.* 2019). Some of these data are included in the Massachusetts' total estimate of cold-stunned strandings between 2009 and 2019, when there were 424 cold-stunned turtles, of which 241 were admitted for rehabilitation: 141 were released, 33 died, 1 was deemed non-releasable, and 66 were transported out of the region (Sea Turtle Stranding and Salvage Network unpublished data 2021).

Ocean Acidification and Prey Availability

It is very likely that the ocean has taken up 20 to 30 percent of total anthropogenic carbon dioxide emissions since the 1980s, leading to ocean acidification rates of 0.017 to 0.027 per decade since the late 1980s (IPCC 2019). It is virtually certain that continued carbon uptake through 2100 will exacerbate ocean acidification, which is projected with high confidence to increase by 100 to 150% (IPCC 2019). Loggerhead turtles are foraging generalists, meaning that they forage on a wide variety of prey; however, their prey often include shell-forming (i.e., calcifying) organisms, which requires the synthesis of calcium carbonate from the calcium and carbonate ions found in seawater. In a more acidic environment, a greater amount of hydrogen ions compete for the available carbonate ions. Thus, ocean acidification may reduce the abundance of calcifying organisms. We conclude that ocean acidification is likely to reduce availability of loggerhead prey, which will diminish the productivity of the DPS.

Summary

Species with high fecundity and low survival of early life stages, such as sea turtles, have increased vulnerability to climate change and elevated levels of environmental variability (Halley *et al.* 2018). In a comparison of 58 sea turtle populations, Fuentes *et al*. (2013) concluded that the NW Atlantic loggerhead regional management unit (equivalent to the DPS) is likely to be the most resilient to climate change, due to its large population size and broad nesting beach distribution. Bolten *et al.* (2019) identified climate change as an emerging threat that will have major effects on the DPS in profound and varied ways. Erosion of nesting habitat, inundation of nests, and reduction of hatching success (due to increased incubation temperature, which also may result in skewed sex ratios) reduce productivity in the short-term and abundance in the long-term. Changes to ocean temperatures and circulation are likely to change migratory paths, reduce prey availability, and alter the location and predictability of prey accumulation, while ocean acidification and oxygen depletion may further stress prey populations and reduce availability; these changes are likely to reduce productivity by lengthening time to maturity and remigration intervals. We conclude that climate change is an increasing threat that adversely modifies the nesting and foraging habitat of the DPS and may result in less stable population trends in the future.

2.3.2.7 Vessel Strikes

Vessel strikes can lead to the injury, debilitation, and/or mortality of loggerheads. Some progress has been made to study vessel interactions with sea turtles, and in Florida, a pilot project incorporates education and voluntary speed reduction in a vessel strike hotspot (Bolten *et al*. 2019). Of 19,111 loggerheads founded stranded in Florida between 1986 and 2014, 5,983 (31.3%) exhibited either definitive or probable vessel strike injuries; the majority (76%) of these vessel-related strandings occurred between March and August, corresponding to nesting and mating seasons (Foley *et al.* 2019a). Based on these data, Foley *et al.* (2019a) estimated that 712 to 2,292 loggerheads are killed annually in Florida as a result of vessel strikes. Of 70 loggerheads found stranded within or just outside the Chesapeake Bay, 15 (21%) died as a result of vessel strike (Barco *et al.* 2016). The loss of reproductive individuals reduces the abundance and productivity of the DPS, which also reduces its resilience. We conclude that vessel strike is a moderate threat to the DPS.

2.3.2.8 Dredging

Harbor and channel dredging can indirectly affect sea turtles by degrading habitat, such as altering benthic foraging areas, decreasing the number and abundance of prey species, and reducing water quality by increasing turbidity and releasing potential contaminants into the water column (Ramirez *et al.* 2017). Trailing suction hopper dredges and other support vessels may strike slow-moving sea turtles or entrain sea turtles in the draghead, as it moves across the seabed. Such direct impacts often result in severe injury and/or mortality. At least 431 loggerheads were taken by hopper dredges along the U.S. Atlantic and Gulf of Mexico coasts from 1995 to 2019 (U.S. Army Corps of Engineers' Operations and Dredging Endangered Species System, accessed June 1, 2021). Annual take levels varied widely depending on dredging effort, location, and timing (i.e., range of 6 to 39 takes annually); for example, hopper dredges in Virginia alone took 19 loggerheads in 2020. The overall 25-year average take is approximately 18 loggerhead turtles per year. These totals include maintenance and expansion dredging done in shipping channels, as well as offshore dredging of sand borrow sites to mine for sand used in beach nourishment projects. We conclude that dredging poses a low-level threat to the DPS.

