



**Billing Code 3510-22-P**

**DEPARTMENT OF THE INTERIOR**

**Fish and Wildlife Service**

**50 CFR Part 17**

**DEPARTMENT OF COMMERCE**

**National Oceanic and Atmospheric Administration**

**50 CFR Parts 223 and 224**

**[Docket No. 200717-0190]**

**RIN 0648-XF748**

**Endangered and Threatened Wildlife; 12-Month Finding on a Petition to Identify the Northwest Atlantic Leatherback Turtle as a Distinct Population Segment and List It as Threatened Under the Endangered Species Act**

**AGENCY:** National Marine Fisheries Service (NMFS), National Oceanic and Atmospheric Administration (NOAA), Commerce; U.S. Fish and Wildlife Service (USFWS), Interior.

**ACTION:** Notification of 12-month petition finding.

**SUMMARY:** We, NMFS and USFWS, announce a 12-month finding on a petition to identify the Northwest Atlantic population of the leatherback turtle (*Dermochelys coriacea*) as a distinct population segment (DPS) and list it as threatened under the Endangered Species Act (ESA). In response to the petition, we completed a comprehensive status review of the species, which also constitutes the 5-year review of the species, to determine potential DPSs following the Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the ESA and to perform extinction risk

analyses. Based on the best scientific and commercial data available, including the Status Review Report, and after taking into account efforts made to protect the species, we conclude that seven populations would meet the discreteness and significance criteria for recognition as DPSs, including the Northwest Atlantic population. However, even if we were to list them separately, all seven DPSs would meet the definition for endangered species (*i.e.*, they are in danger of extinction throughout all or a significant portion of their range). The species is already listed as endangered throughout its range. We have determined that the listing of DPSs is not warranted, and therefore we do not propose any changes to the existing global listing.

**DATES:** This finding was made on [*insert date of publication in the FEDERAL REGISTER*].

**ADDRESSES:** The Status Review Report are available on NMFS' website at <https://www.fisheries.noaa.gov/species/leatherback-turtle>.

**FOR FURTHER INFORMATION CONTACT:** Jennifer Schultz, NMFS Office of Protected Resources, (301) 427-8443, [jennifer.schultz@noaa.gov](mailto:jennifer.schultz@noaa.gov). Persons who use a Telecommunications Device for the Deaf (TDD) may call the Federal Information Relay Service (FIRS) at 1-800-877-8339, 24 hours a day and 7 days a week.

**SUPPLEMENTARY INFORMATION:**

**Background**

The leatherback turtle species as a whole was listed as an endangered species (one determined to be threatened with worldwide extinction) (35 FR 8491; June 2, 1970), under the Endangered Species Conservation Act of 1969, the precursor statute to the ESA (16 U.S.C. 1531 *et seq*). When the ESA was enacted in 1973, it specifically provided for

continuity with the lists previously in effect under the Endangered Species Conservation Act. Section 4(c)(3) of the ESA directed that species on the lists of endangered foreign or native wildlife at the time the ESA took effect would be deemed “endangered species” under the ESA without interruption. *See* 39 FR 1444 (January 9, 1974) (explaining transition provisions); 39 FR 1158, 1172 (January 4, 1974) (setting out the final list of “endangered foreign wildlife,” including “Turtle, Leatherback” at 50 CFR 17.11).

On September 20, 2017, the Blue Water Fishermen’s Association petitioned NMFS and USFWS (together, the Services) to identify the Northwest (NW) Atlantic leatherback turtle population as a DPS and to list it as threatened under the ESA. On December 6, 2017, NMFS published a “positive” 90-day finding in the **Federal Register** (82 FR 57565) announcing the determination that the petition presented substantial information indicating that the petitioned action may be warranted. At that time, NMFS also solicited information on leatherback turtles and announced that it would commence, jointly with USFWS, a status review of the entire listed species, pursuant to ESA section 4(b)(3)(A) and 50 CFR 424.14. The resulting Status Review Report includes all information used to evaluate the petitioned actions and explains the process followed by the Status Review Team (*i.e.*, the Team). The following summarizes that information; for additional details, please see the Status Review Report (see ADDRESSES).

#### **ESA Statutory, Regulatory, and Policy Provisions and Evaluation Framework**

Under the ESA, the term “species” includes any subspecies of fish or wildlife or plants, and any DPS of any vertebrate fish or wildlife which interbreeds when mature (16 U.S.C. 1532(16)). The Services adopted a joint policy clarifying their interpretation of the

phrase “distinct population segment” for the purposes of listing, delisting, and reclassifying a species under the ESA (“Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the Endangered Species Act,” 61 FR 4722 (Feb. 7, 1996; “DPS Policy”). The DPS Policy stipulates two elements that must be considered: (1) discreteness of the population segment in relation to the remainder of the species to which it belongs; and (2) the significance of the population segment to the species to which it belongs.

Section 3 of the ESA defines an endangered species as any species which is in danger of extinction throughout all or a significant portion of its range and a threatened species as one which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range (16 U.S.C. 1532(6) and (20)). Thus, we interpret an “endangered species” to be one that is presently in danger of extinction. A “threatened species,” on the other hand, is not presently in danger of extinction, but is likely to become so within the foreseeable future (that is, within a specified later time). In other words, the primary statutory difference between a threatened and endangered species is the timing of when a species may be in danger of extinction, either presently (endangered) or within the foreseeable future (threatened). The ESA uses the term “foreseeable future” to refer to the time over which identified threats are likely to impact the biological status of the species. The duration of the “foreseeable future” in any circumstance is inherently fact-specific and depends on the particular kinds of threats, the life-history characteristics, and the specific habitat requirements for the species under consideration. The existence of threats to a species and the species’ response to such threats are not, in general, equally predictable or

foreseeable. Hence, in some cases, the ability to foresee a threat to a species is greater than the ability to foresee the species' exact response, or the timeframe of such a response, to that threat. For purposes of making this 12- month finding, the relevant consideration is whether the species' population response (*i.e.*, abundance, productivity, spatial distribution, diversity) is foreseeable, not merely whether the emergence of a threat is foreseeable. The foreseeable future extends only as far as we are able to reliably predict the species' population response to threats.

Pursuant to the ESA and our implementing regulations, we determine whether a species is threatened or endangered based on any one or a combination of the following ESA section 4(a)(1) factors or threats (16 U.S.C. 1533(a)(1), 50 CFR 424.11(c)):

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; or
5. Other natural or manmade factors affecting its continued existence, which could include but are not limited to: fisheries bycatch; vessel strikes; pollution (including marine debris and plastics, contaminants, oil and gas activities, and derelict fishing gear); natural disasters; climate change; and oceanographic regime shifts.

Section 4(b)(1)(A) of the ESA requires us to make listing determinations based solely on the best scientific and commercial data available after conducting a review of the status of the species and after taking into account efforts being made by any State or

foreign nation or political subdivision thereof to protect the species' existence (16 U.S.C. 1533(b)(1)(A)).

### **Approach to the Status Review**

The Services convened a team of NMFS and USFWS biologists (*i.e.*, the Team) to gather and review the best available scientific and commercial data on the leatherback turtle, assess the discreteness and significance of populations by applying the DPS Policy, evaluate the extinction risk of any population segments that meet the DPS criteria, and document all findings in a report (*i.e.*, the Status Review Report). Although the petitioner requested evaluation only of the NW Atlantic leatherback population, we instructed the Team to perform a comprehensive status review to identify and evaluate the status of all potential DPSs.

The Team compiled information on leatherback turtle life history, biology, ecology, demographic factors, and threats. This included the information received in the petition and in response to the **Federal Register** request associated with the 90-day finding (82 FR 57565; December 6, 2017). The Team also requested leatherback nesting data from beach monitoring programs. To evaluate recent abundance and trends, unpublished nesting beach monitoring datasets were often the best available data (*i.e.*, most recent and relevant). The Team assessed these data in terms of standardization (*i.e.*, the use of standardized methodology), consistency (*i.e.*, consecutive seasonal data collection), and duration of data collection (*i.e.*, the number of years that data were collected). When evaluating threats, peer-reviewed information, specifically primary research with large sample sizes and long-term sampling duration, was often the best available data. In some locations, reports from governments or non-governmental

organizations and expert opinion constituted the best available information. The Team also addressed the source and magnitude of any uncertainty and the impact on its conclusions.

The Team evaluated the discreteness and significance of each population and provided their evaluation of whether each population would meet the criteria of the DPS Policy. The DPS Policy states that a population of a vertebrate species may be considered discrete if it satisfies one of the following conditions: (1) it is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors (quantitative measures of genetic or morphological discontinuity may provide evidence of this separation); or (2) it is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the ESA (61 FR 4722, February 7, 1996). While the Team used the term “DPS” in describing and discussing populations that they concluded meet the requirements of discreteness and significance, it is important to note that the DPS term is used throughout the Status Review Report for ease of reference only. A DPS is formally recognized under the ESA only upon a listing action by the Services, and the Services cannot delegate authority to take formal listing actions to status review teams. The information compiled by the Team must be reviewed by the Services, which retain responsibility for making the listing determination after complying with all the requirements of Section 4 of the ESA and considering agency policies. Because we ultimately conclude for the reasons discussed in this finding that it would not be appropriate to disaggregate the existing global listing into DPSs, references in the

Status Review Report (and in this finding when we are reviewing the information presented by the Team) must be understood as references to potential or hypothetical DPSs only.

The Team evaluated significance in terms of the importance of the population segment to the overall welfare of the species, such as: (1) persistence of the population segment in an unusual or unique ecological setting; (2) evidence that loss of the population segment would result in a significant gap in the range of the taxon; (3) evidence that the DPS represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historic range; or (4) evidence that the population segment differs markedly from other populations of the species in its genetic characteristics.

For each population segment that the Team determined would meet the criteria of the DPS Policy (which the Team and we refer to as a “DPS” for ease of reference), the Team performed an extinction risk analysis, which involved the evaluation of demographic factors and threats. Demographic factors reflect the impact that operative threats have had on the species. In some cases those threats or the impacts from the threats are continuing in nature. The demographic factors included abundance, productivity, spatial distribution, and diversity. Because sea turtles spend the majority of their lives at sea, where they are spread across vast distances, it is difficult to estimate total abundance. However, the number of nesting females can be counted directly, or estimated indirectly by counting the number of nests on beaches, during a nesting season. Females nest more than once in a season (*i.e.*, clutch frequency, which is the average number of nests per season) and do not nest every season (*i.e.*, remigration interval,

which is the average number of years between successive nesting seasons). To calculate the index of nesting female abundance at a nesting beach, the Team summed the total number of nests over the most recent remigration interval (*i.e.*, a run-sum) and divided this number by the clutch frequency. The Team performed these calculations only if available data were recent (*i.e.*, last year of the remigration interval occurred in 2014 or more recently), consistent (*i.e.*, seasonal data collected for each year of the remigration interval), and collected in a standardized manner (*i.e.*, data collection methods remained the same over the remigration interval), as further detailed in the Status Review Report. To provide a total index of nesting female abundance for each DPS, we summed the indices of nesting female abundance for all monitored beaches used by that DPS. The total index of nesting female abundance for each DPS is an index (rather than a census) because not all nesting beaches met these criteria. However, the nesting beaches that were not included were generally unmonitored or not recently monitored because they host few nesting females. Even where data were not sufficient to allow for a calculation of the index of nesting female abundance, the Team provided all available data to ensure the analysis would be as robust as possible.

The Team evaluated the productivity for each DPS by evaluating nesting trends (through trend analyses or bar graphs) and productivity metrics. Where available data allowed it, they estimated the long-term trend for individual beaches using a Bayesian state-space model of stochastic exponential population growth (Boyd *et al.* 2017), where the rate parameter describes the annual percent change in observed nest counts (or female counts where applicable) over the period of data collection. This is further explained in the Status Review Report. To reflect current trends over approximately three remigration

intervals, the criteria for trend analyses were as follows: nesting data (*i.e.*, nest or nesting female counts) consistently collected over nine or more years in a standardized manner (for that site), with the most recent data collection in 2014 or later and with a minimum average number of nests of 50 annually. The Team reported the median trend, along with the standard deviation (sd), 95 percent credible interval (CI), and an “f statistic” which is the proportion of the posterior distribution with the same sign as the median (*i.e.*, the confidence that the trend is positive or negative). When the data did not meet the criteria for performing trend analyses, the Team provided bar graphs and/or historical data in the Status Review Report. Based on the trend analysis (where possible) and the best available historical data, the Team characterized the nesting trend for each DPS as decreasing, stable, or increasing. The Team also evaluated the following productivity metrics (if available): average size of nesting female; nesting female survivorship; remigration interval; clutch size; clutch frequency; internesting interval; incubation period; hatching success (the proportion of eggs in a nest that produce live hatchlings); and sex ratio. Each of these metrics contributes to the growth rate, or reproductive potential, of the population.

For each DPS, the Team evaluated spatial distribution, which included the number and location of nesting beaches and foraging areas, as well as spatial structure (*i.e.*, whether the DPS exists as a single population or several subpopulations connected by metapopulation dynamics). The Team also evaluated diversity, which like spatial distribution, is a measure of resilience. In general, diverse populations with broad spatial distributions and metapopulation dynamics are more resilient to threats and environmental changes than less diverse populations with narrow distributions.

For each DPS, the Team next evaluated each of the ESA Section 4(a)(1) factors (or “threats”) as listed above (16 U.S.C. 1533(a)(1), 50 CFR 424.11(c)). For each threat, the Team used the best available information to describe the threat, identify which life stages are affected, and describe the impact to the DPS with as much specificity as the best available information allowed to link the threat to the demographic factor it affected. The best available data often allow only for qualitative assessment. For each DPS, the Team identified the primary threat(s) to its continued existence, as well as other threats. The Team considered the impact of each threat individually, with the primary threat(s) given the greatest weight, and all threats cumulatively, to determine the extinction risk. To assess confidence in the extinction risk determination, the Team identified any sources of uncertainty and the impact of uncertainty on the conclusions. They analyzed all threats assuming the DPS had lost ESA protections going forward because a DPS would not receive such protections if it was not listed under the ESA. For example, a DPS would not have benefits of section 9 take prohibitions or section 7 consultations on actions that may affect the DPS.

The Team performed an extinction risk assessment for each of the seven DPSs by evaluating the demographic factors and threats, as described above. Then, the Team voted, based on the best available data, on whether the extinction risk of each DPS was high, moderate, or low, following the definitions included in NMFS’ internal guidance document, “Guidance on Responding to Petitions and Conducting Status Reviews under the Endangered Species Act, Section II” (*i.e.*, NMFS’ Guidance; November 9, 2017) and in the Status Review Report.

After the Team completed its draft Status Review Report, the Services met to review and discuss that document and conservation efforts. The Services based our status determinations of the DPSs on the best scientific and commercial data available (as compiled and reflected in the Status Review Report) and after taking into account efforts by States and foreign nation, or any political subdivision thereof, to protect the species as mandated by the statute.

### **DPS Analysis**

The following is a summary of the DPS analysis conducted by the Team. For a detailed description of the Team's analyses of discreteness and significance, please see the Status Review Report. As a starting point, the Team considered seven leatherback populations that were previously identified as regional management units (RMUs) by Wallace *et al.* (2010) and recognized as subpopulations under the International Union for Conservation of Nature (IUCN) Red List (<https://www.iucnredlist.org/species/6494/43526147>). The Team found that seven leatherback populations met the discreteness and significance criteria per the DPS Policy and identified the following potential DPSs: Northwest (NW) Atlantic; Southwest (SW) Atlantic; Southeast (SE) Atlantic; SW Indian; Northeast (NE) Indian; West Pacific; and East Pacific.

#### *Discreteness*

The Team evaluated all populations for discreteness and determined that each showed marked separation from the others as a consequence of behavioral and physical factors. Behavioral factors, especially returning to waters off a turtle's natal beach to

breed, have prevented interbreeding, resulting in reproductive isolation, as indicated by genetic discontinuity.

Although some populations use the same foraging areas, tagging and telemetry studies also demonstrate the discreteness of the populations at nesting beaches. Physical factors, such as land masses, ocean currents, and other oceanographic features, have established and reinforced barriers to gene flow among the seven populations.

Genetic data provide the most compelling evidence for discreteness among the seven populations. The most recent and comprehensive global analysis of published and unpublished mitochondrial deoxyribonucleic acid (mtDNA) sequence data (*i.e.*, 28 haplotypes, which are unique sequences of mtDNA) evaluated samples collected from 21 nesting sites representing key regions from all ocean basins (Dutton *et al.* 2007; Dutton *et al.* 2013; Shanker *et al.* 2011; Dutton and Shanker 2015); analyzing the evolutionary relationship of these data revealed three distinct haplogroups (*i.e.*, similar haplotypes that cluster together, relative to other haplotypes) that are geographically segregated across the Atlantic, Indian, and Pacific Oceans (Dutton, unpublished data; NMFS and USFWS 2020). Early mtDNA analyses indicated strong genetic discontinuity, globally ( $F_{ST} = 0.415$ ,  $P < 0.001$ ) and within ocean basins ( $F_{ST} = 0.203$  to  $0.253$ ,  $P < 0.001$ ; Dutton *et al.* 1999). Wallace *et al.* (2010) combined these and other genetic data with nesting, flipper tagging, and satellite telemetry data to identify seven leatherback RMUs, which provided the starting point for our identification of discrete populations.

From this starting point, the Team then evaluated more recent genetic data. Subsequent genetic analyses confirmed genetic discontinuity among the NW, SW, and SE Atlantic populations (Wallace *et al.* 2010; Dutton *et al.* 2013; Carreras *et al.* 2013;

Molfetti *et al.* 2013; Vargas *et al.* 2017). Elevated genetic differentiation at nuclear DNA ( $F_{ST} = 0.211 - 0.86$ ) indicates that males, like females, likely return to the waters off their natal beaches to mate and that male-mediated gene flow may not be as pronounced as previously thought (Dutton *et al.* 2013; see Jensen *et al.* 2013). Nuclear ( $F_{ST} > 0.126$ ,  $P < 0.001$ ; Dutton *et al.* 2013) and mtDNA ( $F_{ST} > 0.061$ ,  $P = 0.05 - 0.001$ ; Dutton *et al.* 2013;  $F_{ST} > 0.061$ ,  $P < 0.01$ ; Vargas *et al.* 2017) analyses indicate genetic discontinuity between the Atlantic populations and the SW Indian population. Preliminary mtDNA results for leatherback turtles nesting at Little Andaman Island, India (Shanker *et al.* 2011; Dutton and Shanker 2015), indicate that this population is closely related to the extinct Malaysian population, with which it shares common haplotypes. It is markedly different from the South African nesting population, as well as those in the West Pacific population (Dutton *et al.* 2007, 2013 and unpublished). Samples from extant and extirpated nesting aggregations of the NE Indian population (Shanker *et al.* 2011; Dutton and Shanker 2015; Dutton *et al.* unpublished data) are genetically differentiated from the SW Indian population ( $F_{ST} = 0.415$ ,  $P < 0.003$ ; Dutton *et al.* 1999) and the West Pacific population ( $X_2 = 49.346$ ,  $P = 0.002$ ; Dutton *et al.* 2007). There is genetic discontinuity between the West and East Pacific populations, as demonstrated by significant genetic differentiation between the samples from Solomon Islands in the western Pacific and Mexico or Costa Rica in the eastern Pacific ( $F_{ST} = 0.270$  and  $0.331$ ,  $P < 0.001$ ; Dutton *et al.* 1999). Genetic discontinuity among all seven populations provides evidence for marked separation from the others and thus discreteness of each population.

Tagging and telemetry studies confirm marked separation of the seven populations because nesting sites remain distant and isolated. Nesting females of one

population have not been tracked to, or observed on, beaches used by another population, even though telemetry data indicate shared use of foraging areas by different populations.

Telemetry studies demonstrate that females nesting on NW Atlantic beaches move throughout most of the North Atlantic from the Equator to about 50° N latitude (Ferraroli *et al.* 2004; Hays *et al.* 2004; James *et al.* 2005a; James *et al.* 2005b; 2005c; Eckert 2006a; Eckert *et al.* 2006b; Hays *et al.* 2006; Doyle *et al.* 2008; Evans 2008; Dodge *et al.* 2014; Fossette *et al.* 2014; Aleksa 2017; Aleksa *et al.* 2018). Turtles originating from beaches of the NW Atlantic appear to mix at foraging areas throughout the North Atlantic Ocean (Fossette *et al.* 2014), but their movements rarely extend into waters south of the Equator. Tagging studies further support the connectivity within and among nesting beaches and foraging areas of the North Atlantic Ocean (Troëng *et al.* 2004; Bräutigam and Eckert 2006; Chacón-Chaverri and Eckert 2007; Turtle Expert Working Group (TEWG) 2007; Sönmez *et al.* 2008; Dutton *et al.* 2013b; Horrocks *et al.* 2016), but turtles tagged in the North Atlantic Ocean have never been found on nesting beaches in Brazil (SW Atlantic population) or Africa (SE Atlantic population). In the South Atlantic Ocean, post-nesting females tracked from nesting beaches in Gabon and Brazil use the same foraging areas, including waters off SW Africa, in the south equatorial Atlantic and off SE Brazil and Uruguay (Almeida *et al.* 2011; Witt *et al.* 2011). Turtles incidentally captured in fisheries off South America (Billes *et al.* 2006, López-Mendilaharsu *et al.* 2009) also demonstrate that turtles originating from the SW and SE Atlantic Ocean beaches share foraging areas. Despite such mixing at foraging areas, there is no evidence for the shared use of nesting beaches. Genetic data indicate

that turtles return to their natal beaches to nest on opposite sides of the Atlantic Ocean (Dutton *et al.* 2013; Vargas *et al.* 2017), and no tag recoveries contradict these data.

In the Indian Ocean, telemetry studies have been conducted at South African nesting beaches in the SW Indian Ocean (Hughes *et al.* 1998; Luschi *et al.* 2006; Robinson *et al.* 2016) and at Andaman Islands nesting beaches in the NE Indian Ocean (Namboothri *et al.* 2012; Swaminathan *et al.* 2019). South African nesting females showed diverse movements that were highly influenced by complex oceanographic currents and features that lead them to foraging destinations in the South Atlantic Ocean, SW Indian Ocean, and Mozambique Channel (Hughes *et al.* 1998, Luschi *et al.* 2006, Robinson *et al.* 2016). About half of the 10 post-nesting females tagged at the Andaman Islands moved westward: two individuals reached the Mozambique Channel; the other half moved southeastward, past the Indonesian islands of Sumatra and Java, with one leatherback reaching an apparent foraging ground off NW Australia before transmissions stopped (Namboothri *et al.* 2012; Swaminathan *et al.* 2019). Despite overlap in one foraging area (*i.e.*, reaching the Mozambique Channel), tagging data do not indicate movement between the distant nesting beaches.

Within the Pacific Ocean, nearly all turtles tracked from East Pacific nesting beaches moved southward across the Equator to forage in open-ocean waters of the SE Pacific Ocean or in the coastal waters of Central America, Peru, and Chile. The movements of post-nesting females from the West Pacific Ocean are dependent on the season in which they nest, with winter-nesting females predominantly tracked into the Southern Hemisphere and summer-nesting females foraging in diverse coastal and oceanic ecosystems throughout the northern Indo-Pacific region (Benson *et al.* 2011).

Telemetry data indicate little or no overlap with foraging destinations utilized by nesting females of the East and West Pacific populations (Bailey *et al.* 2012; Benson *et al.* 2011). However, a genetic study of bycaught turtles off the coast of Chile and Peru indicated that 15 percent of leatherback turtles originated from West Pacific nesting beaches (Donoso and Dutton 2010), suggesting that foraging overlap may be more prevalent than estimated by telemetry data. Still, there is no genetic evidence for contemporary interbreeding between the two populations (Dutton *et al.* 2007), and telemetry and tagging data do not indicate movement between the distant nesting beaches. Thus, flipper tagging and satellite telemetry data support the marked separation, and thus discreteness, of the seven populations at their nesting beaches.

Physical factors likely shape and reinforce the behavior patterns that result in reproductive isolation. Though the species has a global range, with foraging areas extending into high latitudes, nesting mainly occurs on tropical or subtropical beaches. Post-hatchling dispersal is determined by the ocean currents they encounter off nesting beaches. While adults move throughout tropical and temperate waters irrespective of ocean currents, both males and females return to the waters off their natal nesting beach to mate. This natal homing is somewhat flexible, (Dutton *et al.* 2013; Jensen *et al.* 2013), creating reproductive isolation only among distant nesting sites, which may also be physically separated from one another by land masses and oceanographic barriers to gene flow. For example, leatherback turtles in the Atlantic Ocean are physically separated from those in the Pacific Ocean by the Americas. Though leatherback turtles have greater cold tolerance than other sea turtles, they do not appear to venture into latitudes greater than 47° S or 71° N (Eggleston 1971; Eckert *et al.* 2012). Therefore, the low latitude and

cold waters of the Cape Horn Current likely prevent movement between the Atlantic and Pacific Oceans. Within ocean basins, nesting beaches of the discrete populations are separated by long distances of uninterrupted deep water (*e.g.*, the East Pacific Barrier and the mid-Atlantic Barrier). While leatherback turtles clearly cross these open-ocean barriers to reach distant foraging areas, they do not appear to do so for nesting and breeding, but rather return to their natal region to breed and nest (Barragan *et al.* 1998; Dutton *et al.* 1999; Barragan and Dutton 2000; Dutton *et al.* 2013). Within ocean basins, currents shape post-hatchlings' movement patterns, which they may retain as adults (*e.g.*, Fossette *et al.* 2010; Benson *et al.* 2011). The NW Atlantic leatherback population appears to be physically separated from the SE and SW Atlantic populations by the current systems of the South and North Atlantic Gyres, respectively. NW Atlantic leatherback nesting beaches are adjacent to northward moving currents (*e.g.*, Gulf Stream). Leatherback hatchlings from these nesting beaches, therefore, are transported northward, remaining in the North Atlantic Ocean. Those that survive return to their nesting beaches as adults, completing their life stages within the North Atlantic (Fossette *et al.* 2010; Chambault *et al.* 2017). The SE and SW Atlantic populations are similarly retained in the South Atlantic Ocean by the South Atlantic Gyre and the Benguela Current, which flows northward along the SE coast of Africa, restricting movement into the Indian Ocean. Within the Indian Ocean, the Somali Current runs between the nesting beaches of the SW and NE Indian populations. The NE Indian and West Pacific populations likely became isolated as a result of exposed land barriers between Indonesia, New Guinea, and the Philippines as a result of low sea levels within the past 6,000 years (Barber *et al.* 2000). Seasonal monsoons may also play a contemporary role by altering

current directions and hatchling dispersal patterns (Benson *et al.* 2011; Gaspar *et al.* 2012). Thus, physical factors have likely helped to shape, or at least reinforce, the reproductive isolation among distant nesting beaches.

Based on these data, the Team concluded that the seven populations demonstrate discreteness, or marked separation from each other, due to behavioral and physical factors. These are the NW Atlantic, SW Atlantic, SE Atlantic, SW Indian, NE Indian, West Pacific, and East Pacific populations.

### *Significance*

Each of the discrete populations is significant to the species because the loss of any one would result in a significant gap (*i.e.*, a half or quarter of an ocean basin) in the range of the species. Several populations also persist in unique ecological settings. Each population likely possesses unique genetic characteristics and local adaptations as a result of thousands of years of reproductive isolation, but none have yet been identified because all genetic studies have involved neutral markers. Therefore, the Team did not rely on evidence of unique genetic characteristics and local adaptations for its significance finding.

A loss of the NW Atlantic population would result in a gap (*i.e.*, the entire North Atlantic Ocean) of the nesting and foraging range of the species. If the NW Atlantic population were extirpated, it is unlikely that leatherback turtles from other populations would recolonize the North Atlantic Ocean in an ecological time frame (*i.e.*, tens to hundreds of years), leaving a significant gap in the range of the species. Extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by the possession of several unique haplotypes. Leatherback turtles of the NW

Atlantic Ocean also occur in a unique ecological setting; this is the only DPS that regularly forages at high latitudes. Sightings have been documented as far north as Norway and Iceland (Brongersma 1972; Goff and Lien 1988; Carriol and Vader 2002; McMahon and Hayes 2006; Eckert *et al.* 2012). Such high latitude foraging is likely facilitated by the warm Gulf Stream, which meets cold water currents to create highly productive foraging areas. The Team concluded that the NW Atlantic population is biologically significant to the species.

In the SW Atlantic Ocean, leatherback turtles only nest in a small area of the coastline of Brazil. All other nesting in South America occurs above the Equator or on the Pacific Coast. Therefore, the loss of this population would result in a gap of the nesting range of the species (*i.e.*, the SW Atlantic coast). Although SE Atlantic leatherback turtles forage off the coasts of Brazil, Argentina, and Uruguay, they do not breed there. Rather, they return to the waters off western Africa to mate (Vargas *et al.* 2017). Therefore, if the SW Atlantic population were extirpated, it is unlikely that leatherback turtles from other populations would recolonize this region, leaving a significant gap in the nesting range of the species. The extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by the possession of unique haplotypes and high genetic diversity, despite the small population size (Vargas *et al.* 2017). The SW Atlantic population is biologically significant to the species.

Leatherback turtles of the SE Atlantic population nest in West Africa and forage in the South Atlantic Ocean. This population is much more abundant than the SW Atlantic population, which also forages in the South Atlantic Ocean. Therefore, the loss

of this population would result in a gap of the nesting range of the species (*i.e.*, western Africa) and a significant reduction in the abundance of leatherback turtles foraging throughout the South Atlantic Ocean. The extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by the possession of unique haplotypes. The Team concluded that the SE Atlantic population is biologically significant to the species.

In the SW Indian Ocean, leatherback turtles only nest in a small area along the South African and Mozambican coastlines. No other leatherback turtles nest in eastern Africa or in other areas throughout the entire western Indian Ocean. Therefore, the loss of this population would result in a gap of the nesting range of the species (*i.e.*, the SW Indian Ocean). The SW Indian population also occurs in a unique ecological setting: it is the only population to nest on temperate beaches. The warm Agulhas Current, adjacent to the nesting beaches, likely facilitates their high-latitude nesting. The Team concluded that the SW Indian population is biologically significant to the species.

Leatherback turtles nest in small numbers in the NE Indian Ocean. These nesting sites are separated from other Indian Ocean nesting sites by at least 5,000 km. Although western Pacific nesting sites are closer, males and females return to the waters off their natal beaches to breed, preventing interbreeding among NE Indian and West Pacific populations. Therefore, the loss of this population would result in a gap of the nesting range of the species (*i.e.*, the NE Indian Ocean). The extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by the possession of unique haplotypes. The Team concluded that the NE Indian population is biologically significant to the species.

West Pacific leatherback turtles nest in small numbers primarily in four nations of the West Pacific Ocean. These nesting sites are separated from East Pacific nesting sites by over 10,000 km. Though NE Indian nesting sites are closer in distance, male and female philopatry prevents interbreeding. Therefore, the loss of this population would result in a gap of the nesting range of the species (*i.e.*, the West Pacific Ocean). The loss of this population would also result in a gap of the foraging range of the species (*i.e.*, the North Pacific Ocean). The extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by the possession of unique haplotypes. The West Pacific population is ecologically unique in two ways: it is the only population to forage in both hemispheres; and it nests year-round, with nesting peaks in the summer and winter. The Team concluded that the West Pacific population is biologically significant to the species.

Leatherback turtles nesting on eastern Pacific coastlines also forage in the East Pacific Ocean. A loss of this population would result in a gap of the nesting range of the species (*i.e.*, the East Pacific Ocean). Though West Pacific leatherback turtles may forage off the coasts of Peru and Chile, they do not breed there (Donoso and Dutton 2010). Therefore, if the East Pacific population were extirpated, it is unlikely that leatherback turtles from other populations would recolonize this region, leaving a significant gap in the nesting range of the species. The extirpation of this population would also significantly reduce the genetic diversity of the species, as the population possess several unique haplotypes. The East Pacific population is unique in having the smallest nesting female size, clutch size, and egg size of all populations, possibly reflecting unique foraging conditions that are subject to oceanographic regime shifts (*e.g.*, the El Niño

Southern Oscillation, or ENSO). The Team concluded that the East Pacific population is biologically significant to the species.

### *DPS Summary*

The Team found that seven populations met the definition for discreteness. These populations are markedly separated as a result of the behavioral factors of movement (as demonstrated by satellite telemetry and flipper tagging studies) and philopatry, which has led to reproductive isolation (as demonstrated by genetic discontinuity). They are also physically separated by land masses, oceanographic features, and currents. The Team found these seven populations to be significant to the species because the loss of any one of them would result in a significant gap in the range of the species as well as a significant loss of genetic diversity, reducing the evolutionary potential of the species. Some populations also occur in a unique ecological setting. Thus, after reviewing the best available information, the Team identified the following populations as potential DPSs: NW Atlantic, SW Atlantic, SE Atlantic, SW Indian, NE Indian, West Pacific, and East Pacific. The Team defined the potential DPSs as leatherback turtles originating from nesting beaches within the boundaries for each DPS. The range of each DPS, which also includes foraging areas, thus extends beyond the nesting boundaries for most DPSs, and may overlap extensively with the range of another DPS. The boundaries are based on the best available genetic, telemetry, and observational data. When such data were not available, the Team used information on possible barriers to gene flow, such as oceanographic features. For ease of use, the Team applied political boundaries when this did not conflict with biological or oceanographic data. Additional information on the

boundaries is available in the following sections, which summarize the extinction risk analysis for each DPS, and in the Status Review Report.

### **NW Atlantic DPS**

The Team defined the NW Atlantic DPS as leatherback turtles originating from the NW Atlantic Ocean, south of 71° N, east of the Americas, and west of Europe and northern Africa; the southern boundary is a diagonal line between 5.377° S, 35.321° W and 16.063° N, 16.51° W. The northern boundary reflects a straight latitudinal line based on the northernmost documented occurrence of leatherback turtles (Brongersma 1972; Goff and Lien 1988; Carriol and Vader 2002; McMahon and Hayes 2006; Eckert *et al.* 2012). The southern boundary is a diagonal line between the elbow of Brazil, where the Brazilian current begins and likely restricts the nesting range of this DPS, and the northern boundary of Senegal. The boundary between Senegal and Mauritania was chosen because the SE Atlantic DPS does not appear to nest above this boundary (Fretey *et al.* 2007).

The range of this DPS (*i.e.*, all areas of occurrence) extends throughout the North Atlantic Ocean, including the Caribbean Sea, Gulf of Mexico (GOM), and Mediterranean Sea. Available data indicate that the NW Atlantic DPS occurs (at varying levels of frequency) in the waters of the following nations or territories: Albania, Algeria, Anguilla, Antigua and Barbuda, Aruba, Azores, Bahamas, Barbados, Belize, Bermuda, Bonaire, Bosnia and Herzegovina, Brazil, British Virgin Islands, Canada, Cape Verde, Cayman Islands, Colombia, Costa Rica, Croatia, Cuba, Curaçao, Cyprus, Denmark, Dominica, Dominican Republic, Egypt, France, French Guiana, Greece, Greenland, Grenada, Guadeloupe, Guatemala, Guyana, Haiti, Honduras, Iceland, Ireland, Israel,

Italy, Jamaica, Lebanon, Libya, Madeira, Malta, Martinique, Mauritania, Mexico, Montenegro, Montserrat, Morocco, Netherlands Antilles, Nicaragua, Norway, Panama, Portugal, Slovenia, Spain, St. Barthelemy, St. Eustatius, St. Kitts and Nevis, St. Lucia, St. Maarten, St. Pierre and Miquelon, St. Martin, St. Vincent and the Grenadines, Suriname, Sweden, Syria, Trinidad and Tobago, Tunisia, Turkey, Turks and Caicos Islands, United Kingdom, United States (including Puerto Rico and the U.S. Virgin Islands (USVI)), Venezuela, and Western Sahara.

All nesting in this DPS occurs in the NW Atlantic Ocean, concentrated from the southeast United States throughout the Wider Caribbean Region (Dow *et al.* 2007). Leatherback nesting in the NW Atlantic can be grouped into several broad geographical areas, including the U.S. mainland (primarily Florida), North Caribbean (including USVI and Puerto Rico), West Caribbean (Honduras to Colombia), and Southern Caribbean/Guianas (Venezuela to French Guiana; TEWG 2007). The largest nesting aggregations occur in Trinidad, French Guiana, and Panama. The northern-most confirmed nesting occurs in North Carolina, but there has been a crawl recorded as far north as Assateague Island National Seashore, Maryland (Rabon *et al.* 2003). No nesting occurs in the Mediterranean Sea (Casale and Margaritoulis 2010).

Nesting occurs on unobstructed, high-energy beaches with either a deep water oceanic approach or a shallow water approach with mud banks, but without coral or rock formations (TEWG 2007). The main characteristics of leatherback nesting beaches include coarse-grained sand; steep, sloping littoral zone; obstacle-free approach; proximity to deep water; and oceanic currents along the coast (Hendrickson and Balasingam 1966 in Eckert *et al.* 2015). During the nesting season, adult females and

males inhabit the waters off nesting beaches. During a nesting season, females generally stay within about 100 km of their nesting beaches, remaining close to the coast on the continental shelf, and engaging in shallow dives (Eckert *et al.* 2012). Intra-seasonal movement of greater than 100 km also occurs, especially between French Guiana and Suriname (Fossette *et al.* 2007; Georges *et al.* 2007), Panama and Costa Rica (Chacón-Chaverri and Eckert 2007), and among Caribbean nesting beaches, including those on Trinidad (Brautigam and Eckert 2006; Georges *et al.* 2007; Horrocks *et al.* 2016). Adult males migrate from temperate foraging areas in the North Atlantic Ocean to waters off nesting beaches, typically arriving before the nesting season and remaining for the majority of the season (James *et al.* 2005b; Doyle *et al.* 2008; Dodge *et al.* 2014).

Foraging areas of the NW Atlantic DPS include coastal and pelagic waters of the North Atlantic Ocean (Eckert *et al.* 2012; Saba 2013; Shillinger and Bailey 2015). These waters include the GOM, North Central Atlantic Ocean, northwestern Atlantic shelf waters of the United States and Canada, waters along the southeastern U.S. coast, the Mediterranean Sea, and the northeastern Atlantic shelf waters of Europe and northwestern Africa (TEWG 2007). Some post-nesting females also remain in tropical waters to forage (Fossette *et al.* 2010). This DPS is mostly commonly associated with open-ocean and coastal shelf foraging areas off Nova Scotia (Canada), northeastern United States, GOM, northwestern Europe, and northwestern Africa (James *et al.* 2005a, 2006b, 2007; Eckert 2006; Eckert *et al.* 2006; Fossette *et al.* 2010a; Fossette *et al.* 2010b; Dodge *et al.* 2014; Stewart *et al.* 2016; Aleksa *et al.* 2018). Fossette *et al.* (2014) analyzed available satellite telemetry data from 1995 to 2010 on post-nesting females (n = 93) as well as males (n = 4), females (n = 8), and a juvenile (n = 1) from foraging

grounds throughout the Atlantic Ocean. They found widespread use of the North Atlantic Ocean (Fossette *et al.* 2014). High-use areas mainly occurred in the central (25 to 50° N, 50 to 30° W) and eastern Atlantic Ocean, in particular in the waters offshore Western Europe, around Cape Verde (year-round) and the Azores (October to March; Fossette *et al.* 2014). Fossette *et al.* (2014) found that seasonal high-use areas also occurred along the eastern U.S. coast (April to June and October to December) and off Canada (July to December). The GOM is also a high-use foraging area, with a peak in the northeast GOM during August and September (Aleksa *et al.* 2018). Overall, leatherback turtles of the North Atlantic population appear to have a diverse array of foraging habitat available.

### *Abundance*

The total index of nesting female abundance for the NW Atlantic DPS is 20,659 females. The nesting beaches with the greatest abundance have been included in this index, and most beaches with an unquantified number of nests likely host few nesting females. We based this index on 24 nesting aggregations in 10 nations: Trinidad and Tobago (n = 11,324), French Guiana (n = 2,519), Panama (n = 2,251), United States (n = 1,694), Costa Rica (n = 1,306), Suriname (n = 698), Grenada (n = 499), Venezuela (n = 215), Guyana (n = 76), and Nicaragua (n = 10). With the possible exception of Colombia, our total index does not include 31 unquantified but likely small nesting aggregations for which data are not available. It also does not include outdated data published by Dow *et al.* (2007), which includes binned crawls, categorized as less than 25, 25 to 100, 100 to 500, 500 to 1000, or unknown abundance. Crawls or emergences (measured as females or tracks on beaches) include both successful egg-laying and unsuccessful nesting, so the number of crawls represents approximately two to 10 times the number of nests (Dow *et*

*al.* 2007). Because the Dow *et al.* data, which are more than 10 years old and do not provide the number of actual nests, may not be representative of recent nesting trends, we did not include them in our total index. To calculate the indices of nesting female abundance, we added the number of nests over the last 3 years (representing the most recent remigration interval; Eckert *et al.* 2012) and divided by the clutch frequency (site-specific values or, when such values were not available, the average of the site-specific values, *i.e.*, 5.5 clutches per season).

Our total index of nesting female abundance is based on the best available data for this DPS. It is the most robust estimate of nesting females at this time because it only includes available nesting data from recently and consistently monitored nesting beaches. Our total index does not include data from beaches where we were unable to quantify the number of nesting females, either due to the lack of recent or available nesting data or because only crawl data were reported (often on smaller nesting beaches). Scattered nesting may occur on beaches throughout the region, but because these beaches are not monitored, or have not been recently monitored, recent data are not available.

Nesting in the NW Atlantic DPS is characterized by many small nesting beaches. Large nesting aggregations are rare; only about 10 leatherback nesting beaches in the Wider Caribbean Region (about two percent of the DPS's total nesting sites) host more than 1,000 crawls annually (Dow Piniak and Eckert 2011). Only one site, Grande Riviere in Trinidad, hosts more than 5,000 nesting females, representing 29 percent of the total index of nesting female abundance. Relatively large nesting aggregations are also found in Matura (Trinidad), Chiriqui Beach (Panama), and Cayenne and Remire Montjoly (French Guiana). In contrast, most known nesting beaches support a small nesting female

abundance; 71 percent of the total nesting sites record annual crawls of less than 100 (Dow Piniak and Eckert 2011). The number of nesting females is unquantified at 31 beaches (*i.e.*, the majority of nesting sites for the DPS). However, for the reasons identified above, most of those sites have small abundance levels as inferred from the numbers of crawls estimated by Dow *et al.* (2007). Therefore, our total index of nesting female abundance represents the most robust estimate allowed by the best available data and includes the majority of nesting females because the largest nesting aggregations were included. The data regarding additional nesting aggregations are not sufficiently recent, specific, or reliable for inclusion, and the contribution of these nesting aggregations to the total index is expected to be small.

Our total index of nesting female abundance is similar in comparison to other published estimates. TEWG (2007) estimated the abundance of NW Atlantic leatherback turtles using nesting data from 2004 and 2005. At that time, the number of adult females (equating to total index of nesting female abundance in our analysis) was estimated to be approximately 18,700 (range 10,000 to 31,000). While a wide range was provided, the point estimate in TEWG (2007) is similar to, albeit slightly lower than, our total index of 20,659 nesting females. The most recent, published IUCN Red List assessment for the NW Atlantic Ocean subpopulation estimated a total of 20,000 mature individuals (The NW Atlantic Working Group 2019). Our total index, which only includes nesting females, exceeds their estimate, likely due to our use of a 3-year remigration interval, which has increased at some locations in recent years (*e.g.*, 4.5 years at St. Croix; K. R. Stewart, The Ocean Foundation and C. Lombard, USFWS, pers. comm., 2019).

We conclude that the total index of nesting females for the NW Atlantic DPS is 20,659 females. The nesting beaches with the greatest abundance have been included in our total index, and most beaches with an unquantified number of nests likely host few nesting females. Current nesting female abundance is not at a level where stochastic or environmental changes would have catastrophic impacts, but the abundance at several nesting sites with previously high density has declined drastically. However, as we discuss below, a declining nest trend and several existing threats will likely continue to reduce this abundance.

### *Productivity*

The NW Atlantic DPS exhibits decreasing nest trends at nesting aggregations with the greatest indices of nesting female abundance. Though some nesting aggregations indicate increasing trends, most of the largest ones demonstrate declining nest trends. We evaluated nest trends by using nest count data consistently collected using a standardized approach for at least 9 years, with the last year of data in 2014 or more recently and with an average of more than 50 nests annually. When data did not meet these criteria, we evaluated bar graphs provided in the Status Review Report to consider all available data. Thus, these data are representative of the DPS because they include the largest nesting aggregations. With the possible exception of Colombia, nesting aggregations for which data are not available are likely small. Significant declines have been observed at nesting beaches with the greatest historical or current nesting female abundance, most notably in Trinidad and Tobago (Grande Riviere, Fishing Pond, and Tobago), Suriname, French Guiana (Awala-Yalimapo), Florida, and Costa Rica (Tortuguero). Therefore, these nest trends represent the best available data for this DPS.

In Trinidad and Tobago, trends in annual nest counts were largely negative between 2009 and 2017, the years for which data were available. For Trinidad, we analyzed trends for three separately monitored beaches, including Grande Riviere, Matura, and Fishing Pond. The long-term trend was negative for Grande Riviere (median = -6.9 percent; sd = 17.4 percent; 95 percent CI = -43.8 to 26.9 percent;  $f = 0.682$ ; mean annual nests = 13,272), positive for Matura (median = 1.8 percent; sd = 15.1 percent; 95 percent CI = -29.2 to 33.0 percent;  $f = 0.561$ ; mean annual nests = 7,359), and negative for Fishing Pond (median = -19.3 percent; sd = 15.1 percent; 95 percent CI = -49.8 to 12.0 percent;  $f = 0.916$ ; mean annual nests = 3,892). For Tobago, the median trend was -0.9 percent annually (sd = 11.3 percent; 95 percent CI = -25.0 to 21.5 percent;  $f = 0.540$ ; mean annual nests = 452).

For French Guiana, we analyzed nest count data from 2002 to 2017 for Awala-Yalimapo beach in the west and data from 1999 to 2017 for Cayenne and Remire Montjoly beaches in the east. There was a steep decline at Awala-Yalimapo, with a median trend of -19.4 percent annually (sd = 12.2 percent; 95 percent CI = -43.2 to 6.0 percent;  $f = 0.942$ ; mean annual nests = 3,200). In contrast to Awala-Yalimapo, nest counts at Cayenne and Remire Montjoly increased by 2.8 percent annually (sd = 12.9 percent; 95 percent CI = -24.9 to 27.9 percent;  $f = 0.596$ ; mean annual nests = 3,498). In addition, leatherback nesting occurred on remote beaches in western French Guiana until 2013 (*e.g.*, a high of 4670 nests was found in 2003, with 1,270 mean annual nests from 2002 to 2013), but we were unable to analyze trends because monitoring on these remote beaches has been reduced since approximately 2010 due to significant beach erosion and the disappearance of some previously monitored beaches.

Suriname, Grenada, and Panama each had a single time series sufficient for trend analysis. For Suriname, we combined datasets from two beaches, Galibi and Braamspunt, which were monitored between 2001 and 2017. Total nests in Suriname declined by  $-14.6$  percent annually (sd = 9.6 percent; 95 percent CI =  $-36.4$  to 4.5 percent;  $f = 0.953$ ; mean annual nests = 4,586). In Grenada, data on the number of nesting tracks were collected on Levera beach between 2002 and 2018. There was a 7.1 percent annual increase in tracks at Levera during that period (sd = 8.7 percent; 95 percent CI =  $-10.5$  to 25.3 percent;  $f = 0.827$ ; mean annual tracks = 895). In Panama, the nest counts at Chiriqui beach increased by 0.8 percent annually (sd = 7.0 percent; 95 percent CI =  $-14.1$  to 14.6 percent;  $f = 0.557$ ; mean annual nests = 4,463) between 2004 and 2017.

In Costa Rica, the four beaches for which we had sufficient data to analyze annual nest count trends mostly exhibited declining trends. Tortuguero experienced the steepest decrease, with a median trend of  $-10.9$  percent annually (sd = 4.2 percent; 95 percent CI =  $-19.5$  to 2.2 percent) for data collected between 1995 and 2017. Nest counts decreased by  $-3.8$  percent annually at Pacuare beach (sd = 9.3 percent; 95 percent CI =  $-22.6$  to 16.9 percent) between 2004 and 2017, but increased by 1.8 percent annually (sd = 6.0 percent; 95 percent CI =  $-10.8$  to 14.2 percent) at the nearby Pacuare Nature Reserve between 1991 and 2017. Nest counts at Estacion la Tortuga decreased slightly, with a median trend of  $-0.5$  percent annually (sd = 7.0 percent; 95 percent CI =  $-15.7$  to 13.1 percent) between 2002 and 2017.

For the United States, we analyzed annual nest count trends for Florida (statewide data collected between 2008 and 2017), three beaches in Puerto Rico, including Culebra (1984 to 2017), Luquillo-Fajardo (1996 to 2017), and Maunabo (1999 to 2017), and

Sandy Point National Wildlife Refuge in St. Croix, USVI (1982 to 2017). The median trend for Florida was a decline of  $-2.1$  percent annually (sd = 13.0 percent; 95 percent CI =  $-28.3$  to 25.5 percent;  $f = 0.582$ ; mean annual nests = 1,288). Culebra nests decreased by  $-3.7$  percent annually (sd = 5.3 percent; 95 percent CI =  $-14.9$  to 6.8 percent;  $f = 0.791$ ; mean annual nests = 153), while nests increased by 15.9 percent annually at Luquillo-Fajardo (sd = 5.5 percent; 95 percent CI =  $-7.1$  to 15.3 percent;  $f = 0.805$ ; mean annual nests = 283) and by 7.7 percent annually at Maunabo (sd = 4.9 percent; 95 percent CI =  $-2.7$  to 17.4 percent;  $f = 0.945$ ; mean annual nests = 161). In St. Croix, nests increased by 1.7 percent annually (sd = 4.6 percent; 95 percent CI =  $-7.8$  to 10.7 percent;  $f = 0.660$ ; mean annual nests = 399).

These trend data are similar to other recent findings, adding further confidence in declining trends at multiple large nesting aggregations. Because of concerns about declining nest counts throughout the region, the National Fish and Wildlife Foundation (NFWF) convened a NW Atlantic Leatherback Working Group (*i.e.*, the Working Group) to assess recent nesting data and complete a region-wide trend analysis (NW Atlantic Leatherback Working Group 2018). The trend analyses conducted by the Working Group used leatherback nesting data from 23 sites from 14 different nations with at least 10 years of data with consistent within-site methodology, analyzing data for three time periods: 1990 to 2017, 1998 to 2017, and 2008 to 2017. Our approach to trend analyses was similar to that used by the Working Group in that both approaches involved Bayesian analyses of data meeting set criteria. However, the Team decided against aggregating the data over the DPS due to incongruity of data collection methods, collection dates and

duration, and reporting. Despite these differences, the overall conclusion was the same—an overall declining nest trend.

The Working Group found that regional, abundance-weighted trends were negative for all three time periods and became more negative in the more recent time series (NW Atlantic Leatherback Working Group 2018). Specifically, overall nesting trends decreased at  $-4.21$  percent annually from 1990 to 2017 and at  $-5.37$  percent annually from 1998 to 2017, with the most notable decrease ( $-9.32$  percent annually) occurring during the most recent time frame of 2008 to 2017. While site-level trends showed variation within and among sites and across the time periods, overall the sites also reflected the same regional pattern: more negative trends were apparent during the most recent time frame. Seven sites had significant positive nesting trends from 1990 to 2017, but no sites exhibited significant positive trends from 2008 to 2017. The significant decline observed at Awala-Yalimapo, French Guiana ( $-12.95$  percent annually from 1990 to 2017,  $-19.05$  percent annually from 1998 to 2017, and  $-31.26$  percent annually from 2008 to 2017), drove the regional results, but similar significant declines were found at other nesting beaches for the longer time period, including: St Kitts and Nevis ( $-12.43$  percent annually), Tortuguero, Costa Rica ( $-10.42$  percent annually), Suriname ( $-5.14$  percent annually), and Culebra, Puerto Rico ( $-4.61$  percent annually). It should be noted that the other nesting beach in French Guiana (Cayenne) demonstrated an increasing trend ( $7.44$  percent annually from 1990 to 2017 and  $8.19$  percent annually from 1998 to 2017). However, it exhibited a decreasing trend ( $-14.21$  percent annually) from 2008 to 2017. While nesting increased over time at Cayenne, this increase has apparently not resulted from females shifting from Awala-Yalimapo, as turtles that nest at Cayenne are

genetically distinct (Molfetti *et al.* 2013) and females tagged in Awala-Yalimapo are not seen in Cayenne or vice versa (NW Atlantic Leatherback Working Group 2018).

These modeling results demonstrate that there has been a decline in NW Atlantic nesting from 1990 to 2017, with the most significant decreases occurring from 2008 to 2017. Some nesting beaches demonstrated positive trends for the longer time period. However, none showed significant increases over the most recent time period. The cause for the decline is uncertain, but the Working Group identified anthropogenic sources (*e.g.*, fisheries bycatch), habitat losses, and changes in life history parameters (such as remigration interval) as potential drivers of the regional decline. While these results were taken into consideration by the Team when evaluating the extinction risk of the NW Atlantic DPS, the Team also performed its own trend analysis of the data provided to the Team so that the trends were calculated in a manner consistent with other DPSs. Regardless, both trend analyses conclude that the NW Atlantic DPS is experiencing a significant decline in nesting.

In-water abundance studies of leatherback turtles are rare. Archibald and James (2016) assessed the relative abundance of turtles at a foraging area off Nova Scotia, Canada, from 2002 to 2015. This study evaluated opportunistic sightings per unit effort and found a mean density of 9.8 turtles per 100 km<sup>2</sup>, representing the highest in-water density of leatherback turtles reported to date. Archibald and James (2016) concluded that the relative abundance of foraging leatherback turtles off Canada exhibited high inter-annual variability but, overall, showed a stable trend from 2002 to 2015. The authors reported that (at that time) these results were consistent with the stable or, in some cases, increasing trends reported for contributing NW Atlantic nesting beaches over

the last decade (Dutton *et al.* 2005; Girondot *et al.* 2007; Fossette *et al.* 2008; McGowan *et al.* 2008; Stewart *et al.* 2011; Rivas *et al.* 2015). While there were no indications of a decreasing trend, the results should be interpreted with caution because of the small study area, opportunistic data collection, availability bias variance, and lack of understanding of the relative density outside the study area (Archibald and James 2016).

Despite the declining trend in nesting, productivity parameters for the DPS are similar to the species' averages (though some may be declining, as we discuss below). While there is some variation, most productivity parameters are relatively consistent throughout the DPS. The overall survival rate for nesting females is relatively high at 85 percent (Pfaller *et al.* 2018), with mean estimates of 0.70 to 0.99 in French Guiana (Rivalan *et al.* 2005, 2008), 0.89 in St. Croix (Dutton *et al.* 2005), and 0.89 to 0.96 on the Atlantic coast of Florida (Stewart *et al.* 2007, 2014). Remigration intervals range from 1 to 11 years (Schulz 1975; Boulon *et al.* 1996; Chevalier and Girondot 1998; Hilterman and Goverse 2007; Eckert *et al.* 2012; Stewart *et al.* 2014; Rivas *et al.* 2016; Garner *et al.* 2017). In St. Croix and St. Kitts, the median remigration interval appears to be increasing (4.5 years; K. R. Stewart, The Ocean Foundation and C. Lombard, USFWS, pers. 2019; K. M. Stewart, Ross University School of Veterinary Medicine and St. Kitts Sea Turtle Monitoring Network, pers. comm., 2019). Averaging all available data, the mean remigration interval for the DPS is 2.7 years, rounded to 3 years for use in our calculation of the index of nesting female abundance. Average clutch frequency per nesting season ranges from 3.6 to 8.3 throughout the region, with an overall mean of 5.5 nests per season, interspersed with 9 to 10 day internesting intervals (Eckert *et al.* 2015; Garner *et al.* 2017). Recent records indicate that nesting females deposit 80 to 88 eggs per clutch.