2.3.2.9 Power Generation

Power plants impinge sea turtles on their intake screens and entrain turtles in their cooling water structures. Since 2015, the Brunswick Steam Electric Plant has killed 10 loggerheads and released another 10 loggerheads. The St. Lucie Power Plant had 250 non-lethal entrainments in 2017 (see SER-2006-00832 in the NMFS' Environmental Consultation Organizer). Wave turbines may also entrain and kill turtles, and electromagnetic fields around wind and wave turbine installations may cause sensory disruption in migrating turtles (Bolten *et al.* 2019); however, population impacts are unknown. We do not have adequate data to evaluate the magnitude of this threat on the DPS.

2.4 Synthesis

The loggerhead turtle NW Atlantic DPS was listed as a threatened species on September 22, 2011 (76 FR 58868). After reviewing the best available data, including new information that has become available since the 2009 Status Review Report (Conant *et al*. 2009), we agree that it meets the DPS Policy criteria because it is reproductively and geographically discrete from all other loggerhead DPSs and is significant to the species, as the only DPS nesting and foraging in the NW Atlantic Ocean.

Like all sea turtles, the DPS exhibits a complex life cycle of several life stages (i.e., hatchling, juvenile, and adult), occurring across wide-spread and diverse habitats. The majority of nesting occurs primarily in Florida with smaller nesting aggregations in other States and throughout the Wider Caribbean. Foraging occurs at different locations throughout the North Atlantic Ocean (including the GoM and Wider Caribbean) and Mediterranean Sea, depending on life stage and foraging strategy.

The DPS exhibits high levels of nesting, with more than 110,000 nests annually; however, the DPS is subdivided into five RUs and at least eight genetic subpopulations, ranging from low to moderate abundance. Multiple RUs provide spatial and genetic diversity to the DPS, which increases its likelihood of persistence via metapopulation dynamics (i.e., losses at one subpopulation may be buffered by gains at another). The vast majority of nesting (89%) occurs within the Peninsular Florida RU, whose average annual growth rate is not significantly different from zero. The overall nesting trend thus appears to be stable, neither increasing nor decreasing, for over two decades. Increased nesting in the Northern RU may have helped to stabilize the overall trend; however, the lack of growth in the largest RU is reason for concern. This lack of population growth has been attributed to slower juvenile growth rates and delayed maturity, reduced clutch frequency, and longer remigration intervals, all of which may reflect environmental changes and a reduction in prey availability (Ceriani *et al.* 2019). However, we cannot rule out the impact of numerous anthropogenic threats that continue to remove thousands of turtles from the population.

The greatest threat to the DPS is fisheries bycatch, which results in the death of thousands of turtles annually. High bycatch mortality rates in the eastern Atlantic and Mediterranean Sea have persisted for decades, with thousands of small juveniles removed from the population, likely resulting in reduced population recruitment and productivity. Trawl and longline U.S. fisheries remain a significant concern, removing large juveniles and adults from the population and reducing abundance and productivity. Gear modifications have reduced impacts in some U.S. fisheries; however, high bycatch mortality rates in foreign fisheries are likely to continue without additional regulation.

Habitat modification is a major threat that is further exacerbated by climate change. In terrestrial habitats, beach erosion, erosion control methods, and coastal development reduce availability of suitable nesting habitat. Artificial lighting, beach use, beach debris, shoreline structures, and coastal development deter nesting females and prevent hatchlings from reaching the sea. Additionally in marine environments, important foraging habitats are modified by pollution, contaminants, and oil spills. Adults and large juveniles are killed and injured by derelict fishing gear, plastics, and other marine debris, either through ingestion or entanglement. Thus, habitat modification reduces abundance by removing individuals from the population and reduces productivity by reducing nesting and hatching success rates.

Climate change adversely modifies essential habitat. The erosion of nesting habitat, inundation of nests, and reduction of hatching success due to increased incubation temperature reduces productivity in the short-term and abundance in the long-term. Changes to ocean temperatures and circulation are likely to lengthen remigration intervals and age to maturity (reducing nesting rates) by changing migratory paths, reducing prey availability, and altering the location and predictability of prey accumulation. Ocean acidification and oxygen depletion are likely to further stress prey populations and reduce availability. In recent years and at local scales, the impacts of climate change have caused greater nesting beach erosion, reduced hatching success, and increased feminization, reducing productivity. These impacts are likely to increase in the future, further reducing productivity and contributing to endangerment of the species in the foreseeable future.