However, an early study by Carr and Ogren (1959) reported only 67 eggs per clutch. Hatching success is highly variable for nests that remain *in situ*, even for those that are viable and do not experience significant inundation or predation, with estimates as low as 8.9 percent in Costa Rica (Troëng *et al.* 2007) and 10.6 percent in Suriname (Hilterman and Goverse 2007) and as high as 93.4 percent in Florida (Perrault *et al.* 2012). Overall, hatching success is estimated at approximately 50 percent (Eckert *et al.* 2012). Hatchling sex ratios often exhibit a female bias, but less so than for other sea turtle species, with estimated production of anywhere from 30 to 100 percent females in Suriname, Tobago, Colombia, and Costa Rica (Mrosovsky *et al.* 1984; Dutton *et al.* 1985; Godfrey *et al.* 1996; Leslie *et al.* 1996; Mickelson and Downie 2010; Patiño-Martínez *et al.* 2012). However, the proportion of females documented in foraging individuals and strandings ranges from 57 to 70 percent (Murphy *et al.* 2006; James *et al.* 2007; TEWG 2007), and the ratio of females to males during an individual breeding season is thought to be closer to 1:1 (Stewart and Dutton 2014).

We conclude that the DPS exhibits a declining nest trend. In addition, there are indications of decreased productivity within the DPS. In St. Croix, one of the most thoroughly monitored nesting beaches in this DPS, the data from 1981 to 2010 indicate that hatching success and clutch frequency are declining and remigration intervals are increasing (Garner *et al.* 2017). Overall, we have a high degree of confidence in the decreasing nest trend and productivity metrics for this DPS, due to the large amount of data available from the largest nesting aggregations. We acknowledge that data are not available from all nesting beaches, but the data that we have relied upon is the best available and meets established standards. The declining trends reflect reduced nesting

female abundance. In addition, longer remigration intervals and/or reduced clutch frequencies may play a role in this decline. The decline reflects a reduction in productivity that places the DPS at risk given the magnitude and duration of the decreasing trend.

### *Spatial Distribution*

The DPS has a broad spatial distribution for both foraging and nesting. There is significant genetic population structure, with subpopulations connected via various levels of gene flow and metapopulation dynamics. Tagging and telemetry studies indicate considerable mixing of leatherback turtles among nesting beaches and at multiple foraging areas throughout the North Atlantic Ocean.

Nesting is widespread throughout the NW Atlantic beaches, occurring primarily as scattered, small aggregations throughout the Wider Caribbean, but with larger concentrations of nesting activity at certain sites in Trinidad, French Guiana, Suriname, Trinidad, Colombia, Panama, Costa Rica, Puerto Rico, St. Croix, and Florida (Horrocks *et al.* 2016).

Genetic sampling in the NW Atlantic DPS has been generally extensive with good coverage of large populations in this region. However, sampling from some smaller Caribbean nesting aggregations is absent, and there are gaps in sampling or analysis for nesting sites along the coasts of South and Central America (*e.g.*, Guyana, Venezuela, Colombia, and Panama). A comprehensive survey of genetic population structure in the Atlantic Ocean included large sample sizes from five nesting populations representative of the DPS and analysis of longer mtDNA sequences in combination with an array of 17 nuclear microsatellite DNA loci (Roden and Dutton 2011; Dutton *et al.* 2013). The

microsatellite data revealed fine-scale genetic differentiation among neighboring subpopulations (Dutton *et al.* 2013): Trinidad, French Guiana/Suriname, Florida, Costa Rica, and St. Croix. The mtDNA data failed to find significant differentiation between Florida and Costa Rica or between Trinidad and French Guiana/Suriname. However, Dutton *et al.* (2013) show that the mtDNA sequence variation had relatively low statistical power to detect fine scale structure compared to the microsatellite DNA loci. The mtDNA homogeneity between Costa Rica and Florida, with differentiation demonstrated at nuclear DNA loci, suggests that Costa Rica may be the source of founders for the Florida population via one or multiple recent colonization events, likely indicating historic connectivity rather than ongoing demographic connectivity (Dutton *et al.* 2013). Likewise the French Guiana/Suriname and Trinidad populations were undifferentiated with mtDNA likely indicating historic connectivity. However, microsatellite DNA reveal fine-scale genetic structure that is consistent with tagging studies demonstrating a lack of nesting female movement between the two nesting aggregations (TEWG 2007). Significant genetic differentiation has also been reported for Martinique and Guadeloupe and the mainland French Guiana rookery (Molfetti *et al.* 2013). St. Croix likely represents a broader Northern Caribbean subpopulation of the NW Atlantic population that includes multiple neighboring island nesting aggregations in the USVI and Puerto Rico. However, sampling and analysis would be required to determine extent of fine scale structuring (NMFS unpublished data; Dutton *et al.* 2013). The Costa Rica (Tortuguero and Gandoca) and Guiana (French Guiana and Suriname) nesting aggregations are distinct subpopulations based on microsatellite and mtDNA results (Dutton *et al.* 2013), but information on tag returns indicates movement of nesting

females between adjacent beaches of Panama, Colombia, Venezuela and Guyana.

Therefore, these nesting aggregations have “fuzzy” boundaries, likely a result of flexible natal homing. Nesting females use beaches up to 400 km apart between nesting seasons (Troëng *et al.* 2004; Chacón-Chaverri and Eckert 2007) and up to 463 km apart within the same nesting season (Stewart *et al.* 2014). Additional sampling of the remaining nesting sites will be required to determine the extent of fine-scale structuring within the NW Atlantic DPS. However, the available science indicates significant substructure within the DPS.

Tagging studies indicate individual movement and gene flow among nesting aggregations. This is facilitated by the species’ flexible natal homing, *i.e.*, philopatry to a region, rather than a specific beach. In adjacent nesting sites in French Guiana and Suriname, five to six percent of nesting females were observed to shift from one site to the other within a season (TEWG 2007), while Schulz (1971) reported this proportion to be slightly higher at 8.5 percent. In contrast, 35 percent of nesting females in Gandoca, Costa Rica, were estimated to nest at sites other than the study site during an individual season (Chacón-Chaverri and Eckert 2007). The predisposition of nesting females to stray within a nesting season may be influenced by the proximity of alternative nesting sites within a range of approximately 200 km (Horrocks *et al.* 2016). However, even within a given nesting season, females have been observed to move as far as 369 km (Grenada), 463.5 km (Florida), and 532 km (Dominica) from their original location (Horrocks *et al.* 2016). Among nesting seasons, interchange between nesting locations also appears to be frequent and wide-ranging, with maximum distance separating two

nesting sites for an individual female recorded as 1,849 km over an 8-year span (Horrocks *et al.* 2016).

Genetic studies have revealed that turtles from different nesting aggregations use the same foraging areas. Analyzing 684 longline bycatch samples from across the NW Atlantic in a mixed stock analysis and microsatellite assignment, Stewart *et al.* (2016) found that leatherback turtles from Costa Rica were caught in a higher proportion in the GOM (43 percent) compared to the Northeast Distant fishing zone, an area in the northwestern Atlantic Ocean (6 percent), while turtles from Trinidad and French Guiana comprised 54 percent of bycatch in the GOM and 93 percent in the Northeast Distant fishing zone. A study of turtles foraging off Nova Scotia, Canada, similarly assigned most (82 percent) of the 288 sampled turtles to Trinidad (n = 164) and French Guiana (n = 72), with 15 percent (n = 44) from Costa Rica, and the remainder from St. Croix (n = 7) and Florida (n = 1; Stewart *et al.* 2013). These proportions generally represent the relative population sizes for these breeding populations. Microsatellite DNA assignment of wild captured or stranded males (n = 122) throughout the NW Atlantic and Mediterranean found that all males originated from NW Atlantic nesting aggregations (Trinidad: 55 percent, French Guiana: 31 percent, and Costa Rica: 14 percent; Roden *et al.* 2017). No turtles were identified from St. Croix or Florida. One turtle that stranded in Turkey was assigned to French Guiana, while strandings in France were assigned to Trinidad or French Guiana (Roden *et al.* 2017).

The mixing of nesting aggregations at foraging areas is also supported by several tagging and/or satellite telemetry projects, conducted in U.S. waters (Murphy *et al.* 2006; LPRC 2014; Dodge *et al.* 2014, 2015; Aleksa *et al.* 2018), Canada (James *et al.* 2005a,

2005b, 2005c, 2006b, 2007; Bond and James 2017), Atlantic Europe and Mediterranean (Doyle *et al.* 2008; Sonmez *et al.* 2008), and on nesting beaches of various nations (Hildebrand 1987; Hays *et al.* 2004; Ferraroli *et al.* 2004; Eckert 2006; Eckert *et al.* 2006; Hays *et al.* 2006; TEWG 2007; Sonmez *et al.* 2008; Evans *et al.* 2008; Fossette *et al.* 2010a, 2010b; Richardson *et al.* 2012; Bailey *et al.* 2012; Stewart *et al.* 2014; Fossette *et al.* 2014; Horrocks *et al.* 2016; Chambault *et al.* 2017). For instance, turtles from Nova Scotian foraging grounds were tracked to nesting areas off Colombia, Trinidad, Guyana, and French Guiana (Bond and James 2017). The reverse has also been demonstrated: some leatherback turtles from the western Atlantic undertake annual migrations to Canadian waters to forage (James *et al.* 2005c), exemplified by post-nesting adults tracked to the waters off Nova Scotia from a variety of nesting locations, including French Guiana and Trinidad (Fossette *et al.* 2014), Costa Rica, Panama (Evans *et al.* 2008), and Anguilla (Richardson *et al.* 2012). The eastern and western GOM also provide foraging areas for this DPS (Aleksa *et al.* 2018), as observed from tracks of post-nesting turtles from Florida (Hildebrand 1987), Costa Rica (Tortuguero, Gandoca), and Panama (Chiriquí Beach; Evans *et al.* 2008; Evans *et al.* 2012). Evans *et al.* (2008) suggested that the GOM may represent a significant foraging ground for leatherback turtles from the Caribbean coast of Central America.

High use foraging areas may be identified through available telemetry data, but the migration routes to those areas may vary. Ferraroli *et al.* (2004) tracked leatherback turtles from French Guiana and found turtles dispersed widely throughout the North Atlantic but mostly followed two dispersion patterns: (1) moving north to the Gulf Stream area, where they started following the general ocean circulation; and (2) traveling

east, swimming mostly against the North Equatorial Current. Fossette *et al.* (2014) found a relatively broad migratory corridor when turtles departed their nesting sites in French Guiana/Suriname, and their movements overlapped with turtles from Grenada and Trinidad. Fossette *et al.* (2010a, 2010b) found that turtles tracked from nesting beaches in French Guiana, Suriname, and Grenada and turtles caught in waters off Nova Scotia and Ireland displayed three distinct migration strategies: (1) heading northwest to fertile foraging areas off the Gulf of Maine, Canada, and GOM; (2) crossing the North Atlantic Ocean to areas off western Europe and Africa; and (3) residing between northern and equatorial waters. Essentially, tagging data coupled with satellite telemetry data indicate that leatherback turtles of the NW Atlantic DPS use the entire North Atlantic Ocean for foraging and migration (TEWG 2007).

Although adults forage at multiple areas throughout the North Atlantic Ocean (Fossette *et al.* 2014), the range of juvenile leatherback turtles may be more restricted. Using an active movement model, Lalire and Gaspar (2019) found that most juveniles originating from nesting beaches in French Guiana and Suriname cross the Atlantic Ocean at mid-latitudes with north-south seasonal migrations; after several years, they reach the coasts of Europe and North Africa. Eckert (2002) reviewed the records of nearly 100 sightings of juvenile (less than 100 cm curved carapace length (CCL)) leatherback turtles and determined they are generally found in waters warmer than 26°C, suggesting that the first portion of their life is spent in tropical and subtropical waters. After exceeding 100 cm CCL, distribution extends into cooler waters (as low as 8°C), which is considered to be the primary habitat for the species (Eckert 2002).

The wide distribution of nesting and foraging areas likely buffers the DPS against local catastrophes or environmental changes. The fine-scale population structure, with movement of individuals and genes among nesting aggregations, indicates that the DPS has the capacity to withstand other catastrophic events.

### *Diversity*

The NW Atlantic DPS exhibits spatial diversity, as demonstrated by insular and continental nesting, multiple diverse foraging areas, and moderate genetic diversity. The DPS nests along both continental and insular coastlines. Nesting beach habitat also shows considerable diversity, ranging from coarse-grained, sandy beaches to silty, ephemeral shorelines whose dynamics are influenced by estuarine input. The breadth and, in some cases, transiency, of suitable nesting habitat in the western North Atlantic may contribute to consistent, low-level flexibility in natal homing, both within and among reproductive seasons (Bräutigam and Eckert 2006), and this flexibility is thought to surpass that of other sea turtle species (TEWG 2007).

This DPS exhibits some temporal variation in nesting. Nesting generally begins in March or April, peaks in May or June, and ends in July or August (Eckert *et al.* 2012). In French Guiana, a second small nesting peak was documented in Awala-Yalimapo during December and January. However, the number of nests deposited during that time frame decreased from 700 in 1986/1987 to 40 in 1992/1993, and now only a small number of individuals are observed to nest during that time (Girondot *et al.* 2007). Some evidence indicates that the timing of nesting may be modulated by environmental characteristics distant from the nesting beach, such as water temperatures at foraging grounds (Neeman *et al.* 2015).

The foraging strategies are also diverse, with turtles using coastal and pelagic waters throughout the entire North Atlantic Ocean (Fossette *et al.* 2014). Foraging habitats include temperate waters of the GOM, North Central Atlantic Ocean, northwestern shelf (United States and Canada), southeastern U.S. coast, the Mediterranean Sea, and northeastern shelf (Europe; TEWG 2007). Some post-nesting females also remain in tropical waters (Fossette *et al.* 2010). Overall, leatherback turtles in the North Atlantic Ocean appear to have a diverse array of foraging habitat available.

Genetic diversity of the DPS is moderate, with six mtDNA haplotypes (Dutton *et al.* 2013). In St. Croix, a unique haplotype occurs at high frequency. The Florida and Costa Rica nesting aggregations each possess one unique, low frequency haplotype.

Based upon this information, we conclude that nesting location and habitat are diverse, providing some level of resilience against short-term spatial and temporal changes in the environment. However, high-abundance nesting occurs only at a few locations (*e.g.*, Trinidad, French Guiana, and Panama). The foraging diversity likely provides resilience against local reductions in prey availability or catastrophic events, such as oil spills, by limiting exposure to a limited proportion of the total population. Moderate genetic diversity may provide the DPS with the raw material necessary for adapting to long-term environmental changes, such as cyclic or directional changes in ocean environments due to natural and human causes (McElhany *et al.* 2000; NMFS 2017). We conclude that such diversity provides some level of resilience to threats for this DPS.

*Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

Destruction and modification of leatherback turtle nesting habitat results from a variety of activities including coastal development and construction; beach erosion and inundation; placement of erosion control and nearshore shoreline stabilization structures and other barriers to nesting; beachfront lighting; vehicular and pedestrian traffic; beach sand placement; sand extraction; removal of native vegetation; and planting of non-native vegetation (Lutcavage *et al.* 1997; Bouchard *et al.* 1998; USFWS 1999; Dow *et al.* 2007; Eckert *et al.* 2012; NMFS and USFWS 2013). As a result, most nesting beaches are severely degraded by such activities that continue to cause adverse impacts throughout the range of the DPS.

#### Coastal Development and Construction

In many areas, nesting habitat is under constant threat from coastal development and construction (Dow *et al.* 2007; Crespo and Diez 2016; Flores and Diez 2016).

Coastal development impacts include construction of buildings and pilings on the beach; increased erosion; artificial lighting; pollution; recreational beach equipment and other obstacles on the beach; beach driving; increased human disturbance; and mechanized beach cleaning (Lutcavage *et al.* 1997; USFWS 1999; Hernandez *et al.* 2007; Dow *et al.* 2007; Trinidad and Tobago Forestry Division *et al.* 2010; Flores and Diez 2016).

Driftwood found on nesting beaches also has the potential to alter nesting beach habitat and obstruct nesting females and hatchlings, as seen in Gandoca, Costa Rica (Chacón-Chaverri and Eckert 2007). These threats impact nesting habitat by reducing the amount and quality of suitable beaches, preventing or deterring nesting females from using optimal locations, destroying nests, eggs, and hatchlings, and preventing hatchlings from successfully reaching the ocean (USFWS 1999; Chacón-Chaverri and Eckert 2007;

Hernandez *et al.* 2007; Witherington *et al.* 2014). Development involving the construction of tall buildings and clearing of vegetation can also alter sand temperatures and skew sex ratios (Gledhill 2007).

Development occurs to varying extents throughout the range of the DPS, but most leatherback nesting occurs in proximity to some coastal development. The Florida shoreline is extensively developed outside wildlife refuges (Witherington *et al.* 2011). In Grenada, nearly 20 percent of all nests surveyed from 2001 to 2005 occurred in an area affected by development, resulting in ongoing run-off onto nesting beaches (Maison *et al.* 2010). In Trinidad, increasing rural and commercial beachfront development is a concern, especially on the east coast where the main nesting beaches are located (Trinidad and Tobago Forestry Division *et al.* 2010), including Grande Riviere, the largest nesting aggregation of this DPS. Likewise, several Tobago beaches are densely developed for commercial tourism, resulting in reduced turtle access to potential nesting sites due to buildings, umbrellas, and other recreational equipment (Trinidad and Tobago Forestry Division *et al.* 2010). Development in Puerto Rico, in particular Playa Grande-El Paraiso (*i.e.*, Dorado Beach, which is considered to be the most important nesting beach in Puerto Rico), is also a notable concern (Crespo and Diez 2016; Flores and Diez 2016). There, ecosystems continue to be threatened by coastal development, even though the coastal zone is protected by the Maritime-Terrestrial Zone designation (*i.e.*, Coastal Public Trust Lands; Flores and Diez 2016).

Coastal development likely influences leatherback nest placement and subsequent nest success, which is the percentage of nesting attempts (*i.e.*, emergences onto the beach) that result in eggs being deposited. On Margarita Island, Venezuela, Hernandez *et*

*al.* (2007) found that leatherback nesting aggregated towards the portions of the beach with fewer risk factors, such as light pollution and concentrations of beach furniture. This change in nesting behavior resulted in females nesting in less optimum areas (*e.g.*, areas with lower hatching success), thus affecting the reproductive potential of leatherback turtles in this region.

The magnitude of development is also changing in some areas, where nest placement and success may be affected in the future. For instance, the area around Cayenne, French Guiana, is undergoing increased urbanization and recreational use (Fossette *et al.* 2008). In recent years, nesting has increased at Cayenne and eastern beaches compared to the western Awala-Yalimapo beaches (Réserve Naturelle de l'Amana data in Berzins 2018 and KWATA data in Berzins 2018). As such, more nesting in French Guiana is exposed to coastal development and the associated threats, and these threats are likely to continue and increase.

#### Beach Erosion and Inundation

While erosion is often intensified due to anthropogenic influences, natural features in some areas result in high erosion rates and unstable beaches, thus affecting leatherback nesting. For instance, the Maroni River influence in the Guianas (French Guiana especially) has resulted in highly dynamic and unstable beaches, with shifting mudflats making nesting habitat unsuitable (Crossland 2003; Goverse and Hilterman 2003; Fossette *et al.* 2008). Beaches are often created and lost along the coast of French Guiana (Kelle *et al.* 2007). For example, remote beaches in western French Guiana experience significant beach erosion and several disappeared, reducing or preventing monitoring (and likely nesting). In Suriname, Braamspunt Beach at the mouth of the

Suriname River is moving west, out of the established Wia Wia Nature Reserve and may disappear in the next several years (M. Hiwat, WWF, pers. comm., 2018). This is significant in that Braampunt is currently the main nesting beach in Suriname. The second highest nesting area in Suriname, Galibi Beach, is also experiencing significant erosion and becoming narrower. Similar beach erosion is occurring in Guyana, as well as in Trinidad and Tobago (Reichart *et al.* 2003; Trinidad and Tobago Forestry Division *et al.* 2010). At some Trinidad and Tobago nesting sites (*e.g.*, Fishing Pond, Matura, Grande Riviere, and Great Courland Bay), rivers emerge onto nesting beaches and create additional erosion during the nesting season (Godley *et al.* 1993; Lee Lum 2005), intensifying nest loss (up to 35 percent of nests; Trinidad and Tobago Forestry Division *et al.* 2010).

Seasonal erosion also occurs at most Caribbean nesting beaches. A survey of Wider Caribbean Regions found that erosion/accretion was the highest threat to nesting habitat (Dow *et al.* 2007). For example, at Playa Gandoca, Costa Rica, erosion from strong coastal drift currents is thought to be one of the largest obstacles to hatching success, destroying greater than 10 percent of all nests laid in some years (Chacón-Chaverri and Eckert 2007). In 2006 and 2007, coastal erosion and inundation accounted for 33 to 42 percent of nest loss in southern Panama and 29 to 48 percent on Caribbean Colombia beaches (Patiño-Martínez *et al.* 2008).

Inundation of nests is also a concern. Leatherback turtles generally nest closer to the water than other sea turtles (Caut *et al.* 2010). If nests are laid too close to the high tide line, they are subjected to erosion and inundation, which can result in egg mortality from suffocation or curtailed embryonic development (Chacón-Chaverri and Eckert

2007; Caut *et al.* 2010). This inundation phenomenon occurs on multiple nesting beaches and is particularly of concern in areas with high tidal influence and dynamic coastlines. On Krofajapasi beach in Suriname, 31.6 percent of nests laid by females were below the spring high tide level and determined to be “doomed” clutches (Dutton and Whitmore 1983). Similarly, in Gandoca, Costa Rica, 37 percent of nests from 1990 to 2004 were laid in the low tide zone and would have been inundated if not relocated (Chacón-Chaverri and Eckert 2007). In St. Croix, 43 percent of the nests (with a range of 25 to 68 percent) were considered to be “doomed” each season (McDonald-Dutton *et al.* 2001), but beginning in 1983, all doomed clutches were relocated to improve hatching success (Dutton *et al.* 2005). Without intervention, these nests would likely have been lost. On Awala-Yalimapo, French Guiana, 27 of 89 nests were overlapped by tide at least once during the incubation period, and the hatching success was on average significantly lower in overwashed nests (Caut *et al.* 2010). Observed mortality was 100 percent in the intertidal zone at sites along the coasts of Panama and Colombia, with an overall nest loss by erosion and inundation ranging from 16 to 48 percent among three major nesting sites (Patiño-Martínez *et al.* 2008). While levels of inundation and resulting declines in hatching success have been noted at multiple sites throughout the range of the NW Atlantic DPS, the specific impacts of inundation may be variable. Hilterman and Goverse (2007) noted that leatherback nests can tolerate relatively high levels of inundation, so hatching may still be successful despite proximity to the tide line. Because of this, and because it may affect natural sex ratios (Mrosovsky and Yntema 1980), the relocation of nests susceptible to inundation was abandoned in 2002 in Suriname (Hilterman and Goverse 2007); only nests directly threatened by beach erosion are relocated, under

certain circumstances. Other nations still relocate nests to reduce the impacts of erosion. However, as mentioned, such practices may result in cooler nests and affect sex ratios (Spanier 2008). While eggs relocated to hatcheries could have been lost under natural circumstances, due to coastal erosion and inundation in some areas (Dutton and Whitmore 1983, Chacón-Chaverri and Eckert 2007), hatching success in relocated nests is often lower than *in situ* nests (Revuelta *et al.* 2014; Valentin-Gamazo *et al.* 2018; Florida Department of Environmental Protection unpublished data 2018).

Such naturally dynamic areas make it difficult to protect nesting beach habitat and accurately assess leatherback nesting trends. This is particularly noteworthy given that nesting females use high energy, erosion-prone beaches, which often result in high nest loss (Chacón-Chaverri and Eckert 2007; TEWG 2007; Spanier 2008; Trinidad and Tobago Forestry Division *et al.* 2010). However, leatherback turtles in the Guianas seem to have adapted to this constant geomorphological change of beaches. When new beaches develop, they may be colonized within months by nesting females, who take advantage of the fresh, clean sand (or seashells, in Guyana) and absence of entangling or deep-rooted beach vegetation (TEWG 2007).

Nest site selection by leatherback turtles is still poorly understood (Maison *et al.* 2010), but nesting females may be changing their nesting patterns due to erosion. Spanier (2008) found that nesting females at Playa Gandoca, Costa Rica, appear to actively select nest sites that are not undergoing extensive erosion, with slope considered to be the cue for site selection. A similar result was found on Grande Riviere, Trinidad, with a nesting shift from east to west throughout the season as an apparent response to erosion on the eastern end of the nesting beach (Lee Lum 2005). Further, Maison *et al.* (2010) studied

nest placement in Grenada and discovered that leatherback turtles seemed to respond to the accretion of the north facing beach and erosion of the east facing beach in 2005 by nesting more often on the north facing beach. If erosion is increasing in existing nesting locations, nesting may occur in areas with lower success rates, thus affecting productivity. In addition, leatherback nests are deeper than those of other sea turtles; water content and salinity typically increase with depth, leading to a decrease in sea turtle hatching success (Foley *et al.* 2006).

#### Erosion Control, Nearshore Shoreline Stabilization Structures, and Other Barriers

A widespread strategy to reduce coastal erosion is to construct erosion control structures. However, these structures reduce the amount of available nesting habitat. Also, when beachfront development occurs, the site is often engineered to protect the property from erosion. This type of shoreline engineering, collectively referred to as beach armoring, includes sea walls, rock revetments, riprap, sandbag installations, groins and jetties. Beach armoring can result in permanent loss of a nesting beach through accelerated erosion and prevention of natural beach/dune accretion. These impacts can prevent or hamper nesting females from accessing suitable nesting sites (USFWS 1999). Clutches deposited seaward of these structures may be inundated at high tide or washed out entirely by increased wave action near the base of the erosion control structures. As these structures fail and break apart, they spread debris on the beach, thus creating additional impacts to hatchlings and nesting females.

In the southeastern United States, numerous erosion control structures that create barriers to nesting have been constructed. In Florida, the total amount of existing and potential future armoring along the coastline is approximately 24 percent (164 miles;

FDEP, pers. comm., 2018). This assessment of armoring does not include other structures that are a barrier to sea turtle nesting, such as dune crossovers, cabanas, sand fences, and recreational equipment. Additionally, jetties have been placed at many ocean inlets in the United States to keep transported sand from closing the inlet channel. The installation of jetties resulted in lower loggerhead and green turtle nesting density updrift and downdrift of the inlets, leading researchers to propose that beach instability from both erosion and accretion may discourage turtle nesting (Witherington *et al.* 2005). Leatherback nesting near jetties and inlets is low, possibly reflecting their avoidance of such areas. There are some efforts, such as the Coastal Construction Control Line Program, that provide protection for Florida's beaches and dunes while allowing for continued use of private property. However, armoring structures on and adjacent to the nesting beach continue to be permitted and constructed on the nesting beaches of Florida, as in other nations where the DPS nests.

Due to erosion, beach nourishment is a frequent activity in some developed areas, and many beaches are on a periodic nourishment schedule. Beach nourishment may result in direct burial and disturbance to nesting females, if conducted during the nesting season. It may also result in changes in sand density, beach hardness, beach moisture content, beach slope, sand color, sand grain size, sand grain shape, and sand grain mineral content, if the placed sand is dissimilar from the original beach sand (Nelson and Dickerson 1988; USFWS 1999). These changes can affect nest site selection, digging behavior, incubation temperature (and hence sex ratios), gas exchange parameters within incubating nests, hydric environment of the nest, hatching success and hatchling emerging success (Lutcavage *et al.* 1997; Steinitz *et al.* 1998; Ernest and Martin 1999;

USFWS 1999; Rumbold *et al.* 2001; Brock *et al.* 2009). On severely eroded sections of beach, where little or no suitable nesting habitat previously existed, beach nourishment has been found to result in increased nesting (Ernest and Martin 1999). However, on most beaches in the southeastern United States, nesting success typically declines for the first year or two following nourishment, even though more nesting habitat is available for turtles (Trindell *et al.* 1998; Ernest and Martin 1999; Herren 1999; Brock *et al.* 2009). Further, nourishment projects result in heavy machinery, pipelines, increased human activity and artificial lighting on the project beach, further affecting nesting females and beach habitat. Overall, the impacts of beach nourishment to this DPS are not as widespread as other threats to nesting habitat, as Dow *et al.* (2007) found that only four nations (Anguilla, Cuba, Mexico, and United States) reported frequent or occasional beach nourishment.

#### Artificial Lighting

Coastal development also contributes to habitat degradation by increasing light pollution, which can result in hatchling and nesting female disorientation, altering behavior and leading to mortality. In Florida, from 2013 to 2017, a total of 341 leatherback nests (representing the whole or majority of hatchlings in the nest) and five nesting females were disoriented (FWC unpublished data 2018). Artificial lighting ranked as the third highest threat to nesting/hatching turtles in the Wider Caribbean Region (Dow *et al.* 2007). For example, urban development is significant in Puerto Rico, with light pollution (as well as coastal erosion and deforestation) occurring near leatherback nesting beaches (Crespo and Diez 2016). Fortunately, some of the major nesting beaches in this DPS are located in comparatively remote areas, and large-scale

development is currently less of an issue there (Trinidad and Tobago Forestry Division *et al.* 2010; NMFS and USFWS 2013). That said, even within the same country, light pollution is variable. Fossette *et al.* (2008) reported that in French Guiana, light pollution from residential areas is a problem at Cayenne Beach, but it is not an issue at Awala-Yalimapo. Similarly, lighting is not a significant problem on nesting beaches in Trinidad, but is a concern in Tobago (Trinidad and Tobago Forestry Division *et al.* 2010). With the risk of increased development in some of these relatively remote areas, additional light pollution is anticipated, and disorientation of hatchlings and adults from such lighting may become a bigger problem. In Costa Rica, beachfront lighting is increasing and may become problematic at Gandoca Beach (Chacón-Chaverri and Eckert 2007) and Tortuguero (de Haro and Troëng 2006).

Light pollution has been managed to some extent (Witherington *et al.* 2014). Lighting in Florida is regulated by multiple rules and regulations including Florida statutes, the Florida Building Code, and local lighting ordinances (Witherington *et al.* 2014). In addition, the Florida Department of Transportation and local governments have adopted lighting-design standards. A total of 82 municipalities in Florida have adopted lighting ordinances to minimize the impact of lighting on adjacent sea turtle nesting beaches (Witherington *et al.* 2014). However, compliance and enforcement is lacking in some areas. Further, lighting away from areas covered by beachfront ordinances is unregulated, resulting in urban glow. Although outreach and conservation programs control the impacts of lighting in some other locations, such as Costa Rica, Mexico, and Puerto Rico (Lutcavage *et al.* 1997; Crespo and Diez 2016), a majority of nations do not have regulations in place.

## Sand Extraction

Extracting sand from nesting beaches for construction projects has a detrimental effect on the amount of available nesting beach habitat and also accelerates erosion (resulting in the aforementioned associated impacts). Sand mining occurs in most Wider Caribbean nations to varying extent and frequency (Dow *et al.* 2007). In particular, beach sand mining has been extensive at Matura Bay and Blanchisseuse in Trinidad (Trinidad and Tobago Forestry Division *et al.* 2010). Some nations regulate sand mining: in St. Lucia, the Conservation and Management Act of 2014 requires a certificate of environmental approval for projects removing sand from nesting beaches.

## Removal of Native Vegetation

In some nations, upland deforestation and the resultant deposition of debris and garbage can destroy or modify nesting beaches. The debris can block access of gravid (pregnant) females and fatally trap emergent hatchlings (Chacón-Chaverri and Eckert 2007). The accumulation of logs reduces the amount of available nesting habitat, possibly forcing leatherback females to nest in suboptimal locations (TEWG 2007). Deforestation due to coastal development is a notable concern in Puerto Rico (Crespo and Diez 2016).

## Vehicular Traffic

Beach driving also occurs in most nations throughout the range of this DPS (Chacón-Chaverri and Eckert 2007; Dow *et al.* 2007; Trinidad and Tobago Forestry Division *et al.* 2010). In the United States, vehicular driving is allowed on certain beaches in Florida (*e.g.*, Duval, St. Johns, and Volusia Counties). Beach driving reduces the quality of nesting habitat in several ways. Vehicle ruts on the beach can prevent or impede hatchlings from reaching the ocean following emergence from the nest (Mann

1977; Hosier *et al.* 1981; Cox *et al.* 1994; Hughes and Caine 1994). Sand compaction by vehicles hinders nest construction and hatchling emergence from nests (Mann 1977; Gledhill 2007). Vehicle lights and vehicle movement on the beach after dark can deter females from nesting and disorient hatchlings. Additionally, vehicle traffic contributes to erosion, especially during high tides or on narrow beaches where driving is concentrated on the high beach and foredune.

### Vegetation

Beach vegetation (native and non-native) can affect turtle nesting productivity by obstructing nest construction and potentially drying the sand (resulting in egg chamber collapse). Vegetation can form impenetrable root mats that can invade and desiccate eggs and affect developing embryos, impede hatchling emergence, and trap hatchlings (Conrad *et al.* 2011). Non-native vegetation has invaded many coastal areas and often outcompetes native plant species (USFWS 1999). The occurrence of exotic vegetation (or loss of native vegetation) was recognized as a medium-ranked threat in many Wider Caribbean nations (Dow *et al.* 2007). The Australian pine (*Casuarina equisetifolia*) is particularly harmful to sea turtles (USFWS 1999). Australian pines cause excessive shading of the beach that would not otherwise occur. Studies of loggerhead turtles in Florida suggest that nests laid in shaded areas are subjected to lower incubation temperatures, which may alter the natural hatchling sex ratio (Marcus and Maley 1987; Schmelz and Mezich 1988). Fallen Australian pines limit access to suitable nest sites and can entrap nesting females (Reardon and Mansfield 1997). The shallow root network of these pines can interfere with nest construction (Schmelz and Mezich 1988). Dense

stands of Australian pine have overtaken many coastal areas throughout central and south Florida.

While non-native vegetation can affect nesting habitat throughout the range of the DPS, native vegetation can also affect productivity. For instance, at Sandy Point, St. Croix, changing erosion-accretion cycles led to native *Ipomoea pes-caprae*, a creeping vine, extending into the nesting area in some years. Nesting females at Sandy Point typically avoided nesting in vegetation, resulting in more nests laid near the high-tide line (Conrad *et al.* 2011). As a result, *Ipomoea pes-caprae* decreased nest productivity by reducing leatherback hatching and emergence (percentage of hatchlings that emerge from the nest) success rates (Conrad *et al.* 2011).

#### Mitigations to Habitat Modification

Nesting habitat disruptions are minimized in some areas. Several areas in the NW Atlantic DPS range are under U.S. Federal ownership as National Wildlife Refuges in Florida (Archie Carr and Hobe Sound), Puerto Rico (Culebra and Vieques) and St. Croix (Sandy Point). Beaches in some Wider Caribbean countries are also protected. In Trinidad, Matura and Fishing Pond beaches were declared Prohibited Areas in 1990, and the nesting beach at Grande Riviere in 1997. In 1998, the Amana Nature Reserve, which includes Awala-Yalimapo beach and a 30 m wide marine fringe, was established in French Guiana. In Suriname, the Wia Wia Nature Reserve was implemented in 1961 (amended and enlarged in 1966 to protect sea turtles), and in 1969, the Marowijne beaches were declared a sanctuary (the Galibi Nature Reserve; Schulz 1971). In addition, Tortuguero National Park, Costa Rica, was established in 1976 to protect nesting habitat (Bjorndal *et al.*, 1999). Terrestrial habitat in these areas is therefore protected from the

above threats to some extent. USFWS and NMFS also designated as critical habitat for leatherback turtles the nesting beaches at Sandy Point, St. Croix (43 FR 43688; September 26, 1978) and surrounding marine waters (44 FR 17710; March 23, 1979), which benefits the turtles in this DPS. However, if ESA protections did not continue (*i.e.*, if this species were no longer listed), these protections would be lost.

#### Marine Habitat Modifications

In the marine environment, habitat threats include anthropogenic noise and offshore lighting. We discuss other threats to marine habitat and prey (*e.g.*, marine pollution, oil exploration, and climate change) in later sections. Anthropogenic noise impacts the marine habitat of the DPS. Dow Piniak *et al.* (2012) measured hearing sensitivity of leatherback hatchlings. They found that hatchlings are able to detect sounds underwater and in air, responding to stimuli between 50 and 1200 Hz in water and 50 and 1600 Hz in air, with maximum sensitivity between 100 and 400 Hz in water and 50 and 400 Hz in air. This sensitivity range overlaps with the frequencies and levels produced by many anthropogenic sources used in the North Atlantic, including seismic airgun arrays, drilling, low frequency sonar, shipping, pile driving, and operating wind turbines. These noise sources may affect leatherback turtles' marine habitat and subsequently impact distribution and behavior. Offshore artificial lighting occurs in some marine waters of this DPS (Dow *et al.* 2007) but is less of a threat than beachfront lighting throughout the range of the DPS.

#### Summary

We conclude that nesting females, hatchlings, and eggs are exposed to the loss and modification of nesting habitat, especially as a result of coastal development and

armorings, erosion, and artificial lighting. These threats impact the DPS by reducing nesting and hatching success, thus, lowering the productivity of the DPS. Based on the information presented above, we conclude that habitat reduction and modification pose a threat to the NW Atlantic DPS.

*Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

Overutilization is a threat to the NW Atlantic DPS, mostly due to poaching of turtles and eggs in certain nations. Legal harvest of turtles and eggs also occurs in some nations.

While the vast majority of nations within the range of the NW Atlantic DPS protect leatherback turtles from harvest, it is legal in some Caribbean and Central American nations (Brautigam and Eckert 2006; Dow *et al.* 2007; Richardson *et al.* 2013; Horrocks *et al.* 2016). For example, the harvest of leatherback turtles over 20 pounds is allowed in Montserrat and Dominica from October 1 to May 31; Saint Lucia allows leatherback turtles over 65 pounds to be taken from October 2 to February 27; and St. Kitts and Nevis allows take of leatherback turtles over 350 pounds from October 2 to February 27 (Montserrat Turtles Act 2002; Bräutigam and Eckert 2006). In some nations, commercial use is prohibited, but traditional use is allowed, which can still diminish protection. In Colombia, subsistence fishing of sea turtles is permitted, and indigenous use is allowed in Honduras. Traditional or cultural use is permitted in Belize with prior approval (Bräutigam and Eckert 2006). However, regular leatherback nesting does not occur in Belize, and its occurrence in surrounding waters is infrequent, reducing the impact of such mortality. Legal harvest throughout the range of this DPS is not monitored, and the precise magnitude of this threat is not clear. However, we conclude

that legal harvest of turtles is significant because, when it occurs, nesting turtles are targeted, removing the most important individuals from the population. More often, leatherback eggs, rather than turtle meat, are harvested (TEWG 2007; Patiño-Martínez *et al.* 2008), reducing productivity in the DPS.

Poaching of turtles and eggs occurs throughout the NW Atlantic DPS, and Dow *et al.* (2007) ranked it as a threat for all turtle species on the beaches in the Wider Caribbean Region. In Panama, interviews with locals revealed that the development of a new way for cooking leatherback turtle meat has resulted in a recent increase of its consumption in Changuinola, Bocas del Toro Province (CITES Secretariat 2019). Adult turtles are killed in Panama and on remote beaches in Trinidad and Tobago (Troëng *et al.* 2002; Ordoñez *et al.* 2007; Trinidad and Tobago Forestry Division *et al.* 2010). Most poaching, however, targets eggs, and the level often is determined by how much monitoring and activity to deter poachers occur on the nesting beaches. Some of the highest levels of egg poaching occur throughout Costa Rica (Troëng *et al.* 2004). Troëng *et al.* (2007) found that, at a minimum, between 13 to 21.5 percent of nests between 2000 and 2005 were illegally collected at Tortuguero. Poaching of leatherback nests was higher outside Tortuguero National Park (minimum 33 percent) than within the National Park (minimum 9 percent) in 2005 (de Haro and Troëng 2006). At Pacuare Playa, Costa Rica, 55 percent of nests were poached in 2012 (Fonseca and Chacón 2012) and 42 percent were poached in 2017, which was the lowest level since Latin American Sea Turtles (LAST) started to monitor in 2012 (LAST 2017). Poaching at Gandoca Beach has decreased over time (previously 100 percent of nests were poached), but rates still averaged 15.5 percent annually from 1990 to 2004 (Chacón-Chaverri and Eckert 2007). In the Dominican Republic, poaching

is also high. Revuelta *et al.* (2012) determined the poaching of clutches in Jaragua National Park and Saona Island ranged from 0 to 100 percent from 2006 to 2010, with averages of 19 percent on western Jaragua National Park beaches, 71 percent on eastern Jaragua National Park beaches, and 74 percent on Saona. Poaching also occurs at relatively high levels in Colombia (*e.g.*, 22 to 31 percent of clutches at Playona in 2006 and 2007; Patiño-Martínez *et al.* 2008) and, to some extent, in most other Caribbean nations (*e.g.*, Guyana and Grenada). Poaching is likely more prevalent, and occurs at higher levels, on unmonitored or unprotected beaches (Dow *et al.* 2007; TEWG 2007; Troëng *et al.* 2007; Trinidad and Tobago Forestry Division *et al.* 2010; K. Charles, Oceans Spirits Inc., pers. comm., 2018).

Poaching has been significantly reduced at some nesting beaches. In Suriname, high levels of egg poaching (at least 26 percent of nests) occurred in the late 1990s, but due to better monitoring and enforcement, that level has been significantly reduced (Hilterman and Goverse 2007; M. Hiwat, WWF, pers. comm., 2018). Poaching was also a major problem in Trinidad, but levels have been reduced with more people monitoring the beach (Trinidad and Tobago Forestry Division *et al.* 2010). The Marine Turtle Conservation Act of 2004 (MTCA) funds activities in Panama in an attempt to reduce poaching. At Chiriqui Beach, Panama, intense monitoring efforts have attempted to reduce poaching. However, of the monitored nests, 29 leatherback nests (0.7 percent) were still poached in 2017 (Sea Turtle Conservancy 2017). Further, poaching in Panama outside the monitored areas still occurs, with the clandestine sale of eggs widespread (Brautigam and Eckert 2006). In St. Croix, almost 100 percent of nests were lost to poaching prior to 1981 (Garner *et al.* 2017). However, the establishment of the USFWS

Sandy Point National Wildlife Refuge has reduced egg poaching to 0 to 1.8 percent annually as a result of nightly patrols (Garner *et al.* 2017).

Poaching of eggs is widespread throughout the Caribbean, especially on beaches of Costa Rica, Dominican Republic, and Colombia. The total number of individuals affected by poaching cannot be quantified at this time. However, we conclude that many eggs and some adults are affected by illegal poaching at nesting beaches. Adults and eggs are also exposed to legal harvest in some nations. The legal and illegal harvest of nesting females reduces both abundance (through loss of nesting females) and productivity (through loss of reproductive potential), resulting in a high impact to the DPS. Legal and illegal egg harvest reduces productivity only. Thus, we conclude that overutilization poses a threat to the DPS.

#### *Disease or Predation*

For the NW Atlantic DPS, information on diseases is limited, but predation is a well-documented threat.

Much of the available information on disease in leatherback turtles was obtained by necropsy of stranded large juvenile and adult turtles; the health implications of various conditions reported in this species are incompletely understood. Solitary large intestinal diverticulitis of unknown etiology was found in 31 subadult and adult leatherback turtles stranded in U.S. waters (Stacy *et al.* 2015). All lesions were chronic and unrelated to the cause of death in all cases, although risk of perforation and other complications are possible. Adrenal gland protozoal parasites were found in 17 leatherback turtles in North American waters examined from 2001 to 2014; it is not currently known whether parasitism affects adrenal function (Ferguson *et al.* 2016). In addition, leatherback turtles

are hosts for several trematode parasites (flatworms), known species of which also occur in hard-shelled sea turtles (Manfredi *et al.* 1996, Greiner *et al.* 2013). In general, trematodes are frequently encountered without any apparent clinical effect on the turtle host but can affect some heavily parasitized individuals. With regard to other types of potential disease-causing organisms, there are a small number of reports of bacterial infections in stranded individuals (Poppi *et al.* 2012; Donnelly *et al.* 2016). A variety of other bacteria have been documented in nesting females on beaches in Costa Rica (Santoro *et al.* 2008) and St. Kitts (Dutton *et al.* 2013); the majority of identified bacterial species may be considered as potential or opportunistic pathogens for sea turtles. A putative case of fibropapilloma, a virus-associated tumor-causing disease in sea turtles, has been reported in a leatherback; this disease is considered very rare in the species (Huerta *et al.* 2002).

An in-water health assessment was performed on 12 turtles directly caught at-sea and seven turtles bycaught in fishing gear in the NW Atlantic Ocean (Innis *et al.* 2010). Most were determined to be in good health, but several exhibited evidence of past injuries. The blood chemistry of entangled turtles indicated stress, seawater intake, and reduced food consumption associated with entanglement. In addition, Perrault *et al.* (2012) examined baseline blood chemistry metrics (*i.e.*, plasma protein electrophoresis, hematology, and plasma biochemistry) as indicators of health for nesting females in Florida. They found that multiple measures of maternal health significantly correlated with leatherback hatching and emergence success (the percentage of hatchlings that emerge from the nest).

From these data, we estimate that the exposure of eggs, juveniles, and adults to disease is low. The impact of disease cannot be quantified at this time as we have no documentation of any deaths or reductions in productivity directly related to disease. However, disease may compound the effects of or have synergistic effects with other threats to the species and related physiologic derangements. We conclude that disease, alone or in combination with other threats, is likely a threat to the DPS.

Throughout the range of the DPS, predation is a threat to leatherback eggs, hatchlings, and adults. Eckert *et al.* (2012) provides an exhaustive list of the documented predators for each life stage and area. For eggs in the NW Atlantic DPS, predators include ants (*Dorylus spininodis*), fly larvae (*Diptera* spp.), locust larvae (*Acrididae* spp.), mole crickets (*Scapteriscus didactylus*), ghost crabs (*Ocypode quadratus*), vultures (*Cathartidae*), dogs (*Canis familiaris*), cattle (*Bos taurus*; due to trampling), armadillo (*Dasypodidae*), opossum (*Didelphis marsupialis*), coati (*Nasua* spp.), and raccoons (*Procyon lotor*); see Eckert *et al.* 2012).

In particular, dog predation of eggs occurs in many areas (*e.g.*, Colombia, French Guiana, Guyana, Panama, Puerto Rico, and Trinidad and Tobago). In Trinidad, where the largest nesting aggregation occurs, feral dogs are considered to be the primary threat to eggs, even above poaching and coastal erosion (Trinidad and Tobago Forestry Division *et al.* 2010). On Chiriqui Beach, Panama, 54 percent of the monitored leatherback nests were depredated by dogs in 2003 and approximately eight percent in 2004 (Ordoñez *et al.* 2007). Such predation may be reduced as a result of protection efforts funded by the MTCA. In Playa California, Maunabo, Puerto Rico, more than 30 percent of the leatherback nests were depredated by stray dogs in 2012 (Crespo and Diez 2016). A

public outreach project in Puerto Rico was established in 2013 to reduce this impact. Puerto Rico is a U.S. territory; if ESA protections were removed, it is likely that predation rates would be higher.

Egg predation by other species is also a notable concern in some areas. On Gandoca Beach, Costa Rica, dipteran larvae infestation exceeded 75 percent of nests in 2005 and 2006 (Gautreau *et al.* 2008). In French Guiana, on average, mole crickets preyed on 18 percent of all eggs (Maros *et al.* 2003). These threats are likely to continue, as no predator screening typically occurs in Wider Caribbean nations due to the potential for increased poaching as well as logistical difficulties in these areas of high density nesting. Nest loss to predators was found to be the seventh ranked threat to turtles (all species, not specific to leatherback turtles) on nesting beaches in the Wider Caribbean Region, and have been noted to frequently occur in Honduras, Mexico, Panama, Puerto Rico, and Venezuela (Dow *et al.* 2007).

Hatchlings are preyed upon by a wide variety of species, including mole crickets, ghost crabs, horse-eye jack fish (*Caranx latus*), gray snapper (*Lutjanus griseus*), tarpon (*Megalops atlanticus*), vultures, hawks (Accipitridae), gulls (*Larus* spp.), night heron (*Nyctanassa violacea*), frigate birds (Fregatidae), dogs, mongoose (*Atilax paludinosus*), coati, and raccoons (Eckert *et al.* 2012). Again, dogs are a serious threat to leatherback hatchlings in some areas, and especially in Puerto Rico (Crespo and Diez 2016).

There are few documented predators to subadults and adult leatherback turtles, presumably because of their large size and pelagic behavior. Predation by sharks (Elasmobranchii) and killer whales (*Orcinus orca*) has been reported in Barbados and St. Vincent, respectively (Caldwell and Caldwell 1969; Horrocks 1989). Sharks have also

been reported to prey on nesting females off St. Croix, USVI (DeLand 2017; Scarfo *et al.* 2019). Over the past 6 years, researchers at Sandy Point have observed an apparent increase in injuries to leatherback turtles (K. Stewart, NMFS, pers. comm., 2019). These injuries, many of them consistent with shark predation, affect up to 70 percent of all nesting females at the beach (Scarfo *et al.* 2019). While some turtles probably survive these encounters, it is unknown how many encounters result in mortality or reduced nesting effort. Jaguars (*Panthera onca*) prey on nesting females in some areas, including Suriname, French Guiana, Guyana, and Costa Rica (see Eckert *et al.* 2012). While three nesting females were killed by jaguars at Tortuguero, Costa Rica, from 1998 to 2005, this mortality is only considered to be a minor threat and is therefore unlikely to cause a population decline on its own (Troëng *et al.* 2007). Archibald and James (2018) examined 228 leatherback turtles for injuries off Atlantic Canada and on Matura, Trinidad, and found 15.7 percent of turtles exhibited injuries of suspected predatory origin.

Predation on early life stages is natural; however, at high rates, it reduces the viability of the DPS (see the Status Review). Predation primarily reduces productivity via reduced egg and hatching success and the loss of hatchlings. Predation on nesting females reduces abundance and productivity. We conclude that predation is a threat to the NW Atlantic DPS.

#### *Inadequacy of Existing Regulatory Mechanisms*

Many regulatory mechanisms (including state, Federal and international) have been promulgated to protect leatherback turtles, eggs, and nesting habitat throughout the range of the NW Atlantic DPS. We reviewed the objectives of each regulation and to

what extent they adequately address the targeted threat (*i.e.*, the threat that the regulation was intended to address). The effectiveness of many international regulations was evaluated by Hykle (2002), who found that international instruments often do not realize their full potential, either because they do not include all key countries, do not specifically address sea turtle conservation, are handicapped by the lack of a sovereign authority that promotes enforcement, or are not legally binding.

National regulatory mechanisms are described in full in the Status Review Report. Although these regulatory mechanisms provide some protection to the species, most inadequately reduce the threat they were designed to address, generally as a result of poor implementation or incomplete enforcement. Specifically, existing regulatory mechanisms continue to be inadequate to control impacts to nesting beach habitat and overutilization (harvest of turtles and eggs) for this DPS. In addition, regulatory mechanisms are inadequate to reduce several other threats including bycatch in fishing gear, vessel strikes, and marine debris. Despite existing regulatory mechanisms, bycatch from fisheries (discussed in detail along with existing regulatory mechanisms in the Fisheries Bycatch section), incomplete nesting habitat protection, and poaching remain major threats to the DPS.

#### *Fisheries Bycatch*

Fisheries bycatch is the primary threat to the NW Atlantic DPS. Bycatch occurs throughout the range of the DPS, affecting juveniles, subadults, and adults.

Finkbeiner *et al.* (2011) analyzed sea turtle bycatch across all commercial U.S. fisheries from 1990 to 2007. They examined sea turtle bycatch reduction based on the year a particular fishery implemented bycatch reduction measures. Prior to implementing

bycatch reduction measures, approximately 3,800 leatherback interactions, of which 2,300 were lethal, occurred in U.S. Atlantic Ocean and GOM commercial fisheries annually. After bycatch reduction measures were implemented, 1,400 leatherback turtles, 40 of those dead, were estimated to be taken annually in the Atlantic Ocean. The Atlantic/GOM pelagic longline fishery was responsible for the most annual interactions (n = 900) and mortality events (n = 17) in the Atlantic Ocean, followed by the southeast Atlantic /GOM shrimp trawl fishery (Finkbeiner *et al.* 2011). These estimates represent minimum numbers of actual bycatch and mortality. Because the observer coverage for these fisheries is low (so some bycatch may not be observed and observed effort may not be a true representation of actual fleet effort), not all fisheries are observed and thus some are not included in these estimates. Interactions are difficult to observe if gear modifications are in place, and so the methods used are conservative (Finkbeiner *et al.* 2011).

In the Wider Caribbean Region, reports of leatherback bycatch in fisheries are common. In a survey of Caribbean nations, Dow *et al.* (2007) ranked fisheries bycatch among the highest in-water threat to sea turtles. Many fisheries in less industrialized nations are coastal and small-scale, but these fisheries are reported to have significant ecological impacts due to their high bycatch discards and impacts to the marine environment (Shester and Micheli 2011). Of particular concern are leatherback bycatch in artisanal nearshore and offshore gillnet, longline and trawl fisheries (Barrios-Garrido and Montiel-Villalobos 2016). Information on fisheries bycatch is collected mostly from stranding records but also from fisher surveys (Moncada *et al.* 2003; Delamare 2005; Madarie 2006, 2010, 2012) and observations of nesting females. Hilterman and Goverse

(2007) recorded fisheries related injuries on nesting females in Suriname. In 2002, 16.9 percent of the nesting females had fisheries- related injuries; in 2003, at least 18.3 percent had such injuries; and in 2005, 9 percent (Hilterman and Goverse 2007). From 2000 to 2003, an average of 28 leatherback turtles stranded on the Suriname survey beaches. Although no cause of death was immediately apparent, Hilterman and Goverse (2007) indicated that the mortalities were fisheries-related, based upon the fisheries that occur offshore with high bycatch and documented fisheries-related injuries on nesting leatherback turtles at the same time. On the western oceanic nesting beaches of French Guiana, injuries consistent with fisheries interactions (*e.g.*, scars, wounds) were recorded on 8.4 percent (n=1,259) of nesting females in 2003 (Morisson *et al.* 2003). In Venezuela, 55 percent of strandings from 2001 to 2007 (n = 57) exhibited evidence of fisheries interactions (Barrios-Garrido and Montiel-Villalobos 2016). Most recently, an injury assessment of 228 leatherback turtles from two foraging areas off the Atlantic coast of Canada and Trinidad nesting beaches found 19 percent of turtles exhibited injuries indicative of entanglement in lines or nets, and 17 percent showed evidence of hooks; 62 percent of turtles assessed exhibited a minimum of one external injury (Archibald and James 2018).

Fisheries bycatch also occur in the Mediterranean and eastern North Atlantic Ocean. Casale *et al.* (2003) analyzed 411 records of leatherback turtles in the Mediterranean, of which 152 were collected from Italy. Most of these records were from fishery captures (n = 170) or found in unknown circumstances (n = 127). Of those reported by fishermen, set or drift nets had the highest number of interactions (29.4 percent), followed by unknown fishing equipment (22.9 percent), longlines (20.6

percent), unspecified nets (12.9 percent), other fishing equipment (9.4 percent), and trawls (4.7 percent). The main fisheries affecting turtles in the Mediterranean (all turtle species, not just leatherback turtles) are Spanish and Italian surface longlines, North Adriatic Italian trawls, Tunisian trawls, Turkish trawls, Moroccan driftnets, and Italian driftnets (Camiñas 2004). The same types of fishing gear from other nations also affect turtles, but the bycatch numbers are lower (Camiñas 2004). Stranding records from Portugal from 1978 to 2013 found that 49 of 275 leatherback turtles exhibited evidence of fishery interactions (the cause of stranding could not be determined in most cases due to decomposition state; Nicolau *et al.* 2016). Multifilament nets accounted for approximately 41 percent of the strandings, followed by monofilament nets, traps/pots, and longlines. Coastal artisanal fisheries were recognized as a particular threat in Portugal.

Based upon these summary reports and stranding assessments, it is clear that fisheries have a large impact on the NW Atlantic DPS. In the following paragraphs, we review information on specific gear interactions, including the following fisheries: gillnet, longline, trawl, pot/trap, and other.