Several other threats reduce the abundance of adults and large juveniles. Vessel strikes likely kill hundreds to thousands of loggerheads annually. Hundreds of loggerheads are legally harvested in Haiti and Grenada annually and are likely illegally harvested at some levels in other areas within the range of the DPS. Additional threats include egg poaching, predation, disease, dredging, and power plants. Regulatory mechanisms do not adequately address any threat.

This species' long evolutionary history, persisting throughout millions of years of large-scale climatic and sea-level changes, demonstrates its potential to adapt. The relevant question, however, is whether the DPS can adapt fast enough, given unprecedented rates of climate change in the context of other anthropogenic threats. Broad nesting and foraging distributions likely allow for latitudinal shifts; however, threats occur throughout the range of the DPS. Large abundance and population substructure likely provide the DPS with adequate resilience to withstand stochastic disturbances, but again, the magnitude and multitude of threats reduces this resilience. The DPS nests at multiple beaches and forages throughout the North Atlantic Ocean, such that the loss of a single nesting beach or foraging area may be tolerated. While this redundancy would not protect the DPS from large-scale environmental changes, such as those likely to occur as a result of climate change, it likely protects the DPS against local catastrophic events.

After evaluating the best available information, we have determined that the status of the DPS remains unchanged. It continues to be at risk from intense (fisheries bycatch and habitat modification), numerous (vessel strike, overutilization, predation, disease, dredging), and increasing (climate change) threats. Together, these threats have prevented meaningful population growth in recent decades. We conclude that the DPS should retain its threatened status.

3.0 RESULTS

3.1 Recommended Classification

_____**Downlist to Threatened**
_____**Uplist to Endangered**
_____**Delist** *(Indicate reason for delisting per 50 CFR 424.11):*
  _____*Extinction*
  _____*Recovery*
  _____*Original data for classification in error*
_x___**No change is needed**

4.0 RECOMMENDATONS FOR FUTURE ACTIONS

The publication of this 5-year review follows recovery planning efforts led by NMFS and USFWS (Bolten *et al*. 2019). The effort provided recommendations for future actions, which we summarize here:

- Protect nesting beach habitat through long-term nesting beach protection and practices that maintain these beaches as natural environments;
- Improve monitoring and reporting of legal and illegal harvest of turtles; and
- Continue U.S. efforts to reduce fisheries bycatch and encourage international bycatch mitigation efforts.

We also recommend increased efforts to reduce marine pollution, especially plastics and discarded fisheries gear, and the removal of marine debris from foraging and internesting areas. Finally, we recommend additional research into the impacts of climate change, especially those that have the potential to reduce productivity of the DPS. Such research should include monitoring for increases in remigration intervals and age of first reproduction.

5.0 REVIEWERS and REFERENCES

5.1 Reviewers

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U.S. FISH AND WILDLIFE SERVICE
5-YEAR REVIEW
NORTHSWEST ATLANTIC OCEAN LOGGERHEAD DPS

Current Classification: Threatened

Recommendation resulting from the 5-Year Review:

____ Downlist to Threatened
____ Uplist to Endangered
____ Delist
_X__ No change needed

Review Conducted By: Jennifer Schultz and Karen Frutchey

FIELD OFFICE APPROVAL:

Division Manager, Classification and Recovery, Florida Ecological Services Field Office, U.S. Fish and Wildlife Service

Signature LOURDES MENA Digitally signed by LOURDES MENA
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REGIONAL OFFICE APPROVAL:

Assistant Regional Director- Ecological Services, Southeast Region, U.S. Fish and Wildlife Service

Signature JOHN TIRPAK Digitally signed by JOHN TIRPAK
Date: 2023.03.01 12:52:02 -05'00'

NATIONAL MARINE FISHERIES SERVICE
5-YEAR REVIEW
NORTHWEST ATLANTIC OCEAN LOGGERHEAD DPS

Current Classification: Threatened

Recommendation resulting from the 5-Year Review

____ Downlist to Threatened
____ Uplist to Endangered
____ Delist
_X__ No change is needed

Review Conducted By: Jennifer Schultz and Karen Frutchey

HEADQUARTERS APPROVAL:

Assistant Administrator, NOAA Fisheries

__X__Concur _____ Do Not Concur

Signature RAUCH.SAMUEL.D.III.1365850948 Digitally signed by RAUCH.SAMUEL.D.III.1365850948 Date: 2023.03.15 14:36:03 -04'00' **Date**__________