#### Gillnet Fisheries

Gillnet fisheries are common throughout the range of this DPS. Due to the nature of the gear and fishing practices (*e.g.*, relatively long soak times), bycatch in gillnets is among the highest source of direct sea turtle mortality (Upite *et al.* 2013; Wallace *et al.* 2013; Upite *et al.* 2018). Upite *et al.* (2018) evaluated observed fishery interactions and post-interaction mortality and determined a 79 percent sea turtle mortality rate for Northeast and Mid-Atlantic gillnet gear from 2011 to 2015. Wallace *et al.* (2013)

calculated leatherback bycatch in gillnets throughout the NW Atlantic Ocean of 0.015 turtles/set, with a 21 percent median mortality rate (not considering post-interaction mortality). This gear was classified as having a relatively high bycatch impact on the NW Atlantic leatherback population. Small scale fisheries are of particular concern, given the magnitude of bycatch, nearshore distribution, and limited monitoring (Lewison *et al.* 2015). When nets are used in waters off nesting beaches, where leatherback turtles mate, nesting females and mature males are often captured and killed.

The largest documented bycatch of leatherback turtles in gillnet gear occurs off the coast of Trinidad. Lee Lum (2006) estimated that more than 3,000 leatherback turtles were captured by coastal surface gillnets off Trinidad annually, with an approximate 30 percent mortality rate. These captures involved adult turtles, occurring off the north and east coasts of Trinidad during January to August, *i.e.*, the breeding and nesting season, when nesting females and adult males occur in the waters off nesting beaches (Lee Lum 2006). Gilman *et al.* (2010) extrapolated leatherback bycatch estimates (Lee Lum 2006; Gearhart and Eckert 2007) to the entire Trinidad Spanish mackerel and king mackerel surface gillnet fishery, and estimated that almost 7,000 turtles were captured in 2000. Additionally, Eckert *et al.* (2013) worked with drift gillnet fishermen to identify leatherback bycatch hot spots off the north and east coasts of Trinidad (where the nesting beaches are), with capture probability increasing from March to July and a secondary peak in October.

Whereas most of the documented leatherback bycatch off Trinidad occurs in surface drift gillnet fisheries, bottom set gillnet fishing also captures leatherback turtles (Gass 2006; S. Eckert, WIDECASST, pers. comm., 2018). The magnitude of effort and

turtle bycatch in this fishery are lower than for surface nets, but mortality rates are higher (approximately 70 percent; Gass 2006). As such, the bottom set gillnet fishery is thought to have a comparable level of mortality to the drift gillnet fishery (approximately 500 to 1,000 leatherback turtles annually; Gass 2006; S. Eckert, WIDECAST, pers. comm., 2018). The Sea Turtle Recovery Action Plan for the Republic of Trinidad and Tobago noted that drowning in gillnets is that nation's most significant cause of sea turtle mortality (Trinidad and Tobago Forestry Division *et al.* 2010). Bond and James (2017) tracked a female from Canadian waters to a nesting beach off Trinidad, but the turtle was confirmed dead, entangled in coastal fishing gear, just prior to the date of her first predicted nesting event. Venezuelan fishers have also been seen hauling leatherback turtles from Trinidad waters into their boats (Brautigam and Eckert 2006). Together, drift and bottom-set gillnets off the Trinidad beaches, which host the largest nesting aggregation in the DPS, are estimated to kill well over 1,000 leatherback turtles annually, and they thus pose a large threat to the DPS.

High levels of gillnet bycatch occur in other Caribbean and South American nations, also off major nesting beaches. In French Guiana, bycatch was confirmed to be high in the Maroni estuary (Chevalier 2001; Girondot 2015). In 2003, 26 leatherback turtles were caught in coastal gillnets and released off the Cayenne and Montjoly nesting sites (Gratiot *et al.* 2003 in TEWG 2007). Delamare (2005) conducted fishermen interviews and estimated an average of 1,149 leatherback captures in 2004 and 2005 by bottom-set or drifting gillnets in French Guiana. No estimate of mortality was provided, but it is likely similar to Trinidad fisheries, *i.e.*, 70 and 30 percent, respectively. In Suriname, a World Wildlife Fund survey of fishermen estimated leatherback bycatch in

drifting gillnets at 584 in 2006, 174 in 2010, and 424 in 2012 (Madarie 2006; Madarie 2010; Madarie 2012). Most of the turtles were captured alive. In Colombia, 10 to 40 leatherback turtles are killed annually by gillnets (Patiño-Martínez *et al.* 2008). Longline and driftnet gillnet fisheries in Moroccan waters off the northwestern Africa coast capture approximately 100 leatherback turtles annually (Benhardouze *et al.* 2012).

Although not at as high a rate as in the Caribbean (based upon observed interactions), gillnet bycatch occurs in U.S. and Canadian waters. Although South Carolina, Georgia, Florida, Louisiana, and Texas have prohibited gillnets in their State waters, active gillnet fisheries remain in other states and U.S. Federal waters. No cumulative estimates of leatherback bycatch in gillnet fisheries in U.S. waters are available due to the limited observed interactions. However, from 2003 to 2017, fishery observers recorded lethal and non-lethal bycatch in fixed sink, drift sink, and drift floating gillnets throughout the U.S. Atlantic Exclusive Economic Zone (EEZ) and GOM (NMFS unpublished data). From 2012 to 2016, 27 leatherback turtles (coefficient of variation = 0.71, 95 percent CI over all years: 0–68) were bycaught with 21 mortalities in sink gillnet gear in the Georges Bank and Mid-Atlantic regions (Murray 2018). From 1989 to 1998, U.S. drift pelagic gillnets captured 54 leatherback turtles, but that gear is no longer used. Hamelin *et al.* (2017) reviewed leatherback entanglement records reported by Canada in Atlantic Canadian waters between 1998 and 2014. Gillnets, mainly targeting groundfish, were involved in 24 of 205 entanglements (11.7 percent), particularly in Newfoundland and Labrador (n = 15). Often, gillnet entanglements involve the vertical lines associated with gear (M. James, DFO, pers. comm., 2019).

Gillnet bycatch occurs in the eastern North Atlantic Ocean and in the Mediterranean Sea. As in other areas, sea turtles have the potential to interact with set gillnets and drift gillnets. The United Nations (UN) established a worldwide moratorium on drift gillnet fishing effective in 1992; the General Fisheries Commission for the Mediterranean prohibited driftnet fishing in 1997; a total ban on driftnet fishing by the European Union fleet in the Mediterranean went into effect in 2002; and the International Commission for the Conservation of Atlantic Tunas (ICCAT) banned driftnets in 2003. Nevertheless, unregulated driftnetting continued to occur in some areas (*e.g.*, the Mediterranean Sea and off Europe; Pierpoint 2000; Camiñas 2004). In the Atlantic Ocean, leatherback bycatch has been reported from NE Atlantic tuna driftnet fisheries by English, French and Irish vessels (Pierpoint 2000). Of 20 leatherback turtles found in nets in British and Irish waters (1980 to 2000), eight were caught in the NE Atlantic tuna driftnet fishery (with 25 percent mortality) and one was caught in a hake gillnet (Pierpoint 2000).

Historically, driftnet fishing in the Mediterranean Sea caught large numbers of sea turtles. And today an estimated 600 illegal driftnet vessels operate in the Mediterranean, including fleets based in Algeria, France, Italy, Morocco, and Turkey (Environmental Justice Foundation 2007). Out of 411 records of leatherback turtles (stranded, captured, sighted, or found in unknown circumstances) in the Mediterranean Sea, 170 turtles were captured by fishermen, of which 29.4 percent were caught by set or drift nets (Casale *et al.* 2003). Driftnets and gillnets in Greece, Israel, Italy, Tunisia and Turkey have reported documented leatherback interactions, and occasional leatherback bycatch occurs in Croatian artisanal gillnet fisheries (Camiñas 2004; Ergene and Ukar 2017). In particular,

Karaa *et al.* (2013) reviewed 36 leatherback bycatch records from Tunisia fisheries in the Gulf of Gabes, and found that gillnets are the dominant threat to leatherback turtles in the region. A similar result (*e.g.*, gillnets being a high threat to leatherback turtles in the area) was found in the Adriatic Sea (Lazar *et al.* 2012). The first leatherback recorded on the Aegean coast of Turkey was caught in a gillnet (Taskavak *et al.* 1998). Further, a review by Casale (2008) found that leatherback turtles are taken in the drift gillnet fishery in Spain at a rate of 0.065 turtles/day-boat.

Throughout the range of the NW Atlantic DPS, effective gillnet bycatch reduction measures have not been required, but measures to reduce leatherback bycatch have been discussed in some areas (*e.g.*, Trinidad; Eckert 2013). If nations have a closed season for fishing, at least in the nesting season (*e.g.*, Suriname; Madarie 2006), nesting females are afforded some level of protection from gillnet bycatch. Some nations have prohibited gillnet gear; St. Barthelemy does not allow trammel nets in its territorial waters and St. Lucia prohibits fishing within 100 meters of shore to protect nesting turtles. There are gillnet and trammel net restrictions in Curacao (Ministry of Health, Environment, and Nature 2014, UN Environment Programme 2017). In the United States, gillnets with stretched mesh seven inches and larger are prohibited at certain times off North Carolina and Virginia to protect sea turtles (50 C.F.R. 223.206(d)(8); 71 FR 24776, April 26, 2006). While no gear modifications are currently required for U.S. gillnet fisheries, Federal U.S. fisheries are subject to section 7 of the ESA, 16 U.S.C. 1536(a)(2), and through formal consultations on specific fisheries, measures may be required to minimize the impact of incidental take in gillnets (NMFS 2013). Regardless of some of these

protective measures, gillnet bycatch (especially off nesting beaches) results in the loss of thousands of mature individuals annually.

### Longline Fisheries

Leatherback turtles are known to interact with longline fishing gear, most commonly pelagic longlines (Lewison *et al.* 2004; Zollett 2009; Wallace *et al.* 2010; Wallace *et al.* 2013). There is significant concern over the effects of pelagic longline fishing, which extends globally throughout temperate and tropical waters, including several high pressure fishing areas in the North Atlantic Ocean (Fossette *et al.* 2014; Gray and Diaz 2017). In international waters, numerous flag states have high seas longline fisheries that frequently catch leatherback turtles (Lewison *et al.* 2004). Individuals are found entangled and hooked in this gear, mostly by the flippers (Witzell and Cramer 1995; Coelho *et al.* 2015; Huang 2015). Leatherback bycatch in longlines throughout the NW Atlantic Ocean was calculated at 0.062 turtles per set, classifying the gear as a relatively low bycatch impact relative to other sea turtle populations (Wallace *et al.* 2013; Lewison *et al.* 2015). However, because longline fisheries are widespread across leatherbacks' distribution and use millions of hooks each year, they pose a large threat to the NW Atlantic DPS and are estimated to kill thousands of individuals (mature and immature) annually.

Pelagic longline fishing is widespread throughout the range of the DPS and involves a number of nations, so an accurate estimate of total bycatch is difficult to obtain. In the Atlantic Ocean from 2002 to 2013, the largest longline fishing fleets belonged to Taiwan, Japan, Spain, Belize, and China, with the Taiwanese fleet comprising the largest distant-water longline effort throughout the region (Angel 2014;

Huang 2015). In an assessment of the impact of ICCAT fisheries on sea turtles, Gray and Diaz (2017) estimated leatherback interactions with pelagic longlines in the ICCAT area from 2012 to 2014 (15 to 16 fleets). Using a combination of published and assigned sea turtle bycatch rates as a function of estimated fishing effort submitted to ICCAT by its members, Gray and Diaz (2017) found a high degree of overlap in the central North Atlantic Ocean and equatorial waters (some of which are outside this DPS). Within the NW Atlantic region, an estimated 7,138 leatherback interactions occurred in 2012, 6,036 in 2013 and 4,991 in 2014 (Gray and Diaz 2017). Applying a reasonable estimated mortality rate of 21.4 percent, as seen in other high seas pelagic longline gear (Huang 2015), results in an average annual estimated mortality of 1,296 leatherback turtles from 2012 to 2014. However, this is likely an underestimate of total mortality, as the high seas mortality rate in Huang (2015) was based upon the disposition of the turtle when boarded and therefore did not account for post-interaction mortality; 240 of 459 leatherback turtles caught from 2002 to 2013 were alive and 121 were of unknown status (Huang 2015). Angel *et al.* (2014) conducted a risk assessment of turtles from the impacts of tuna fishing in the ICCAT region and found the NW Atlantic RMU (which is comparable to the NW Atlantic DPS; Wallace *et al.* 2010) has high-moderate vulnerability to longline gear, with as many as 270 million longline hooks annually from 2000 to 2009. In particular, Fossette *et al.* (2014) analyzed leatherback satellite tracks (converted to densities) overlaid with longline fishing effort from 1995 to 2009 in the Atlantic Ocean. In the North Atlantic Ocean, a total of four seasonal high-susceptibility areas were identified: one in the central northern Atlantic in international waters, one along the east coast of the United States, and one each in the Canary and Cape Verdean basins (Fossette

*et al.* 2014). These areas partly occurred in the EEZs of eight nations (Cape Verde, Gambia, Guinea Bissau, Mauritania, Senegal, Spain/Canaries, United States, and Western Sahara). Given the species' flexible diving behavior, it is reasonable to expect that turtles are likely to encounter pelagic longlines throughout the Atlantic Ocean, regardless of whether they are engaged in foraging or migratory behavior (Fossette *et al.* 2014).

Bycatch in U.S. Atlantic and GOM pelagic longlines has been extensively studied in the last decade. Current estimates of leatherback interactions with the U.S. Atlantic pelagic longline fishery are lower than previous years. In the late 1990s and early 2000s, estimates of Atlantic U.S. pelagic longline bycatch were around 1000 leatherback turtles annually (NMFS 2001; Yeung 2001; NMFS 2018), with bycatch rates of about 0.15 to 0.5 turtles per 1000 hooks (Watson *et al.* 2005). In 2005, after the United States required pelagic longline gear modifications (50 CFR 635.21), the fleet was estimated to have interacted with 351 leatherback turtles outside experimental fishing operations (Walsh and Garrison 2006). NMFS (2018) estimated 239 leatherback interactions in the U.S. Atlantic pelagic longline fishery in 2011, 596 in 2012, 363 in 2013, 268 in 2014, 299 in 2015, and 339 in 2016. The majority of interactions occurred in the GOM, Mid-Atlantic Bight, Northeast Coastal, and Northeast Distant areas (NMFS 2018). The post-interaction mortality estimate for the most recently available 3-year period (2013 to 2015) for leatherback turtles is 30.13 percent (L. Desfosse, NMFS, pers. comm., 2018). Based on the average leatherback interaction estimate for the entire U.S. pelagic longline fleet from 2011 to 2016 (351), the estimated average annual mortality for the U.S. pelagic longline fishery is 106 leatherback turtles.

Leatherback interactions also occur in Canadian pelagic longline fisheries. From summer to fall, primarily on the Scotian Shelf, encounters with leatherback turtles have been documented in the large pelagic longline fishery since 2001 (DFO 2012). With observer coverage ranging from 5 to 30 percent since 2001, there were 102 reported interactions with pelagic longlines from 2001 to 2005, and 36 from 2006 to 2010 (DFO 2012). Mortality rates are estimated to be in the range of 21 to 49 percent, resulting in an estimated mortality of 13 to 44 leatherback turtles annually. Based on an analysis of Canadian observer data from 2002 to 2010, the bycatch rate in this fishery is estimated to have declined from 120-190 leatherback turtles annually from 2002 to 2006 to 60-90 leatherback turtles annually from 2006 to 2010, largely as a result of gear modifications (Hanke *et al.* 2012).

In the Mediterranean Sea, longlining is prevalent. Drifting longlines targeting swordfish (*Xiphias gladius*), albacore (*Thunnus alalunga*), and bluefin tuna (*T. thynnus*) are considered to be the most dangerous fishing gear for turtles in the Mediterranean Sea (Lucchetti and Sala 2010). Drifting longlines (mainly for albacore tuna) in Spain, Italy, Greece, and Albania have documented leatherback interactions (Camiñas 2004). In the western Mediterranean, swordfish longlines appeared to be responsible for most of the leatherback bycatch and entanglements (Camiñas 1998; Camiñas 2004). Casale *et al.* (2003) reviewed bycatch rates for longline fisheries targeting swordfish and estimated the average Mediterranean longline bycatch rates at 0.0025 leatherback turtles/1000 hooks, with a maximum rate of 0.0510 leatherback turtles/1000 hooks in the Tyrrhenian Sea of Italy (Casale *et al.* 2003; Casale and Margaritoulis 2010). Of 170 leatherback fishery captures in fisheries from the Mediterranean Sea, approximately 35 involved longlines

(Casale *et al.* 2003). While leatherback turtles are encountered in Mediterranean longlines, loggerheads are the most common species caught; only 0.1 percent of turtles captured during an observer program in Spain, Italy and Greece were leatherback turtles (3 out of 2,370 observed turtles; Laurent *et al.* 2001). However, given the extensive longline effort in the Mediterranean Sea (Casale 2008), leatherback bycatch in the Mediterranean is still a concern. Lewison *et al.* (2004) estimated a range of 250 to 10,000 leatherback turtles bycaught in the Mediterranean in 2000, with 6 percent observer coverage.

Longline bycatch of leatherback turtles in the range of the NW Atlantic DPS also occurs in waters off Cape Verde (Melo and Melo 2013; Coelho *et al.* 2015), Morocco (Benhardouze *et al.* 2012), and Brazil (Pacheco *et al.* 2011). Given the wide distribution of both pelagic longline gear and leatherback turtles, bycatch of individuals in longline gear can occur wherever and whenever the gear and sea turtle distribution overlap.

Large circle hooks (non-offset) have been found to reduce leatherback bycatch by as much as 55 percent compared to traditional J-style hooks (Andraka *et al.* 2013; Coelho *et al.* 2015). While the vessels of certain nations may employ large circle hooks, there are no obligations for international longline fleets to adopt such bycatch mitigation measures (Richardson *et al.* 2013). In 2005, an ICCAT resolution encouraged circle hook research (ICCAT 2005), but no legally binding measure to require circle hooks exists (Gilman 2011). Without the widespread use of non-offset circle hooks, it is likely that the high bycatch rates of leatherback turtles in pelagic longline gear will continue throughout the North Atlantic high seas fisheries.

Since 2004, the United States has issued regulations that require modifications to pelagic longline gear in the U.S. Atlantic and GOM to reduce the bycatch and post-interaction mortality of sea turtles; these regulations (50 CFR 635.21(c)(2)) specify hook type and size (18/0 or 16/0 circle hooks depending on the area), bait type, use of turtle disentangling equipment and handling guidelines. Swimmer *et al.* (2017) recently analyzed pelagic longline interactions before (1992 to 2001) and after (mid-2004 to 2015) these regulations were promulgated. Throughout the study period, 844 leatherback turtles were captured. Overall, turtle bycatch was highest in the Northeast Distant statistical reporting area (0.3 turtles/1000 hooks), followed by the Northeast Coastal, GOM, and Caribbean areas. Bycatch rates were higher for years prior to 2004; after the regulations, Atlantic leatherback bycatch rates declined by 40 percent (0.13 to 0.078 turtles/1000 hooks). Within the Northeast Distant area alone, where additional restrictions include a large circle hook (18/0) and limited use of squid bait, rates declined by 64 percent (0.44 to 0.16 turtles/1000 hooks; Swimmer *et al.* 2017). Gilman and Huang (2017) found similar results: fish versus squid bait lowered catch rates of leatherback turtles, and wider circle hooks reduced leatherback catch rates relative to narrower J and tuna hooks. Capture probabilities are lowest when using a combination of circle hook and fish bait.

Efforts have been made to reduce interactions in Canadian waters as well. Circle hook use has been recommended in the swordfish-directed Canadian longline fleet since 2003, whereas corrodible circle hooks have been required in the pelagic longline fishery since 2012 (DFO 2013; C. MacDonald, DFO, pers. comm., 2019). There is no mandatory hook size restriction for the Canadian longline fleet, but license holders almost exclusively use 16/0 circle hooks (C. MacDonald, DFO, pers. comm., 2019). De-hooking

and line-cutting kits are required on swordfish longline fishery vessels (C. MacDonald, DFO, pers. comm. 2019).

Some fishing fleets in the Atlantic Ocean (*e.g.*, U.S., Canadian, ICCAT vessels) use large circle hooks and modified bait, but these measures are not required in all areas (Watson *et al.* 2005; Gilman *et al.* 2007; Gilman 2011). Some nations in the Wider Caribbean Region have implemented circle hook provisions; in Belize, the high seas fishing fleet adopted the use of circle hooks on 10 percent of the fleet and are required to report capture of sea turtles by longlines (Belize Fisheries Department 2017). Because the measures are not widely required, the number of vessels that do not employ bycatch reduction measures is likely higher than the number of vessels that do, and so we conclude on the basis of the best available information that leatherback bycatch in pelagic longline fisheries is still a significant threat (Lewison *et al.* 2015).

Leatherback interactions with bottom longlines also occur. Directed shark fisheries using bottom longlines in the Atlantic Ocean and GOM may capture or entangle leatherback turtles (NMFS 2012), and the GOM reef fishery is also anticipated to take leatherback turtles (NMFS 2011). On February 7, 2007, NMFS published a rule that required commercial shark bottom longline vessels to carry the same dehooking equipment as the pelagic longline vessels; this rule was promulgated to reduce post-interaction mortality (72 FR 5633).

The Canadian east coast groundfish longline fishery targets a wide variety of groundfish species, including cod, haddock, pollock and white hake. Observer coverage has ranged from 2 to 30 percent depending on area, and there have been no reported interactions of leatherback turtles in the observer database since 2001 (DFO 2012).

However, there have been three reports from Quebec logbooks and 10 reports of interactions with groundfish longline gear to non-governmental groups (DFO 2012). This indicates that the risk of interactions in this gear may be higher than documented through the observer program.

Bottom longlines are also used in the Mediterranean Sea (Casale 2008). While there have not been any documented leatherback captures from this gear type, loggerheads have been caught at high rates in Tunisia, Libya, Greece, Turkey, Egypt, Morocco, and Italy (Casale 2008), and interactions with leatherback turtles are possible.

Commercial pelagic longline fisheries do not operate in some Caribbean nations, such as in Panama where effort is limited to vessels under six tons (Executive Decree 486, December 28, 2010). However, other Caribbean nations allow commercial pelagic longline fishing, and many find leatherback turtles with longline hooks (Réserve Naturelle de l'Amana data in Berzins, Office National de la Chasse et de la Faune Sauvage, pers. comm., 2018 and KWATA data in Berzins 2018). While no longlines exist in the Caribbean Dutch nations of Bonaire, St. Eustatius and Saba, there are efforts to introduce circle hooks into the trolling fishery (Ministry of Economic Affairs 2014). We consider longline bycatch to be a widespread threat to this DPS, likely resulting in the loss of thousands of individuals annually.

#### Trawl Fisheries

Leatherback turtles may interact with bottom and midwater trawl gear throughout the North Atlantic Ocean. The highest reported trawl bycatch of leatherback turtles of the NW Atlantic DPS is likely from the southeastern U.S. shrimp trawl fishery. Epperly *et al.* (2003) anticipated an average of 80 leatherback mortalities a year in shrimp trawl

interactions, dropping to an estimate of 26 leatherback mortalities in 2009 due to reduction in fishing effort (Memo from Dr. B. Ponwith, SEFSC, to Dr. R. Crabtree, SERO, January 5, 2011). The 2014 NMFS Southeast U.S. Shrimp Fishery Biological Opinion estimated 167 annual leatherback captures (144 mortalities) in the Atlantic Ocean and GOM shrimp otter trawl fishery, with an additional 34 captures in try nets (single nets testing for shrimp concentrations; NMFS 2014). The majority of these interactions were in the GOM. However, a more recent study of the GOM and southeastern U.S. Atlantic coast shrimp otter trawl fishery found fewer leatherback captures: from 2007 to 2017, only 3 leatherback turtles were reported in the observer data (with coverage levels around 2 percent of nominal days at sea; Babcock *et al.* 2018).

In the mid-Atlantic and northeastern U.S. waters, observers reported 9 leatherback captures in bottom otter trawl gear and 5 captures in midwater trawls from 1993 to 2017 (NMFS unpublished data 2018). In the Wider Caribbean Region, leatherback turtles are reported captured in trawls in French Guiana (Ferraroli *et al.* 2004; TEWG 2007), Guyana (Reichart *et al.* 2003), Suriname (Madarie 2010), Trinidad (Forestry Division *et al.* 2010), and Venezuela (Alio *et al.* 2010).

Since 1980, there were eight reports of leatherback turtles incidentally captured by trawl gear in British and Irish waters (Pierpoint 2000). In the Mediterranean Sea, leatherback bycatch in bottom trawls off Tunisia (Caminas 2004) and Egypt (Casale 2008) has also been reported.

Trawl bycatch reduction measures (*e.g.*, turtle excluder devices (TEDs)) are in place in some nations. The southeastern U.S. shrimp fishery has required TEDs since the early 1990s. However, TEDs that were initially required for use in the U.S. Atlantic

Ocean and GOM shrimp fisheries were less effective for leatherback turtles as compared to smaller, hard-shelled turtle species, because the TED openings were too small to allow leatherback turtles to escape. To address this problem, NMFS issued a final rule on February 21, 2003, to amend the TED regulations (68 FR 8456) to require modified TEDs in the southeastern United States (Atlantic Area and GOM Area) that exclude leatherback turtles, as well as large benthic immature and sexually mature loggerhead and green sea turtles. TEDs are also required in summer flounder trawls operating off Virginia (south of Cape Charles) and North Carolina (64 FR 55860, October 15, 1999; 67 FR 19933, April 17, 2002).

TEDs are also used outside the United States. Shrimp harvested with commercial fishing technology that may adversely affect sea turtles generally cannot be imported into the United States per Public Law 101-162, Section 609(b), enacted on November 21, 1989 (16 U.S.C. 1537 note). The import ban does not apply to nations that have adopted sea turtle protection programs comparable to that of the United States (*i.e.*, require and enforce TED use) or whose fishing activity does not present a threat to sea turtles (*e.g.*, nations fishing in areas where sea turtles do not occur). Although most certifications are done on a national basis, the U.S. State Department guidelines allow some individual shipments of TED-harvested shrimp from uncertified countries with appropriate documentation. Approximately 40 nations are currently certified to import shrimp into the United States, and five fisheries have been determined as having their products eligible for importation with proper documentation (83 FR 22739, May 16, 2018). Specifically, on May 8, 2018, the U.S. State Department certified 13 nations on the basis that their sea turtle protection programs (*e.g.*, use of TEDs) are comparable to that of the

United States: Colombia, Costa Rica, Ecuador, El Salvador, Gabon, Guatemala, Guyana, Honduras, Mexico, Nicaragua, Nigeria, Panama, and Suriname. It also certified 26 shrimp-harvesting nations and one economy as having fishing environments that do not pose a danger to sea turtles. In addition, one fishery from a non-certified nation within the range of the NW Atlantic DPS (the French Guiana domestic trawl fishery) has been authorized to import shrimp products, provided certain documentation accompanies the imports. Sixteen nations have shrimping grounds only in cold waters where the risk of taking sea turtles is negligible: Argentina, Belgium, Canada, Chile, Denmark, Finland, Germany, Iceland, Ireland, the Netherlands, New Zealand, Norway, Russia, Sweden, the United Kingdom, and Uruguay. Ten nations (Bahamas, Belize, China, the Dominican Republic, Fiji, Jamaica, Oman, Peru, Sri Lanka, and Venezuela) and Hong Kong only harvest shrimp using small boats with crews of less than five that use manual rather than mechanical means to retrieve nets or catch shrimp using other methods that do not threaten sea turtles. Use of such small scale technology is not believed to adversely affect sea turtles. For those nations within the geographical range of the NW Atlantic DPS, the threat of shrimp trawling is minimized with TED use.

TEDs are also required in trawl fleets in Trinidad, Belize, Brazil, and Venezuela, but those gear modifications do not currently meet the U.S. certification protocol. On June 20, 2019, the European Union passed a regulation (PE-CONS 59/1/19 Rev 1) that requires technical measures concerning: the taking and landing of marine biological resources; the operation of fishing gear; and the interaction of fishing activities with marine ecosystems. Specific to sea turtles, the regulation requires shrimp trawl fisheries to use a TED in European Union waters of the Indian and West Atlantic Oceans,

consisting of waters around Guadeloupe, French Guiana, Martinique, Mayotte, Réunion and Saint Martin.

TEDs are not required in Mediterranean trawls. Some nations, like Belize, St. Barthelemy, Venezuela (industrial fishing only), and the Caribbean Netherlands (Bonaire, St. Eustatius, Saba), have banned trawling (Bolivarian Republic of Venezuela Official Gazette N° 5.877, March 14, 2008; Ministry of Economic Affairs 2016; Belize Fisheries Department 2017), and Costa Rica does not allow the issuance of any new permits for shrimp trawling (Costa Rica Ministry of Environment and Energy 2017). Curacao prohibits fishing in its territorial waters and inland bays with dragnets (and certain fish traps). These initiatives reduce the impact of trawling on leatherback turtles.

#### Pot/Trap Fisheries

Leatherback turtles are commonly entangled in the vertical lines of pot and trap gear. Entanglements have been mostly reported from U.S. and Canadian waters, but line entanglements have occurred in other areas where similar gear is used (*e.g.*, Britain; Godley *et al.* 1998).

Due to high numbers of entanglement reports, a Sea Turtle Disentanglement Network (STDN) was established by NMFS in the northeastern United States (Maine to Virginia) in 2002. This program relies primarily on reports from the public and subsequent documentation and disentanglement by trained responders. From 2008 to 2017, 267 leatherback entanglements were reported in vertical fishing line (STDN unpublished data). Of those fisheries that could be identified, 79 were lobster, 21 were fish traps or fish lines, 18 were conch (or a combination of conch and lobster), and 5 were crab gear; 144 entanglements were from unidentifiable fishing gear. While most unknown vertical line entanglements likely involve pot/trap gear, this cannot always be

conclusively determined. The majority of the leatherback turtle reports (67 percent) were from Massachusetts waters. Of the 267 leatherback entanglements, 221 were released alive and 46 were found dead.

Given the nature of their injuries, it is probable that not all animals released alive from entanglements survived. Currently there are limited empirical data on leatherback survival from pot/trap entanglements. Innis *et al.* (2010) found that at least some of the disentangled individuals were able to resume normal behavior and migratory patterns, but two leatherback turtles were entangled at least twice, and a third disentangled turtle had significant forelimb skin and muscle injuries. The effects of entanglement may be sub-lethal initially, but could result in subsequent mortality. By assessing the injuries experienced by each turtle that was documented to have been entangled and using NMFS' post-interaction mortality guidance (NMFS 2017), the resulting mortality rate for northeastern U.S. vertical fishing line interactions for all sea turtle species combined was calculated at 55 percent from 2013 to 2017 (NMFS unpublished data). When the mortality estimate includes those turtles that were not disentangled and assumed to have died, the rate increases to 61 percent. As a result (and applying the latest 5 year mortality rate to the last 10 years of entanglement data), 147 to 163 leatherback turtles died from vertical fishing line gear (most of which were likely pot/trap gear) in the northeastern U.S. waters from 2008 to 2017, based on opportunistically reported data. An additional 36 leatherback turtles were reported entangled in trap/buoy lines from North Carolina to Texas from 2008 to 2017 (STSSN unpublished data). Of those 36 entanglements, 32 turtles were found alive and 4 dead, but these southeastern U.S. numbers do not incorporate potential post-interaction mortality so the total lethal interactions were likely

higher. Further, this information is likely an underestimate of actual entanglements and mortality given the opportunistic reporting nature of the program; therefore, it is clear that leatherback interactions with vertical fishing lines are a threat to this DPS.

Entanglements in Canadian waters are also frequently reported under circumstances similar to the U.S. STDN program, *i.e.*, opportunistically by fishermen or the public. Between 1998 and 2014, 205 leatherback entanglements were reported in Canada along the Atlantic coast, with most from Nova Scotia (136) and Newfoundland (40; Hamelin *et al.* 2017). Entanglements mostly involved pot fisheries (44 percent; n = 91), including snow crab (n = 37), inshore lobster (n = 31), rock crab (n = 10), whelk (n = 8), and hagfish (n = 3) fisheries. Trap net fisheries were involved in 26 percent of the entanglements (n = 53). Of the overall 205 reports, the majority of turtles were reported alive and successfully released (n = 174), and the other 15 percent (n = 31) were reported dead in gear. However, the number of dead turtles is likely an underestimate of actual entanglement-associated mortality (Hamelin *et al.* 2017).

Leatherback turtles are also found entangled in vertical fishing lines in European waters. Since 1980, 83 leatherback turtles were bycaught in British and Irish waters, with the method of capture identified in 58 cases (Piedpoint 2000). The majority of captures (n = 36) were rope entanglements, usually buoy lines used in pot fisheries for crustaceans or whelk, with a 61 percent recorded mortality (Pierpoint 2000).

Some types of aquaculture use vertical lines similar to pot/traps and may pose an entanglement risk (Price *et al.* 2017). Four leatherback turtles (two alive, two dead) in Canadian and U.S. waters have been opportunistically reported in aquaculture gear to date (Price *et al.* 2017). However, as this industry is anticipated to grow in the near

future, leatherback interactions with aquaculture lines, and subsequent injury or mortality, may increase.

These data comprise the best available information on pot/trap fishery interactions with the NW Atlantic DPS. However, due to the high probability of underreporting leatherback turtle entanglements by fishers, the ad hoc nature of public reporting, and the uncertainty about post-release survivorship, the leatherback mortality rate due to entanglements in vertical lines is likely underestimated (Hamelin *et al.* 2017). Estimates indicate that approximately 622,000 vertical lines are deployed from fishing gear in U.S. waters from Georgia to the Gulf of Maine (Hayes *et al.* 2018). There are currently no existing mitigation measures to reduce leatherback bycatch in vertical fishing lines, but efforts to reduce the amount of vertical lines in the water to assist with large whale conservation in the United States may help reduce the impact to the DPS (<https://www.greateratlantic.fisheries.noaa.gov/protected/whaletrp/>).

#### Other Gear Types

Leatherback turtles are also susceptible to bycatch in pound nets, weirs, and purse seine fisheries. In the United States, pound nets set in Virginia waters have entangled leatherback turtles. On June 23, 2006, NMFS issued a regulation (71 FR 36024) requiring offshore pound nets set in a portion of the lower Chesapeake Bay from May 6 through July 15 of each year to use modified pound net leaders, a gear modification consisting of vertical hard lay lines spaced at least two feet apart on the top portion of the leader, and eight inch or smaller stretched mesh on the bottom portion of the leader. From 2013 to 2017, 16 leatherback turtles have been found entangled in the hard lay lines of the leaders, of which two were dead (NMFS 2018). While individuals may continue to be

entangled in modified pound net leaders, the impact of the pound net fishery on the NW Atlantic DPS is likely minor given the few nets set in the lower Chesapeake Bay using this gear (approximately four to six) and the frequency of live interactions. From 2008 to 2017, the STDN also documented leatherback captures in weirs set off Massachusetts; these turtles were found alive, either entangled in the netting (n = 2) or free swimming in the weir (n = 4).

Purse seines are used to catch a variety of fish species and are commonly used in the ICCAT area to catch tuna (Angel *et al.* 2014). Leatherback captures have occurred in Atlantic purse seine fisheries, and this bycatch may have a minor impact on the DPS. In British and Irish waters, two leatherback turtles were reported to be captured in purse seine gear between 1980 and 2000 (Pierpont 2000). Clermont *et al.* (2012) reported a total capture of 67 leatherback turtles in more than 9000 observed Atlantic purse seine sets between 1995 and 2011, with only four found dead (representing 10 percent observer coverage). Most of the interactions were adults (75 percent). However, not all of the purse seine effort reported by Clermont occurs in the NW Atlantic DPS range. Thus, purse seine interactions with this DPS may be a fraction of the total captures reported. For those purse seines in the ICCAT region using fish aggregating devices and for those setting over free-swimming tuna schools, the effort (through 2011) was concentrated in the tropics, off West Africa between Namibia and Mauritania and off Venezuela (Clermont *et al.* 2012; Angel *et al.* 2014). While leatherback and purse seine interactions may occur where distribution and effort overlap, the magnitude of the purse seine impacts on the NW Atlantic DPS is lower than the bycatch values presented in Clermont *et al.* (2012). Further, Angel (2014) found that the direct impacts on turtles from purse seine

fishing operations appears to be minor in comparison to the impacts from longline fishing, especially as most purse seine captures are released alive.

#### Summary of Fisheries Bycatch

We conclude that most immature and adult leatherback turtles of this DPS are exposed to bycatch in multiple fisheries throughout their range. Bycatch in gillnet fisheries, in particular, is a major threat with high mortality rates (Lee Lum 2006; Gilman *et al.* 2010; Girondot 2015), annually killing thousands of NW Atlantic leatherback turtles. When set off nesting beaches, gillnets result in high mortality of nesting females and mature males (Lee Lum 2006; Eckert 2013). Longline bycatch is considered to be a widespread threat throughout the DPS and a primary source of leatherback mortality (Lewison *et al.* 2004), resulting in the death of thousands of leatherback turtles annually. In general, bycatch mortality reduces abundance by removing individuals from the population. When nesting females are killed, it also reduces productivity. We conclude that fisheries bycatch is the primary threat to the NW Atlantic DPS.

#### *Vessel Strikes*

Vessel strikes are a threat to the NW Atlantic DPS. Injuries from vessel strikes may include blunt force trauma and propeller parallel slicing wounds affecting the carapace, flippers, head, and/or underlying organs (Work *et al.* 2010). Most of what is known about vessel strikes comes from stranding records; the most extensive stranding network is found in the United States: the Sea Turtle Stranding and Salvage Network (STSSN). In the United States (Maine through Texas), 957 leatherback turtles were reported stranded, captured, or entangled from 2008 to 2017, and of those, 204 had probable vessel-related injuries (STSSN unpublished data). For example, at least 72

leatherback turtles stranded in Massachusetts with vessel strike wounds between 2006 and 2018, including at least three adult females that had previously been documented nesting in the Caribbean (Dourdeville *et al.* 2018; Mass Audubon Wellfleet Bay Wildlife Sanctuary, unpublished data, 2019). It is sometimes difficult to determine whether the vessel related wounds occurred before or after the turtle died (Stacy *et al.* 2015).

However, a recent study estimated that approximately 93 percent of Florida stranded turtles with vessel strike wounds were killed by those injuries (Foley *et al.* 2019). Based on the best available information, it is reasonable to conclude that approximately 190 leatherback turtles were killed as a result of vessel strikes in U.S. Atlantic and GOM waters from 2008 to 2017. This number is likely an underestimate as strandings represent a small percentage of turtles that are injured or die at sea, and many vessel strikes are not reported, detected, or recovered.

Vessel strikes have been documented in other nations as well, including in Portugal (Nicolau *et al.* 2016), Britain (Godley *et al.* 1998), and off the coast of Tunisia in the Strait of Sicily (Karaa *et al.* 2013; Caracappa *et al.* 2017). While there is very limited observational information on vessel collisions in Atlantic waters of Canada, there has been at least one recorded vessel strike (DFO 2012). More recently, an injury assessment of leatherback turtles (n = 228) on Atlantic Canada foraging grounds and on a Trinidad nesting beach found only 1.3 percent of turtles exhibited injuries consistent with vessel strikes (Archibald and James 2018). However, this low injury rate may indicate that there is low survivorship of vessel strikes. Females with carapace damage from propellers have been also observed on Costa Rican nesting beaches (de Haro *et al.* 2006).

Leatherback behavior data can help predict the potential for vessel strikes. Based on telemetry data for leatherback turtles (n = 15) on the northeastern U.S. shelf, leatherback turtles spent over 60 percent of their time in the top 10 m of the water column and over 70 percent of their time in the top 15 m (Dodge *et al.* 2014). Additional turtle-borne camera and autonomous underwater vehicle research in the waters off Massachusetts suggests that turtles surface frequently and engage in subsurface swimming (within the top 2 m) when occupying shallow, well-mixed, coastal environments, increasing the probability of a vessel strike (Dodge *et al.* 2018). Based on 24 free swimming leatherback turtles tagged in Canadian waters from 2008 to 2013, Wallace *et al.* (2015) found these leatherback turtles primarily occupied the upper 30 m of the water column and had shallow 4 to 6 minute dives. Given most leatherback activity occurs in the top 15 to 30 meters of the water column in temperate shelf waters of the NW Atlantic Ocean and vessel traffic is high along the U.S. East coast, the risk of vessel strikes is likely higher than the documented interactions would suggest (DFO 2012; Hamelin *et al.* 2014).

While observational data are limited, it is reasonable to conclude that, based upon the best available information, mortality due to vessel strikes may occur wherever vessel traffic and leatherback distribution (juvenile and adult) overlap. The impact is likely minimized in areas with less frequent vessel traffic (*e.g.*, less developed areas) and decreased leatherback turtle presence. Nesting females and mature males may be especially vulnerable to vessel strikes because they occur in the waters off nesting beaches, which are coastal areas where vessel traffic is more prevalent. Vessel strikes affect the NW Atlantic DPS by lowering abundance (if the interaction results in

mortality) and affecting future reproductive potential (productivity) when nesting females are killed. We conclude that vessel strikes pose a threat to the NW Atlantic DPS.

### *Pollution*

Pollution includes contaminants, marine debris, and ghost fishing gear. The detection of pollution impacts on leatherback turtles is opportunistic and thus likely underestimated. While plastic ingestion is not always fatal, it can reduce ability to feed, affect swimming behavior and buoyancy control, potentially lead to chemical contamination and chronic effects, and weaken physical condition, which could impair the ability to avoid predators and survive threats (Nelms *et al.* 2016). Entanglement in marine debris results in injuries that can reduce fitness, cause eventual death, reduce ability to avoid predators, reduce ability to forage and/or swim efficiently due to drag, and lead to starvation or drowning (Nelms *et al.* 2016). Pollution on the beach and in the water occurs throughout the range of the NW Atlantic DPS.

Dow *et al.* (2007) defined marine pollution as agriculture, petroleum, sewage, industrial runoff, vessel discharges, declining water quality, and marine debris. They found pollution in the marine environment to be among the greatest threats to all sea turtle species in the Wider Caribbean Region. Dow *et al.* (2007) defined beach pollution as agriculture, petroleum/tar, sewage, industrial runoff, and beach litter/debris; they found pollution on the beach to be a threat. Pollution on the beach and in the water occurs throughout the range of the NW Atlantic DPS.

Leatherback turtles are susceptible to adverse effects from pollution. Marine pollution, including direct contamination and structural habitat degradation, can also affect leatherback habitat. In particular, the Mediterranean is an enclosed sea, so organic

and inorganic wastes, toxic effluents, and other pollutants rapidly affect the ecosystem (Camiñas 2004).

Of particular concern, due to their immune, reproductive, and endocrine disrupting nature, are persistent organic pollutants (POPs), such as polychlorinated biphenyls (PCBs), polybrominated diphenyl ethers (PBDEs), and pesticides (Bergeron *et al.* 1994; Bishop *et al.* 1991, 1998; Keller *et al.* 2004). These chemicals have been identified in both adults and eggs in several areas occupied by this DPS. Guirlet *et al.* (2010) measured maternal transfer of organochlorine contaminants (OCs) from 38 nesting females in French Guiana. PCBs were found to be the dominant OC, followed by pesticides, but OC concentrations were lower than concentrations measured in other marine turtles (potentially due to the lower trophic level diet and offshore foraging areas). All OCs detected in nesting adults were detected in eggs, suggesting a maternal transfer of OCs. In French Guiana, hatching success has been shown to be low when OCs are present in the sand (most likely originating from pesticide use in plantations and malaria prophylaxis (Guirlet 2005). However, a link between OCs and embryonic mortality could not be determined (Guirlet *et al.* 2010). Stewart *et al.* (2011) also recorded PCB, OC, and PBDE concentrations for nesting and stranded leatherback turtles in the southeastern United States. Their results also suggested maternal transfer of POPs in leatherback turtles, but Stewart *et al.* (2011) found higher levels of PCBs and pesticides than those found in French Guiana (Guirlet *et al.* 2010). While finding that leatherback contaminant concentrations were substantially lower than concentrations in other reptile studies that demonstrated toxic effects, Stewart *et al.* (2011) suggested that sub-lethal effects (especially on hatchling body condition and health) may nevertheless be occurring in this

species. De Andres *et al.* (2016) similarly monitored PCB and PBDE concentrations in eggs laid in Costa Rica (18 nests). POP levels were similar to those reported in French Guiana nesting females (Guirlet *et al.* 2010) and slightly lower than those in Florida (Stewart *et al.* 2011). Further, De Andres *et al.* (2016) found a significant negative relationship between PBDE levels and hatching success, suggesting potential harmful effects of these contaminants on leatherback reproduction. OCs (and mercury) have also been documented in turtles that stranded in the United Kingdom (Godley *et al.* 1998). A leatherback that stranded off the coast of Wales, U.K. was found with PCB levels one-to-three orders of magnitude higher than the lowest levels reported for fish taken in the North Atlantic, but similar to the lowest concentrations reported from oceanic cetaceans (Davenport *et al.* 1990). Even with the recent restriction of the use of POPs, due to the widespread persistent nature of these chemicals and continuing atmospheric deposition (Ross *et al.* 2009) it is probable that similar chemical concentrations occur in other areas of this DPS.

Other contaminants have also been documented in leatherback turtles and their eggs. Heavy metals (*e.g.*, arsenic, cadmium, chromium, mercury, lead, etc.) enter the environment from a variety of sources (Guirlet *et al.* 2008; Perrault 2012). In particular, mercury can affect a variety of functional processes in wildlife, including the nervous, excretory and reproductive systems (Wolfe *et al.* 1998). Mercury, cadmium, and lead were recorded in nesting females (n = 46) and eggs in French Guiana (Guirlet *et al.* 2008). Maternal transfer of all three elements was documented, and female lead levels increased throughout the nesting season (Guirlet *et al.* 2008). This could be explained, in part, by external contamination via ingestion of contaminated prey or polluted water

during nesting, as the French Guiana coast environment is exposed to significant environmental pollution via anthropogenic and natural sources. While mercury concentrations were lower than values reported for other sea turtle species, cadmium levels documented in French Guiana were at the same level shown to impact gonadal development in other turtle species and may impact reproductive processes and lower fertility (Guirlet *et al.* 2008). In Massachusetts, entangled turtles had significantly higher blood lead concentrations than directly captured turtles (Innis *et al.* 2010). While similar to those reported in French Guiana (Guirlet *et al.* 2008), blood concentrations of mercury and cadmium were at levels high enough to induce carcinogenic, teratogenic, and toxic effects in a variety of species (Innis *et al.* 2010).

Mercury and selenium have also been recorded in nesting females and eggs in Florida and St. Croix. Animals persistently exposed to mercury can experience selenium deficiency, which is of concern because selenium is important to hatching and emergence success (Perrault *et al.* 2011). However, high levels of selenium can be toxic and negatively impact hatching success (Perrault *et al.* 2013). Mercury concentrations in nesting females from Florida were found to be higher than in St. Croix, which could be a result of different migratory and foraging areas, whereas hatchling blood mercury values were higher in St. Croix (Perrault *et al.* 2011; Perrault *et al.* 2013). It is interesting to note that in St. Croix, no correlations were found between mercury or selenium concentrations and hatching or emergence success, which is different from results in Florida (Perrault *et al.* 2011; Perrault *et al.* 2013). Hazard quotient results by Perrault *et al.* (2013, 2014) imply that mercury and selenium levels could pose a threat to leatherback turtle reproductive success and/or hatchling health and survival. Leatherback hatching and

emergence success rates are already low compared to other species of sea turtles (Bell *et al.* 2004; Perrault *et al.* 2011), so the impacts of pollution and contamination on hatching success is a notable concern. In addition, mercury was found to be higher in adults than juveniles/sub-adults stranded along the U.S. Atlantic coast, suggesting potential physiological concerns due to accumulation and ongoing inputs into the environment (Perrault *et al.* 2012). It is clear that additional long-term research is needed to better understand the relationship of non-essential elements in turtle development and reproduction.

Marine debris (most notably plastic pollution) is a threat throughout the range of the NW Atlantic DPS (Girondot 2015). Several global reviews have outlined the persistent and widespread nature of the issue, both as an ingestion and an entanglement threat (Mrosovsky *et al.* 2009; Schuyler *et al.* 2014; Nelms *et al.* 2016; Lynch 2018). Law *et al.* (2010) assessed plastic content at the surface of the western North Atlantic Ocean and Caribbean Sea from 1986 to 2008, and found the highest concentration of plastic debris was observed in subtropical latitudes and associated with large-scale convergence zones, which include foraging areas targeted by leatherback turtles.

Ingestion of marine debris is a concern for leatherback turtles, especially given the similarity of their preferred prey (*e.g.*, gelatinous zooplankton) to some plastics. In particular, plastic bags appear similar to jellyfish in the marine environment, leading to mistaken and inappropriate triggering of the sensory cue to feed (Schuyler *et al.* 2014; Nelms *et al.* 2016). While plastic ingestion is not always fatal, it can reduce ability to feed, affect swimming behavior and buoyancy control, potentially lead to chemical

contamination and chronic effects, and weaken physical condition, which could impair the ability to avoid predators and survive threats (Nelms *et al.* 2016).

Marine debris ingestion can occur in any location, but given the enclosed nature of the sea and intense human pressure, the Mediterranean Sea in particular is a hot spot for plastic marine debris and other pollutants (Camiñas 2004; Cozar *et al.* 2015). Marine debris ingestion has been documented from leatherback turtles stranded in Tunisia (Karaa *et al.* 2013), Israel (Levy *et al.* 2005), the northern Adriatic Sea (Poppi *et al.* 2012), and the Strait of Sicily (Caracappa *et al.* 2017). Of particular note, 30 to 73 percent of turtles stranded in the Bay of Biscay (France) were found to have ingested plastic annually from 1979 to 1999 (out of 87 leatherback turtles necropsied; Duguay *et al.* 2000). The seasonal rate of ingestion was inversely related to the abundance of jellyfish, leading the authors to propose that the depletion of jellyfish led to debris ingestion as potential prey. Cozar *et al.* (2015) conclude that the effects of plastic pollution on marine life are anticipated to be frequent in the high plastic-accumulation region of the Mediterranean Sea.

In U.S. waters, marine debris ingestion has also been documented in stranded leatherback turtles. However, ingestion does not always cause mortality and is typically an incidental finding. Of 41 leatherback turtles necropsied from North Carolina to Texas between 2008 and 2017, 17 had ingested plastics or marine debris (STSSN unpublished data 2018). From Maine to Virginia during that same time period, 10 necropsies detected ingestion, but the total number of necropsied turtles, out of the 677 strandings in the region, is currently unknown. It is likely that many more stranded turtles ingested some level of marine debris (STSSN unpublished data 2018). Out of 33 leatherback turtles examined in New York Bight (an area with dense population), 30 percent had synthetic

material ingestion, mostly consisting of thin, clear plastic (Sadove *et al.* 1989). Of two leatherback turtles stranded in North Carolina during 2017 whose gastrointestinal tracts were analyzed, microplastics were present in both (Duncan *et al.* 2018).

Marine debris ingestion is not limited to microplastics or plastic bags. Off the northeastern U.S. coast, necropsies of disentangled leatherback turtles that have died post-release have documented considerably large pieces of plastic (*e.g.*, 83 by 35 cm) in their stomachs (Innis *et al.* 2010). These numbers likely underestimate the true marine debris ingestion rate because many turtles likely ingest marine debris and do not strand.

Leatherback turtles can also become entangled in marine debris. From 2008 to 2017, the Northeast U.S. STDN documented 24 entanglements from miscellaneous sources not attributed to obvious fisheries entanglements, as described above (STDN unpublished data). These unknown entanglements could involve a myriad of sources but are considered as entangling marine debris. The Sea Turtle Recovery Action Plan for the Republic of Trinidad and Tobago noted that entanglement in lost or abandoned fishing gear (primarily nets) poses a threat to leatherback turtles in the marine and terrestrial environment (Forestry Division *et al.* 2010).

Marine debris is also a problem on nesting beaches and can reduce nesting success. Pollution and debris often are deposited on high energy beaches, which are also the preferred nesting habitat of leatherback turtles (TEWG 2007). Coastal and inland littering (which can ultimately reach the sea) is a problem throughout Trinidad and Tobago, and ocean borne debris is particularly prevalent on the east and north coasts, which host the main leatherback nesting beaches (Trinidad and Tobago Forestry Division *et al.* 2010). Extensive debris on nesting beaches is not uncommon throughout the

Caribbean, often carried by rivers to the sea and later washed ashore (*e.g.*, in Costa Rica; Chacón-Chaverri and Eckert 2007). Debris on nesting beaches may impede females during the nest-site selection stage, limit and degrade the amount of habitat available, and/or result in aborted nesting attempts (Chacón-Chaverri and Eckert 2007). If line or netting is encountered on nesting beaches, entanglement of nesting females and hatchlings is also a risk.

The majority of the NW Atlantic DPS is exposed to pollution throughout all life stages. These threats are a result of the developed nature of many of the nations within the range of the DPS. The degree of impact is difficult to quantify, especially given the widespread nature of pollution and the diverse types of impacts. Contaminants may affect this DPS by reducing productivity, if hatching success is lowered, and by lowering abundance, if contamination results in mortality. Marine debris affects the DPS by lowering abundance, when it causes death through ingestion or entanglement, and reducing productivity, when hatchlings and nesting females are affected. While, we do not have quantitative estimates of the number of individuals that are killed or injured as a result of pollution, we conclude that it is prevalent throughout the range of the DPS and constitutes a threat to the NW Atlantic DPS.

#### *Oil and Gas Exploration*

Oil and gas activities have the potential to impact the NW Atlantic DPS directly (*e.g.*, exposure to oil following oil spills) and indirectly (*e.g.*, increased probability of vessel strikes and habitat degradation/destruction). In addition to lethal effects, sublethal effects may occur and include displacement from primary foraging areas with accompanying energy costs (TEWG 2007).

Several areas within the range of the NW Atlantic DPS have intense oil and gas development and exploration close to major nesting beaches. The potential for oil spills is of particular concern in the Wider Caribbean Region due to its effect on all life stages in the marine environment. The biggest oil producing nations in South America are Brazil, Mexico, Venezuela, and Colombia. Although only three Caribbean nations currently have exportable oil and natural gas reserves (Barbados, Cuba, and Trinidad and Tobago, with Trinidad and Tobago the only significant exporter), in 2017, a major oil field was discovered off Guyana, which will likely lead to extensive new development and extraction. As a result, marine traffic is likely to increase in the area as well as the possibility for oil spills. In Panama, contamination from oil spills, primarily in area of the Trans-Isthmus oil pipeline and the Panama Canal, is of particular concern (Bräutigam and Eckert 2006; Ruiz *et al.* 2006). Some Caribbean nations (*e.g.*, Belize, French Guiana) have permanent moratoria on oil and gas exploration in offshore waters.

In the United States, oil and gas extraction primarily occurs in the GOM (BOEM 2016; BOEM 2017), an area with leatherback foraging and migratory habitat (Aleksa *et al.* 2018). Increased shipping traffic and marine noise due to oil and gas explorations in the GOM pose a direct threat for leatherback turtles in foraging grounds and migratory routes, due to the potential for vessel strikes and harassment (Wallace *et al.* 2017; Ward 2017). Oil spills regularly occur in the GOM, from small amounts of varying types of oil product to large catastrophic spills. In 2010, a major oil spill occurred in the north-central GOM, affecting important foraging habitat used by leatherback turtles (Deepwater Horizon NRDA Trustees 2016). Evans *et al.* (2012) tracked a post-nesting leatherback from Chiriqui Beach, Panama, into the GOM during the Deepwater Horizon oil spill. The

track followed similar tracks from turtles in previous years and did not seem to change once entering areas with visible oil slicks (on two occasions). Injuries to leatherback turtles caused by the GOM Deepwater Horizon oil spill could not be quantified (Deepwater Horizon NRDA Trustees 2016). However, given that the GOM is important habitat for leatherback turtles (Aleksa *et al.* 2018) and leatherback turtles were documented in the Deepwater Horizon oil spill zone during the oil spill period, the Deepwater Horizon NRDA Trustees (2016) concluded that leatherback turtles were exposed to Deepwater Horizon oil, and some portion of those exposed likely died.

In Atlantic Canada, impacts from oil and gas may also occur. Several petroleum production projects occur offshore of Nova Scotia (<https://www.cnsopb.ns.ca/offshore-activity/offshore-projects>). Howard (2012) determined that oil pollution from coastal refineries, ships, small engine vessels, and oil and gas exploration and production is a risk to leatherback survival in Canada. There are also offshore oil and gas platforms in the North (United Kingdom, Denmark) and Mediterranean Seas, where similar impacts to leatherback turtles may also occur (EU Offshore Authorities Group 2018; <https://euoag.jrc.ec.europa.eu/node/63>). In particular, the Mediterranean Sea has been declared a “special area” by the International Convention for the Prevention of Pollution from Ships (MARPOL), in which deliberate petroleum discharges from vessels are banned, but numerous repeated offenses are still thought to occur (Pavlakis *et al.* 1996). Some estimates of the amount of oil released into the region is as high as 1,200,000 metric tons (Alpers 1993). Direct oil spill events also occur, as in Lebanon in 2006 when 10,000 to 15,000 tons of heavy fuel oil spilled into the eastern Mediterranean (UN Environment Programme 2007).

In summary, oil and gas activities are prevalent in foraging, migratory, and offshore nesting habitats of the NW Atlantic DPS, potentially exposing all life stages to oil associated threats, such as direct mixing in oil, oil ingestion, vessel strikes, and nesting beach contamination. Oil and gas activities have the potential to affect this DPS by reducing productivity (*e.g.*, if hatching success is reduced by oil spills) and potentially lowering abundance (*e.g.*, if oil exposure results in mortality). As such, oil and gas activities are a threat to the NW Atlantic DPS.

#### *Natural Disasters*

Natural disasters, such as hurricanes and other storms, and natural phenomena, such as *Sargassum* events on or near nesting beaches, pose a threat to the NW Atlantic DPS.

Hurricanes are common in the Caribbean and southeastern United States. Hurricanes and tropical storms impact nesting beaches by increasing erosion and sand loss and depositing large amounts of debris. In 2017, Hurricane Maria devastated the islands of Dominica, St. Croix, and Puerto Rico, and even though the nesting season was nearly over, many beaches were impacted, including Maunabo, Puerto Rico (one of the most abundant nesting beaches on the island; R. Espinoza, Conservación ConCiencia, pers. comm., 2017). Dewald and Pike (2014) found that a lower level of leatherback nesting attempts occurred on sites that were more likely to be impacted by hurricanes. These types of storm events may ultimately affect the amount of suitable nesting beach habitat, potentially resulting in reduced productivity, especially as leatherback turtles typically nest on high energy beaches (TEWG 2007).

Hurricanes may also result in egg loss by destroying and inundating nests. However, hurricanes are usually aperiodic so the impacts are expected to be infrequent. Hurricanes also typically occur after the peak of the leatherback hatching season and would not be expected to affect the majority of incubating nests (USFWS 1999). That said, according to the Intergovernmental Panel on Climate Change (IPCC), climate change may be increasing the frequency and patterns of hurricanes (IPCC 2014) potentially causing such impacts to nests to become more common in the future.

*Sargassum* is a genus of macroalgae found in temperate and tropical waters. When large amounts of *Sargassum* wash ashore, they form thick mats that have the potential to disrupt females' nesting activities and impede hatchlings' access to the ocean (Maurer *et al.* 2015). In 2011 and 2015, large amounts of *Sargassum* were present in the Caribbean (mainly Trinidad and Tobago and Grenada) and frequently washed ashore, covering large expanses of sandy shoreline on nesting beaches. While females still nested in these areas, hatchlings needed intervention to reach the ocean (Wang and Hu 2016; Audroing, TVT, pers. comm., 2018; K. Charles, Ocean Spirits Inc., pers. comm., 2018). Most recently, large amounts of *Sargassum* were found in 2018 on Caribbean beaches, causing Barbados to declare a national emergency in June 2018. Such widespread blanketing of *Sargassum* on leatherback nesting beaches throughout the Caribbean has the potential to impact future hatching success and survival.

In summary, natural disasters and phenomena have the potential to impact the NW Atlantic DPS. However, given the infrequent and temporary nature of the occurrences, only a small proportion of eggs, hatchlings, and nesting females are exposed to these threats. Impacts include egg and hatchling mortality that affect productivity of

the DPS. Seasonal losses at individual beaches may be large, but we do not expect such impacts to be spatially or temporally widespread. However, we conclude that natural disasters pose a threat to the DPS.

### *Climate Change*

Climate change is a threat to the NW Atlantic DPS. The impacts of climate change include increases in temperatures (air, sand, and sea surface); sea level rise; increased coastal erosion; more frequent and intense storm events; and changes in ocean currents. These impacts may affect leatherbacks through alterations of the incubation environment, reduction of nesting habitat, and changes in prey as described in the following subsections.

Modeling results show that global warming (rise in average surface temperature) poses a “slight risk” to females nesting in French Guiana and Suriname relative to those nesting in Gabon, Congo, and West Papua (Dudley *et al.* 2016). As global temperatures continue to increase, some beaches will experience changes in sand temperatures, which in turn will alter the thermal regime of incubating nests. Changing sand temperatures at nesting beaches may result in changing sex ratios of hatchling cohorts and reduced hatching output (Hawkes *et al.* 2009). Leatherback turtles exhibit temperature-dependent sex determination (Binckley and Spotila 2015) and warmer temperatures produce more female embryos (Mrosovsky *et al.* 1984; Hawkes *et al.* 2007). In the NW Atlantic DPS, the pivotal temperature (the temperature at which a sex ratio of 1:1 is produced) is estimated to be between 29.25°C and 30.5°C (Eckert *et al.* 2012) but there are variations in measurements (Girondot *et al.* 2018), over time, and among locations. An increase over that temperature would result in more female hatchlings. Such increases in female

hatchling output have already been documented (Patiño-Martínez *et al.* 2012), and with an increase in temperatures from climate change, these trends are likely to continue if other nesting factors remain constant. For example, Patiño-Martínez *et al.* (2012) developed a model to relate measured incubation temperature to sex ratio and estimated that females nesting at Caribbean Colombian beaches currently produce approximately 92 percent female hatchlings. Under all future climate change scenarios, complete feminization could occur as soon as 2021 (Patiño-Martínez *et al.* 2012). In St. Eustatius, leatherback hatchling production was female biased from 2002 to 2012, with less than approximately 24 percent of males produced every year (Laloë *et al.* 2016). Future warming air temperatures will exacerbate this female bias, and female leatherback sex ratios are projected to consistently reach 95 percent after 2028 on that island, which has dark and light sand beaches (Laloë *et al.* 2016). Warming trends in Costa Rica are expected to be higher than the global average and resulting female-biased sex ratios are also expected (Gledhill 2007). While the assumption is that most nesting beaches will become female-biased due to increased sand temperatures, this may not be the case in all areas. In Grenada, increased rainfall (another effect of climate change) was found to have a cooling influence on nests, so that more male producing temperatures (less than 29.75°C) were found within the clutches (Houghton *et al.* 2007). Further, due to the tendency of nesting females to deposit some clutches in the cooler intertidal zone of beaches, the effects of long-term climate on sex ratios may be mitigated (Kamel and Mrosovsky 2004; Patiño-Martínez *et al.* 2012).

Hatching success is affected by warming temperatures. Extremely high sand/nest temperatures are anticipated to result in embryonic mortality (Gledhill 2007, Santidrián

Tomillo *et al.* 2012, Valentin-Gamazo *et al.* 2018). In Costa Rica, warmer conditions can exacerbate the effects of biotic contamination and mold infestations of developing embryos (Gledhill 2007), resulting in reduced hatching success.

Temperature increases are likely to be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions that reduce productivity (Patiño-Martínez *et al.* 2014; Santidrián Tomillo *et al.* 2015). These impacts may affect nests in different ways, but the result (*e.g.*, reduced hatching output) is similar. Very wet conditions may inundate nests or increase fungal and mold growth, reducing hatching success (Patiño-Martínez *et al.* 2014). Very dry conditions may affect embryonic development and decrease hatchling output. Under climate change scenarios, very dry conditions are expected for St. Croix, an area already showing decreased productivity and reduced first time nesting female abundance (Santidrián Tomillo *et al.* 2015; Garner *et al.* 2017). Santidrián Tomillo *et al.* (2015) assessed climatic conditions on hatchling output at four nesting sites (Sandy Point, St. Croix; Pacuare, Caribbean Costa Rica; Playa Grande, Pacific Costa Rica; Maputaland, South Africa), and found that St. Croix had the highest projected warming rate (+ 5.4°C), highest absolute temperature and lowest precipitation levels. With these further increases in dryness and air temperatures, hatchling productivity is expected to be compromised by the end of the 21st century in this area (Santidrián Tomillo *et al.* 2015). Santidrián Tomillo *et al.* (2015) suggested that the lack of rain is what reduces developmental success and hatchling emergence. However, Rafferty *et al.* (2017) evaluated long-term climate data for St. Croix, using climate data collected from a nearby weather station, and

found no significant trend in incubation temperatures or precipitation that could be associated with observed decreases in productivity at this location.

Finally, incubation temperatures can also influence hatchling morphology and locomotion (Mickelson and Downie 2010). Leatherback hatchlings originating from nests incubated at lower temperatures exhibited carapace and front flipper length-width ratios that significantly improved their crawling speeds relative to those hatchlings incubated at high temperatures (Mickelson and Downie 2010).

Sea level rise is another threat to leatherback turtles. Thornalley *et al.* (2018) found that the Labrador Sea deep convection and the Atlantic Meridional Overturning Circulation, a system of ocean currents in the North Atlantic, have been unusually weak over the past 150 years or so, and this weakened state may have modified northward ocean heat transport, as well as atmospheric warming by altering ocean-atmosphere heat transfer. Further, the documented weakening of this system is related to above-average sea level rise along the U.S. East Coast (Caesar *et al.* 2018). Sea level rise may result in intensified erosion and loss of nesting beach habitat (Fish *et al.* 2005; Fuentes *et al.* 2010; Fonseca *et al.* 2013). In Bonaire, up to 32 percent of the current beach area could be lost with a 0.5 m rise in sea level, with lower, narrower beaches being the most vulnerable (Fish *et al.* 2005). Ussa (2013) predicted a 20 to 25 percent loss in beach areas due to sea level rise by the year 2100 within the Archie Carr National Wildlife Refuge, Florida, as well as areas adjacent to the Refuge. With the threat of increasing sea level rise, protection of developed coastlines often involves shoreline armoring that reduces the amount of beach available, thus creating a smaller amount of space for turtles to nest (Hawkes *et al.* 2009). Along such developed coastlines, rising sea levels may cause

severe effects on eggs, because nesting females are forced to deposit eggs seaward of shoreline armoring, potentially subjecting them to repeated tidal inundation and/or egg exposure from exacerbated wave action near the base of these structures.

Sea level rise is expected to result in more nests being inundated, reducing hatching success. On Playona Beach, Colombia, Patiño-Martínez *et al.* (2014) found that nests in wet sand suffered higher mortality (emergence success of zero percent for wettest nests to 64 percent for the driest nests), suggesting that nesting success should be expected to decrease under future climate change sea level rise scenarios. Inundation is likely to reduce hatching success (Patiño-Martínez *et al.* 2008; Caut *et al.* 2010) and will continue to occur (or worsen) with sea level rise.

However, leatherback turtles may be less susceptible than other species of sea turtles to loss of nesting habitat, because they exhibit lower nest-site fidelity (Dutton *et al.* 1999). Nesting beaches in the Guianas are already highly dynamic and interseasonally variable, and leatherback nesting females have been successful in those areas despite the fact that some beaches disappear between nesting years (Plaziat and Augustinus 2004; Kelle *et al.* 2007; Caut *et al.* 2010). If global temperatures increase and there is a range shift northwards, beaches not currently used for nesting could in the future become used by leatherback turtles, potentially offsetting some loss of accessibility to beaches in southern portions of the range. Leatherbacks' behavioral flexibility may allow for opportunities to colonize new beaches, but whether turtles can colonize nesting areas that become available, either thermally or geographically, by climate change, and whether these colonized areas provide incubation regimes that will lead to successful nesting,

emergence success, and hatchling fitness cannot be known at this time (Hawkes *et al.* 2009).

Observed changes in marine systems are associated with other aspects of climate change, including rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean temperatures of the U.S. northeastern continental shelf and surrounding NW Atlantic waters have warmed faster than the global average over the last decade (Pershing *et al.* 2015). New projections for the U.S. northeastern shelf and NW Atlantic Ocean suggest that this region will warm two to three times faster than the global average and existing projections from the IPCC may be too conservative (Saba *et al.* 2015). This increase in northeastern shelf waters is relevant for NW Atlantic leatherback turtles, as they rely on U.S. and Canadian waters to forage during the warmer months (James 2005a, 2006b, 2007; Dodge 2014, 2015).

Global warming is expected to expand leatherback foraging habitats into, and increase residency time in, higher latitude waters (James *et al.* 2006a; McMahon and Hays 2006; Robinson *et al.* 2009). For example, leatherback turtles have extended their range in the Atlantic north by around 200 km per decade over the last two decades as warming has caused the northerly migration of the 15°C sea surface temperature (SST) isotherm, the lower limit of thermal tolerance for leatherback turtles (McMahon and Hays 2006). Documented weakening of the Meridional Overturning Circulation is related to above-average warming in the Gulf Stream region and an associated northward shift of the Gulf Stream (Caesar *et al.* 2018). This weakening of the deep, cold-water circulation in the North Atlantic is likely to continue to occur with global warming. Migratory routes may be altered by climate change as increasing ocean temperatures shift range-limiting

isotherms north (Robinson *et al.* 2009). Post-nesting females from French Guiana were found to migrate northward toward the Gulf Stream north wall, targeting similar habitats in terms of physical characteristics, *i.e.*, strong gradients of SST, sea surface height, and a deep mixed layer (Chambault *et al.* 2017). Hatchling dispersal may also be affected by changes in surface current and thermohaline circulation patterns (Hawkes *et al.* 2009; Pike 2013).

The effects of global warming are difficult to predict, but changes in reproductive behavior (*e.g.*, remigration intervals, timing and length of nesting season) could occur (Hawkes *et al.* 2009; Hamann *et al.* 2013). Robinson *et al.* (2014) found that the median nesting date at Sandy Point (St. Croix) occurred on average 0.17 days earlier per year, between 1982 and 2010. However, Neeman *et al.* (2015) found that increased temperatures at the foraging grounds tend to delay leatherback nesting. Temperatures at the nesting beaches (Playa Grande, Costa Rica; Tortuguero, Costa Rica; and St. Croix) did not affect the timing of leatherback nesting (Neeman *et al.* 2015). Because the relation between temperatures (local sea surface and the foraging grounds) and timing of nesting is complex, Neeman *et al.* (2015) indicated that further study is needed at the nesting beaches to determine how environmental conditions change within the season and how these changes affect nesting success. Robinson *et al.* (2014) suggests that shifts in the nesting phenology may make the Atlantic populations more resilient to climate change.

Extreme precipitation events over most of the mid-latitude and tropical regions will very likely become more intense and more frequent (IPCC 2014). Changes in the frequency and timing of storms or changes in prevailing currents could lead to increased

beach loss via erosion (Van Houtan and Bass 2007; Fuentes and Abbs 2010). More frequent and intense storm events will have the same effect on leatherback nesting success as previously described for natural disasters.

In summary, climate change is likely to affect multiple life stages of turtles in the NW Atlantic DPS. Likely impacts include altering sex ratios and reducing nest success, reducing nesting beach habitat and nests due to sea level rise and storms, and potentially changing distribution. Climate change therefore has the potential to alter productivity and diversity. These impacts could be more severe in certain areas with more dynamic beach environments, or could be widespread throughout the DPS. Impacts are likely to range from small, temporal changes in nesting season to large losses of productivity. That said, leatherback turtles are considered to be the best able to cope with climate change of all sea turtle species due to their wide geographic distribution and relatively weak nesting site fidelity. Overall, we conclude that climate change is a threat to the NW Atlantic DPS.

### *Conservation Efforts*

Next we consider “conservation efforts” under Section 4(b)(1)(A) (16 U.S.C. 1533(b)(1)(A)).<sup>1</sup> There are numerous efforts to conserve the leatherback turtle. The following conservation efforts apply to the NW Atlantic DPS (for a description of each effort, please see the section on conservation efforts for the taxonomic species): African Convention on the Conservation of Nature and Natural Resources (Algiers Convention); Central American Regional Network; Convention on the Conservation of Migratory Species of Wild Animals; Convention on Biological Diversity; Convention on

---

<sup>1</sup> For a related discussion of existing regulatory mechanisms to protect turtles, which are considered separately under Section 4(a)(1)(D), see the discussion above at “*Inadequacy of Existing Regulatory Mechanisms*.”

International Trade in Endangered Species of Wild Fauna and Flora; Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention); Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region, Specially Protected Areas and Wildlife (SPAW); Convention on the Conservation of European Wildlife and Natural Habitats; Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region (Abidjan Convention); Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum); Convention for the Protection and Development of the Marine Environment of the North East Atlantic; Convention on Nature Protection and Wildlife Preservation in the Western Hemisphere (Washington or Western Hemisphere Convention); Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region (Cartagena Convention); Cooperative Agreement for the Conservation of Sea Turtles of the Caribbean Coast of Costa Rica, Nicaragua, and Panama (Tri-Partite Agreement); Council Regulation (EC) No. 1239/98 of 8 June 1998 Amending Regulation (EC) No. 894/97 Laying Down Certain Technical Measures for the Conservation of Fishery Measures (Council of the European Union); Council Directive 92/43/EEC on the Conservation of Natural Habitats and of Wild Fauna and Flora (EC Habitats Directive); Food and Agricultural Organization (FAO) Technical Consultation on Sea Turtle-Fishery Interactions; Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC); MARPOL; Inter-American Tropical Tuna Convention (IATTC); IUCN; North American Agreement for Environmental Cooperation; Protocol Concerning Specially Protected

Areas and Biological Diversity in the Mediterranean; Ramsar Convention on Wetlands; Regional Fishery Management Organizations (RFMOs); UN Convention on the Law of the Sea (UNCLOS); and UN Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing. Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce its risk of extinction.

#### *Extinction Risk Analysis*

After reviewing the best available information, the Team concluded that the NW Atlantic DPS is at high risk of extinction. The total index of nesting female abundance is 20,659 females at consistently monitored beaches, and the most recent annual rate of decline is estimated to be approximately nine percent (NW Atlantic Leatherback Working Group 2018). The best available nest data reflect a steady decline for more than a decade, becoming more pronounced since 2008 (Eckert and Mitchell 2018; NW Atlantic Leatherback Working Group 2018). This decreasing trend is observed when all available nest data are combined and at most nesting beaches (NW Atlantic Leatherback Working Group 2018), including the largest nesting aggregation in Trinidad (*i.e.*, Grande Riviere, which is declining at 6.9 percent annually). In terms of productivity, the DPS exhibits low hatching success, while other key parameters such as clutch size, remigration interval, and clutch frequency are similar to species' averages. There are also indications of decreased productivity within the DPS at one of the most intensively monitored nesting beaches (*i.e.*, Sandy Point, St. Croix; Garner *et al.* 2017). The declining region-wide nest trend and potential changes in productivity make the DPS highly vulnerable to threats.

However, the DPS exhibits broad spatial distribution and some diversity. Based upon genetic data, as well as flipper tagging and satellite telemetry data, this DPS shows significant spatial structure with some connectivity among nesting and foraging areas. Further, nesting occurs in a variety of habitats, including islands and mainland, as well as muddy, sandy, and shelly beaches. The DPS uses multiple, distant, and diverse foraging areas, including oceanic and coastal waters throughout the North Atlantic Ocean, Mediterranean Sea, and GOM, providing some resilience against reduced prey availability. While the numerous and diverse nesting and foraging locations, along with moderate levels of genetic diversity, provide some level of buffer to the DPS, the highest concentrations of nesting occur in Trinidad, French Guiana, and Panama, where a catastrophic event could have a disproportionate impact on the DPS.

The primary threat to the NW Atlantic DPS is bycatch in commercial and artisanal, pelagic and coastal fisheries. Gillnet fisheries, in particular those off nesting beaches, are the greatest concern given the high mortality rate. In particular, the coastal surface drift gillnet fishery off Trinidad kills an estimated 1,000 adult leatherback turtles annually (Lee Lum 2006; Eckert *et al.* 2008; Eckert 2013). Bycatch, and subsequent mortality, in Trinidad bottom set gillnets and surface gillnets in Suriname and French Guiana are major threats to the NW Atlantic DPS. Trinidad and French Guiana host the highest number of nesting females in this DPS, so the continued mortality of adults in that area is of significant concern. Further, no adequate regulatory mechanism is currently in place (*e.g.*, no gear modifications or closures) to address this incidental bycatch. These fisheries and the related mortality rates have been occurring for years (Lee Lum 2006; Eckert 2013). Longline fisheries are the most widespread threat, occurring throughout the

Atlantic Ocean by fisheries from multiple nations, incidentally capturing thousands of leatherback turtles annually based on the best available data. Longline gear modifications (*e.g.*, circle hooks) are sometimes, but not consistently, used. Fishery bycatch in pot/trap gear, especially off the northeastern U.S. coast and in Canadian waters, and trawls are also significant threats. Fisheries bycatch reduces abundance by removing individuals from the population; when those individuals are nesting females, it reduces productivity as well. Given the lack of observer coverage and reporting, cumulative mortality due to fisheries bycatch is likely higher than available estimates.

Additional threats to the DPS include habitat loss, the legal and illegal harvest of turtles and eggs, predation, vessel strikes, pollution, climate change, oil and gas activities, and natural disasters. Coastal development and armoring, erosion (natural and anthropogenic), and artificial lighting are some of the most significant stressors on nesting beach habitat, reducing nesting and hatching success (*i.e.*, productivity). Habitat loss and modification is also anticipated to increase over time with additional development and climate change. Legal and illegal harvest of turtles and eggs reduces abundance and productivity. Illegal egg poaching occurs in several nations, particularly Costa Rica, Dominican Republic, and Colombia. While reduced in some nations, illegal poaching still occurs on unmonitored beaches throughout most of the Caribbean, including Suriname and Trinidad. While leatherback eggs and hatchlings are preyed upon by many species, the biggest threat is from feral dogs. Egg predation by dogs occurs in many nations, but it is a particular concern in Colombia, French Guiana, Guyana, Panama, Puerto Rico, and Trinidad and Tobago. Intervention (*e.g.*, screening) to reduce predation is not used in most places, partially due to the concern of attracting poachers as

well as the infeasibility of implementing effective measures at high-density or remote beaches. Egg predation reduces productivity.

Vessel strikes are also a threat, killing numerous leatherback turtles each year. While exposure to vessel strikes may be most severe in developed areas, the total impacts are high, affecting both abundance and productivity. Pollution, ingestion of plastics, and entanglement in marine debris are threats to all leatherback turtles, most likely resulting in injury and compromised health, and sometimes mortality. Exposure to pollution is widespread in the NW Atlantic Ocean, but effect data are limited. Oil and gas activities are threats with the potential to grow in some Caribbean areas. Natural disasters (hurricanes) and phenomenon (large *Sargassum* events) have an intermittent impact to the NW Atlantic DPS. Climate change is likely to result in reduced productivity due to greater rates of coastal erosion and sea level rise and subsequent nest inundation and habitat loss, reduced hatching success, changing sex ratios, and distributional changes. Although many international, national, and local regulatory mechanisms are in place, they do not adequately reduce the impact of these threats.

The cumulative impact of these multiple threats is potentially large (Andersen *et al.* 2017). Innis *et al.* (2010) reported that many individuals are simultaneously exposed to multiple threats, including: entanglement, injury, plastic ingestion, adrenal gland parasitism, diverticulitis, and burdens of environmental toxins (Innis *et al.* 2010). Such cumulative pressures affect individual survival and productivity. In some cases, it is possible to directly link individual threats to demographic reductions (*e.g.*, high mortality in gillnets off nesting beaches reduces nesting female abundance). More often, however, several threats contribute synergistically to demographic reductions. For example,

reductions in hatching success may be caused by one or more of the following threats alone or in combination: erosion, poaching, predation, climate change, and pollution.

We find that the NW Atlantic DPS is affected by numerous severe threats. These present, ongoing threats injure or kill turtles and contribute to the declining nest trend. The Team evaluated whether the DPS is at risk of extinction currently or would become so within the foreseeable future. To answer this question, they asked how long it would take for the total index of nesting female abundance to be reduced by 50 percent, a drastic decline that would reduce abundance to a level where demographic risks would present an independent threat to the DPS's continued existence, and whether this time period places the DPS at risk currently or within the foreseeable future. Using estimates of the mean time to maturation for the population (approximately 19 years; Avens *et al.* in review) and mean nesting longevity (approximately 11 years; Avens *et al.* in review) of the species, they estimated a generation time of approximately 30 years. They then considered three different scenarios. First, they calculated the time until 50 percent reduction in the total index of nesting female abundance using data on a significant and influential, well-documented, threat: gillnet bycatch mortality of 1,000 adult turtles annually off the largest nesting aggregation, *i.e.*, Trinidad. Assuming that half of the turtles at Trinidad killed are female, total index of nesting female abundance would decrease by 50 percent in 28 years, which is approximately one generation.

Second, the Team used regional nest trend data from the NW Atlantic Leatherback Working Group (2018). Using the most recent trends as is typical for population projections (*i.e.*, -9.32 percent per year from 2008 to 2017), they found that the total index of nesting female abundance would fall by 50 percent within 8 years (95

percent CI: 6 to 13 years). Using trends from the longer data set (−4.21 percent per year from 1990 to 2017), the total index of nesting female abundance would fall by 50 percent within 17 years (95 percent CI: 11 to 31 years). Finally, using their calculation of nest trend for the highest abundance nesting area in the DPS, Trinidad (−7.3 percent per year, 95 percent CI: -34 to 18 percent), the Team found that the total index of nesting female abundance would decrease by 50 percent within 10 years (95 percent CI: 3 years to “never;” however, “never” is highly unlikely, given that there is a 75 percent likelihood that the true value of the nest trend in Trinidad is negative ( $f = 0.754$ )). There are several caveats with using nest trend data: adult females typically account for, at most, a small percentage of the population; trends in nesting female abundance may not be an index of the remainder of population; stable age distribution is assumed; and time series of available data do not always span one generation (*let alone* multiple generations required to reach stable age distribution). Despite these caveats, all scenarios resulted in a 50 percent reduction in the total index of nesting female abundance in less than one generation. While the first scenario did not involve the use of nest trend data, it did result in a 50 percent reduction within one generation when considering only one threat (albeit the most severe), and we know that the DPS faces many large-impact threats, (suggesting that the first scenario understates the DPS’s degree of risk).

For the purpose of the extinction risk analysis, the Team discussed whether the resulting range of time periods (8 to 28 years) suggests a present risk of extinction or a risk of extinction within the foreseeable future. The Team did not have a unanimous view. All but one Team member were present to vote on the level of extinction risk. Eight Team members concluded with moderate confidence that the DPS is at high extinction

risk due to threats and the declining trend that has accelerated in recent years. Their confidence was moderate rather than high due to some resilience provided by the abundance, spatial distribution, and diversity for this DPS. Two Team members concluded with low confidence that the DPS is at moderate extinction risk. Their confidence in this conclusion was low due to the declining trend that has accelerated in recent years. The Terms of Reference called for a simple majority, and after voting, the Team concluded that the DPS met the definition for high risk of extinction. We agree with the Team's overall conclusion that a 50 percent decline in less than one generation equates to a current, high risk of extinction. We find support for this conclusion in well documented examples of other leatherback populations that have quickly declined despite larger abundances (*e.g.*, the Mexico nesting aggregation declined from 70,000 nesting females in 1982 to under 1,000 nesting females by 1994; Spotila *et al.* 2000).

We conclude that the NW Atlantic DPS is presently in danger of extinction due to the number and magnitude of threats, of which fisheries bycatch is the greatest concern. These present and ongoing threats have resulted in imminent and substantial demographic risks (*i.e.*, declining trends and reduced abundance). Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce the risk of extinction. We conclude that the NW Atlantic DPS is in danger of extinction throughout its range and therefore meets the definition of an endangered species. The threatened species definition does not apply because the DPS is currently at risk of extinction (*i.e.*, at present), rather than on a trajectory to become so within the foreseeable future.

### **SW Atlantic DPS**

The Team defined the SW Atlantic DPS as leatherback turtles originating from the SW Atlantic Ocean, north of 47° S, east of South America, and west of 20° W; the northern boundary is a diagonal line between 5.377° S, 35.321° W and 12.084620° N, 20° W. The southern boundary is based on the Antarctic circumpolar current which prevents sea turtles from nesting further south. The western end of the northern boundary is based at the “elbow” of the Brazilian coast, where the Brazilian Current begins and likely restricts the northern nesting range of this DPS. We placed the eastern boundary at the 20° W meridian as an approximate midpoint between SW Atlantic and SE Atlantic (*i.e.*, turtles that nest in western Africa) nesting beaches and to reflect both DPS’s wide foraging range throughout the South Atlantic Ocean. However, due to its low abundance, the SW Atlantic DPS is less likely to be encountered compared to the more abundant SE Atlantic DPS.

The SW Atlantic DPS only nests on the southeastern coast of Brazil, primarily in the state of Espírito Santo, on a continuous stretch of beach, less than 100 km in length, with concentrated nesting in Povoação and Comboios. While there is occasional, limited nesting south of these primary nesting beaches, the sand becomes coarser further south, and the excavation of nests becomes more difficult because the sand falls back into the holes (Thomé *et al.* 2007).

While nesting is limited geographically, the overall range of this DPS (*i.e.*, all areas of occurrence) is extensive, as demonstrated by individuals tracked to numerous foraging areas. Leatherback turtles of this DPS use coastal waters off South America from the “elbow” of Brazil southwards to Uruguay and Argentina, where quality foraging areas allow for coastal foraging in addition to open-ocean foraging (Almeida *et al.* 2011).

Individuals of this DPS are also known to migrate to the waters off western Africa and forage in the oceanic habitat in between South America and Africa (Almeida *et al.* 2011). Likewise, Prosdocimi *et al.* (2014) found 84 to 86 percent of leatherback turtles sampled from the foraging grounds off Argentina and Elevação do Rio Grande (an elevated offshore area across from Brazil) to originate from western African beaches.

### *Abundance*

The total index of nesting female abundance for the SW Atlantic DPS is 27 females. We based this index on nest monitoring data from Projeto TAMAR, the Brazilian Sea Turtle Conservation Program, which has established an index nesting survey area along 47 km of beach (10 km along Povoação and 37 km along Comboios; IAC Brazil Annual Report 2018), where complete daily surveys have been conducted during the primary nesting season from September through March, since the 1986/1987 nesting season. Some nesting occurs along the non-index stretches of Povoação and the beaches to the northern part of the area, but it is minor relative to nesting on the index survey area (Thomé *et al.* 2007). To calculate the index of nesting female abundance (*i.e.*, 27 nesting females) for the Espírito Santo index area, we divided the total number of nests between the 2014/2015 and 2016/2017 nesting seasons (*i.e.*, a 3-year remigration interval; Thomé *et al.* 2007) by the clutch frequency (5 clutches/season; Thomé *et al.* 2007; Tiwari *et al.* 2013).

Minimal, scattered nesting has been reported on beaches outside Espírito Santo (Barata and Fabiano 2002; Thomé *et al.* 2007; Bezerra *et al.* 2014), but these beaches exhibit suboptimal sand characteristics for nesting, limiting the possibility of substantial nesting expansion into those areas (Thomé *et al.* 2007). Therefore, while the nest counts

from the index beach surveys do not provide a full estimate of all nesting for the DPS, they provide a high-quality dataset, account for the majority of the nests (approximately 80 percent; Colman *et al.* 2019), and are used for determining our index of nesting female abundance and the nest trend in the next section.

Our total index of nesting female abundance is similar to the IUCN Red List assessment's estimate of 35 mature individuals (female and male, assuming a 3:1 sex ratio) based on nesting data through 2010 (Tiwari *et al.* 2013).

The total index of nesting female abundance (*i.e.*, 27 nesting females at the index beach) places the DPS at risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb losses in individuals. Due to its small size, the DPS has limited capacity to buffer such losses. Given the intrinsic problems of small population size, we conclude that the nesting female abundance is a major factor in the extinction risk of the SW Atlantic DPS.

### *Productivity*

The SW Atlantic DPS exhibits an increasing, although variable nest trend. Long-term monitoring data for this small DPS are limited to the index nesting survey area in southeastern Brazil, where data has been collected between the 1986/1987 and 2016/2017 nesting seasons. Over the 31-year data collection period, the mean annual number of nests for these beaches was 35. While this is below the criterion of 50 annual nests for conducting a trend analysis, we determined that this site should nevertheless be included due to the high quality and consistency of the data, and the fact that these data accurately

represent the low level of nesting of this DPS. The median increase in nest counts was 4.8 percent annually (sd = 5.8 percent; 95 percent CI = -8.4 to 15.5 percent;  $f = 0.832$ ; mean annual nests = 35). As the index area hosts the majority of known nesting activity, these data are representative of the entire DPS. We conclude that nesting has increased from 1986 to 2017. Our trend estimate is similar to that of the IUCN Red List assessment, which characterizes the population as increasing (Tiwari *et al.* 2013). It is also in agreement with the recent study by Colman *et al.* (2019), which describes the trend as increasing but variable, with the mean annual number of nests increasing from 25.6 nests in the first 5 years to 89.8 nests in the last 5 years of monitoring (between 1988 and 2017).

While the long term trend indicates an increase in nesting, the most recent 3 years of data (*i.e.*, 30, 64, and 38 nests from 2014 to 2016) show a marked reduction in nests compared to the previous 3 years (*i.e.*, 78, 124, and 102 nests from 2011 to 2013). The reason for this reduction is unknown. It could reflect declining nesting female abundance or changes in productivity metrics (*i.e.*, a longer remigration interval or reduced clutch frequency) related to environmental shifts or prey availability. Therefore, there is uncertainty regarding whether the increasing trend will continue.

The productivity parameters for this DPS are fairly typical for the species. In Brazil, the average clutch size appears to be on the lower end of the range for Atlantic populations; conversely, Brazilian nests tend to have a higher average number and percentage of eggs per clutch (Thomé *et al.* 2007). Therefore, the egg production of this DPS appears to be weighed more towards production of viable, hatchling-producing eggs compared to other Atlantic populations (Thomé *et al.* 2007). Nesting females produced

an average of 3,496 hatchlings annually over the past 10 years of nesting, which was calculated by multiplying 60.4 nests annually, 87.7 eggs per nest, and 66.0 percent hatching success (Colman *et al.* 2019). This estimate does not include the limited nesting outside the index area. The mean size of nesting females (CCL) has changed from 159.8 cm, with a range of 139 to 182 cm (Thomé *et al.* 2007) to 152.9 cm  $\pm$  10.0 SD, with a range of 124.7 to 182.0 cm; the decrease was statistically significant and may indicate recruitment (Colman *et al.* 2019). Hatching success has increased from a mean of 65.1 percent (with a range of 53.3 to 78 percent; Thomé *et al.* 2007) to a mean of 66 percent (with a range of 38.8 to 82.4 percent; Colman *et al.* 2019).

While the overall nest trend for this DPS is increasing, there is uncertainty regarding the continuation of this trend, given the data for the past 3 years. The population remains extremely small, and thus overall productivity is limited. Additionally, the potential for population growth is not clear, given the limited suitable nesting habitat available. We conclude that limited productivity places the DPS at risk of extinction.

### *Spatial Distribution*

The SW Atlantic DPS comprises a single, small nesting aggregation concentrated on the beaches of one state in Brazil (Espírito Santo). A tagging study has shown interesting movements along 300 km of the coast, including over 100 km on either side of known nesting beaches (Almeida *et al.* 2011), indicating connectivity throughout this area. The nesting spatial distribution is extremely restricted, with nesting constrained to a small area, with little suitable nesting habitat into which it can expand. Conversely, the DPS exhibits a broad foraging range, extending south to waters off Uruguay and

Argentina, throughout the pelagic waters of the South Atlantic, and across to western Africa (Almeida *et al.* 2011).

The wide distribution of foraging areas likely provides some level of buffer for the DPS against local catastrophes or environmental changes that could limit prey availability. However, the limited nesting range, and apparent lack of suitable nesting beaches into which to expand, renders the DPS highly susceptible to detrimental environmental impacts, both acute (*e.g.*, storms and singular events) and chronic (*e.g.*, sea level rise and temperature changes). Any such change would impact the entire extent of the DPS's nesting habitat. With no metapopulation structure, the DPS has reduced capacity to withstand other catastrophic events. Thus, despite widely distributed foraging areas, the extremely narrow nesting distribution and lack of population structure increases the extinction risk of the SW Atlantic DPS.

#### *Diversity*

Despite its extremely low nesting female abundance, the Brazilian nesting aggregation has the second-highest haplotype diversity among all Atlantic populations ( $h = 0.498 - 0.532$ ; Dutton *et al.* 2013; Vargas *et al.* 2017). According to Thomé *et al.* (2007), while most nesting occurs from September through March, sporadic nesting has been recorded throughout the year, which may provide temporal resilience if environmental conditions limit nesting during the primary nesting season. The use of estuarine waters (of the Rio de la Plata) as a year-round foraging ground is an unusual characteristic shared with the SE Atlantic DPS (Lopez-Mendilaharsu *et al.* 2009; Prosdocimi *et al.* 2014). Despite genetic and foraging diversity, the limited size and range of the nesting aggregation reduces the resilience of this DPS.

*Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

Within the limited nesting range of the SW Atlantic DPS, habitat modification is a threat. The 2015 collapse of a tailings dam at an ore mine upstream of the index nesting survey area had an undetermined, but potentially long-term, impact on the nesting beach of the DPS. Tens of millions of cubic meters of heavy metal-laden mining waste entered the Doce River and ultimately passed through the mouth of the river, in the middle of the index nesting area. Nests laid near the river mouth were relocated to prevent hatchlings from entering polluted waters. Hatching success was not significantly different between years in the period of 2012 to 2017, which include three seasons before (2012–2014) and three seasons after (2015–2017) the mining event (Colman *et al.* 2019). While no difference was noted in the distribution of nests following the dam breach, non-lethal impacts to individuals encountering the polluted waters, especially hatchlings, could not be measured. Such impacts may have occurred but may not be evident for decades following the spill. Projeto TAMAR is monitoring for heavy metals in eggs and nesting females and is closely watching for changes in fitness and reproductive parameters (Thomé *et al.* 2017). As a result of the dam's collapse, the Brazilian Federal government is implementing a marine protected area (APA-Area de Protecao Ambiental da Foz do Rio Doce), including about 100 kilometers of coastline, which should encompass the entire extension of the index nesting beaches, with both coastline and surrounding marine areas. Such a measure is an environmental compensation for the dam's collapse, and should be implemented with specific resources in the coming years (ICMBio, MMA, Brazil; J. Thomé, Projeto TAMAR, pers. comm., 2019).

Beach erosion and tidal flooding are also threats to this DPS. According to Thomé *et al.* (2007), occasional relocation of nests and nest protection occur when inundation or predation risk is considered high. The majority of nests are relocated when in danger of beach erosion or tidal flooding (J. Thomé, Projeto TAMAR, pers. comm., 2019).

Although coastal light pollution has been documented to be increasing in Brazil, nesting has not been notably impacted thus far (Colman *et al.* 2018). The lack of impact may be attributable to conservation strategies including the creation of protected areas and minimization of direct lighting on the nesting beaches. Nests are relocated from heavily lit areas. All light sources with a light intensity greater than 0 lux (lux = lumen per m<sup>2</sup>) on these beaches are prohibited by a Federal ordinance (Portaria IBAMA 11/1995). Construction, lighting, and poaching were not considered a significant problem at the leatherback nesting beaches by Thomé *et al.* (2007). However, such problems persist in several other turtle nesting beaches in Brazil (Mascarenhas *et al.* 2004; Lara *et al.* 2016). More recently, coastal development and artificial lighting have been identified as potential threats for leatherback turtles on the beaches of Espírito Santo (TAMAR/Unpublished data) and further research is needed to better understand these threats. Nests are relocated from heavily lit areas. Colman *et al.* (2018) found a negative relationship between nest density and light levels. Additionally, as oil industry and other economic developments are explored, the potential threat to the nesting habitat may increase (Thomé *et al.* 2007).

A significant portion of the nesting beach is protected as a Federal reserve under Brazilian Decree no. 90222 (September, 25 1984), which covers 15 km of Comboios Beach, south of the mouth of the Doce River. An additional 22 km, south of the reserve,

falls within indigenous land that has restricted access under Federal law. No Federally protected areas exist north of the Doce River mouth, where Povoação Beach occurs. However, local, state, and Federal regulations provide some coastal zone protections in that area.

#### *Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

Overutilization poses a threat to the SW Atlantic DPS. Though specific information on leatherback turtle harvests is not available, there was historically traditional harvest of sea turtles and eggs in Espírito Santo (Hartt 1941; Medeiros 1983). This harvest, however, has been largely curtailed through the work of Projeto TAMAR, which promoted other economic activities and hired ex-turtle hunters to protect nests (Marcovaldi *et al.* 2005; Almeida and Mendes 2007). The capture of leatherback turtles was banned in Brazil in 1968, and full protection for all sea turtles was enacted in 1986 (Marcovaldi and Marcovaldi 1999). At present, egg poaching has been reduced, and there is no known subsistence hunting for sea turtles of any species (Thomé *et al.* 2007). As previously noted, there is protection for or limited access to much of the nesting habitat south of the Doce River. However, this protection does not extend north of the river, where additional nesting occurs. Because of the very small size of the population, even very low levels of egg poaching have the potential to impact the population. Therefore, we conclude that overutilization poses a threat to the SW Atlantic DPS.

#### *Disease or Predation*

While we could not find any information on disease for this DPS, predation is a threat to the SW Atlantic DPS. Invertebrates, reptiles, and mammals prey on eggs, while hatchlings fall prey to land, air, and marine predators. According to Thomé *et al.* (2007),

relocation and protection of nests may be undertaken when inundation (primarily) or predation (secondarily) risk is considered high (J. Thomé, Projeto TAMAR, pers. comm., 2019). Predators include foxes (*Cerdocyon thous*), raccoons (*Procyon cancrivorus*), and domestic dogs, although there are no quantitative estimates of predation for this DPS (J. Thomé, Projeto TAMAR, pers. comm., 2019). Some predation of large juveniles and adults occurs in the marine environment, especially by sharks (Bornatowski *et al.* 2012), but the frequency and impact on those populations is not well understood. For this DPS, predation primarily impacts productivity (*i.e.*, reduced egg and hatching success). We conclude that predation is a threat to the SW Atlantic DPS, but that there is insufficient information to classify disease as a threat.

#### *Inadequacy of Existing Regulatory Mechanisms*

The SW Atlantic DPS is protected by several regulatory mechanisms. For each, the Team reviewed the objectives of the regulation and to what extent it adequately addresses the targeted threat.

Beach habitat is protected throughout much of the nesting range of this DPS. The vast majority of nesting occurs in Espírito Santo, where beaches have been protected since 1982. All light sources with a light intensity greater than 0 lux (lux = lumen per m<sup>2</sup>) on these beaches are prohibited by a Federal ordinance (Portaria IBAMA 11/1995).

The take of leatherback turtles is illegal throughout the SW Atlantic Ocean. Regional regulations include: Brazil Portaria, Manter proibida a captura de tartarugas marinhas das espécies *Caretta*, *Dermochelys coriacea*, *Eretmochelys imbricata* e

*Lepidochelys olivacea*<sup>2</sup> No.27/1982; Uruguay Presidential Decree 144 and additional legislation to reduce bycatch and prevent habitat alteration, and to prevent the removal of individuals from their natural environment; Argentina National Decree 666 from 1997; and various laws prohibiting hunting and selling sea turtles. Harvest and consumption of sea turtles are illegal under Brazilian law (Law on Environmental Crimes N° 9605/1998). While these protections are mostly effective, very low levels of egg poaching still exist (Thomé *et al.* 2007).

Fisheries bycatch is the primary threat to the SW Atlantic DPS. Although regulations address this issue to some extent, they do not do so adequately and it continues to be a threat. In 2001, Brazil established the National Plan for the Reduction of Incidental Capture of Sea Turtles in Fishing Activities (Marcovaldi *et al.* 2005). However, bycatch continues to be a major problem. In Brazil, the use of TEDs in trawl fisheries is mandatory (Instrução Normativa MMA No. 31; December 13, 2004), but most fishermen nevertheless do not use such gear, and there is little or no enforcement by authorities (IAC Brazil Annual Report 2018). The UN established a worldwide moratorium on drift gillnet fishing effective in 1992, the General Fisheries Commission for the Mediterranean prohibited driftnet fishing in 1997, and the International Commission for the Conservation of Atlantic Tunas (ICCAT) banned driftnets in 2003. Despite these and other numerous regulations and international instruments to protect sea turtles, significant bycatch still occurs in artisanal and commercial fisheries operating in the territorial waters of Argentina, Uruguay, and Brazil and on the high seas (González *et*

---

<sup>2</sup> Prohibition of the capture of sea turtles of the species *Caretta caretta*, *Dermochelys coriacea*, *Eretmochelys imbricata*, and *Lepidochelys olivacea*.

*al.* 2012).

In summary, while numerous regulatory mechanisms have been enacted to provide some protections to leatherback turtles, their eggs, and nesting habitat throughout the range of this DPS, they have been inadequate. Many do not effectively reduce the threat that they were designed to address, generally as a result of limited implementation or enforcement. Fisheries bycatch, in particular, remains a major threat to the DPS despite regulatory mechanisms. We conclude that the failure to implement and enforce effective regulations is a threat to the DPS.

#### *Fisheries Bycatch*

Fisheries bycatch is the primary threat to the SW Atlantic DPS. Leatherback turtles are captured as bycatch in commercial and artisanal fisheries, along coastal foraging and breeding areas, and on the high seas. The extensive foraging range of this DPS makes it vulnerable to interactions with fisheries off the coasts of Brazil, Uruguay, and Argentina, in the pelagic waters of the South Atlantic Ocean, and along the coastal waters off western Africa. Recoveries of females tagged in Espírito Santo are scarce, however. Three were found dead on the Brazilian coast (incidentally captured in fisheries around the Doce River mouth (TAMAR, unpublished data)), one in Argentina (Alvarez *et al.* 2009), and one in Namibia, West Africa (Almeida *et al.* 2014). Fisheries interaction information specific to this DPS is limited, because the data do not differentiate among individuals from this DPS and SE Atlantic individuals that forage within the same range. Because the SE Atlantic DPS is much more abundant than the SW Atlantic DPS, most fishery interactions likely involve SE Atlantic individuals. However, data about bycatch involving the SE Atlantic DPS is informative because the impact to the SW Atlantic DPS

individuals is likely to be proportional to their relative presence in the area. Bycatch in gillnets; surface, deep-water longline hooks; and trawls are the principal causes of sea turtle deaths, with not only higher interaction numbers, but higher mortality rates than other fishery interactions (Kotas *et al.* 2004; Pinedo and Polacheck 2004; Tudela *et al.* 2005; Giffoni *et al.* 2013).

Coastal gillnet fisheries interactions are one of the largest threats to the survival of the SW Atlantic DPS. In an analysis of Brazilian fishery data from 1990 to 2012, Giffoni *et al.* (2013) documented 237 leatherback turtle interactions, and 31 percent mortality, in coastal set, fixed, encircling, and pelagic drift gillnets. The actual number of interactions is likely substantially higher, as many interactions go unreported.

Smaller scale artisanal gillnet fisheries occur in coastal waters that are used by SW Atlantic individuals for mating, access to nesting beaches, and foraging. Thomé *et al.* (2007) described the occurrence of artisanal gillnet fisheries close to the nesting beach but indicated that Brazil was investing resources in developing lower-impact fishing techniques. However, such fisheries occur throughout other important coastal foraging areas off South America. Additionally, coastal artisanal gillnet fishery interactions with leatherback turtles are known to occur off the western coast of Africa, where some individuals from the SW Atlantic DPS forage (Riskas and Tiwari 2013). The Rio de la Plata estuary, an important foraging area off Uruguay, has numerous documented instances of leatherback turtle entanglements, including mortalities from coastal bottom-set gillnet fisheries (Fallabrino *et al.* 2006; Lopez-Mendilaharsu *et al.* 2009; Velez-Rubio *et al.* 2013).

Larger-scale commercial ocean gillnet fisheries are also a significant threat for the SW Atlantic DPS, with high bycatch rates reported off Brazil in drift and set gillnets (Fiedler *et al.* 2012; Ramos and Vasconcellos 2013). Drift gillnet fishing off Brazil started in 1986, targeting hammerhead sharks (Domingo *et al.* 2006). Marcovaldi *et al.* (2006) reported that leatherback turtles comprised about 70 percent of all sea turtles captured in Brazilian driftnet shark fisheries. From 2002 to 2008, 351 sea turtles were incidentally caught in 41 fishing trips and 371 sets. Leatherback turtles accounted for 77.3 percent of the take ( $n = 252$  turtles, capture rate = 0.1405 turtles/km of net) with 22.2 to 29.4 percent of turtles dead upon retrieval and no estimate of post-release mortality for those released alive. The annual catch by this fishery ranged from 1,212 to 6,160 leatherback turtles, as estimated based on bootstrap procedures under different fishing effort scenarios in the 1990s (Fiedler *et al.* 2012). In 1998, a Brazilian Federal ordinance limited the use and transport of bottom and drift gillnets over 2.5 km long. Such regulations were difficult to enforce, and vessels from the ports of Itajaí, Navegantes and Porto Belo, Santa Catarina, Brazil, deployed nets up to 7,846 m long between 2005 and 2006 (Kotas *et al.* 2008). In 2010 the ordinance was suspended, permitting unrestricted fishing with driftnets (Fiedler 2012). The shark drift gillnet fishery declined steeply in later years, with no vessels operating in 2009 (UNIVALI/CTTMar 2010) likely because of target species reduction, reduced profitability, and IBAMA Normative Instruction N166/2007 which temporarily stopped the issuance of new driftnet fishing licenses and established a 2-year deadline by which vessels were to replace driftnets with other gear. Various other gillnet fisheries, such as bottom gillnets for sharks and mollusks, have reported leatherback mortalities as well,

such as that occurring off Uruguay (Fallabrino *et al.* 2006; Laporta *et al.* 2006; Eckert *et al.* 2009) and the western coast of Africa (Riskas and Tiwari 2013).

Longline fisheries pose a significant threat to the SW Atlantic DPS, as the spatio-temporal distribution of leatherback turtles overlaps with longline fishing effort (Fossette *et al.* 2014). In a review of reported, observed takes in hook and line fishery (primarily longline) interactions with leatherback turtles in all of Brazil from 1990 to 2012, 1061 takes were documented, with 3 percent of the taken turtles found dead on the line and another 37.5 percent of unknown condition after release (Giffoni *et al.* 2013). High frequencies of leatherback deaths from bycatch have been documented on longline fishing vessels from southern Brazil and Uruguay (Kotas *et al.* 2004; Pinedo and Polacheck 2004; Domingo *et al.* 2006; Giffoni *et al.* 2008; Monteiro 2008). Between 2004 and 2005, in a study off southern Brazil, eight leatherback turtles were captured, with a mean capture rate of 0.03 turtles per 1,000 hooks (Monteiro 2008). In 1999, there were 70 longliners in the Brazilian fleet, with 33 vessels operating out of southern Brazil and fishing a total of 13,598,260 hooks (ICCAT 2001). However, the overall effort in the area was much higher, as longliners from Uruguay, Chile, Japan, Taiwan, and Spain fish in this area (Folsom 1997; Weidner and Arocha 1999; Weidner *et al.* 1999). Scientific observers documenting 10 trips by longline vessels from the Uruguayan fleet operating in the SW Atlantic Ocean between 26° and 37° S between April 1998 and November 2000 observed 27 incidentally caught leatherback turtles (Balestre *et al.* 2003). The prevalence of leatherback interactions in pelagic longline fisheries is likely a result of the longline fleet fishing the productive areas in the convergence zone of the Brazilian Current and the cold waters from the Falklands Current (Kotas *et al.* 2004), which coincides with

important sea turtle foraging and developmental habitat. As with gillnets, the scope of the longline threat to the SW Atlantic DPS spans across the South Atlantic Ocean in both coastal and oceanic waters. In addition to exposure to longline fisheries off South America, coastal longline fisheries off Cameroon, Angola, and Namibia, and pelagic longlines in the Gulf of Guinea and the eastern portion of the South Atlantic Ocean have also been documented to take leatherback turtles (Honig *et al.* 2007; Riskas and Tiwari 2013; Angel *et al.* 2014; Huang 2015; Gray and Diaz 2017). Additional evidence of longline interactions comes from the stranding data, where flipper injuries on some of the stranded leatherback turtles could have been caused by interactions with pelagic longlines. Onboard observers in longline fisheries off Brazil have reported that leatherback turtles tend to be foul-hooked in the flipper rather than the mouth (Kotas *et al.* 2004; Pinedo and Polacheck 2004; Lima 2007). In 2017, Brazil enacted a law (PORTARIA INTERMINISTERIAL No 74, DE 10- DE NOVEMBRO DE 2017) requiring the use of circle hooks in the pelagic longline fisheries as well as keeping specified dehooking and gear removal equipment on board any Brazilian longline vessel. Specifically, the Brazilian government required the use of 14/0 or larger circle hooks for all longline vessels targeting swordfish or tuna

(<https://www.jusbrasil.com.br/diarios/166677996/dou-secao-1-06-11-2017-pg-81>).

Trawl fisheries also impact the SW Atlantic DPS, mainly along coastal waters off southern Brazil, Argentina, and Uruguay (Gonzalez Carman *et al.* 2011; Velez Rubio *et al.* 2013; Monteiro *et al.* 2016). Although there are fewer interactions with trawl fisheries relative to other fisheries (*i.e.*, gillnet and longline fisheries), mortality rates in trawl fisheries are far higher (Miller *et al.* 2006; Laporta *et al.* 2013). Observation of the

Uruguayan bottom trawl fishery, during a tagging and data collection program designed to increase the understanding of the fishery impacts on sea turtles, documented 17 leatherback interactions from April 2002 to June 2005 (Laporta *et al.* 2013). Coastal bottom trawl and artisanal gillnet fisheries were the main causes of death of leatherbacks found stranded in Uruguay (Velez Rubio *et al.* 2013). Recorded interactions in coastal trawl fisheries are also known from Gabon, Congo, and Namibia (Riskas and Tiwari 2013).

Other fisheries such as corrals, pound nets, and pots appear to present a much lower risk for leatherback turtles than to other sea turtle species. From 1990 to 2012, Giffoni *et al.* (2013) documented only two leatherback turtles (both alive) of the 8,367 total sea turtles taken in those fisheries.

While specific information is not available to permit calculating an estimate of overall bycatch and mortality rates of SW Atlantic leatherback turtles, it is clear that fisheries bycatch, especially in gillnets and longlines, is a major threat to the DPS. Immature and adult individuals are exposed to high fishing effort throughout their foraging range and in coastal waters near nesting beaches. Bycatch mortality is also high, with reported rates of up to 31 percent (Giffoni *et al.* 2013). Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when nesting females are incidentally captured and killed. Given the small size of the DPS, the loss of even a small number of individuals from fishery interactions has the potential to reduce abundance and productivity. Therefore, we conclude that fisheries bycatch is the primary threat to the SW Atlantic DPS.

#### *Vessel Strikes*

There is little information regarding vessel strikes for the SW Atlantic DPS. Many of the primary foraging areas for this DPS off the coasts of Argentina, Uruguay, and Brazil are experiencing increased vessel traffic from fishing vessels, cargo transport, and tourism (López-Mendilaharsu *et al.* 2009; Fossette *et al.* 2014), so leatherback turtle interactions with vessels may occur. Affected individuals likely include immature and mature turtles. Impacts range from injury to mortality. We conclude from the best available information that vessel strikes are likely a threat to the DPS.

### *Pollution*

As with all leatherback turtles, entanglement in and ingestion of marine debris and plastics is a threat that likely kills several individuals a year. Multiple studies have implicated the ingestion of marine debris and/or entanglement in cases of injury or death of turtles found in waters occupied by the SW Atlantic DPS (Bugoni *et al.* 2001; Eckert *et al.* 2009; Schulyer *et al.* 2013; Scherer *et al.* 2014). However, no individuals were assigned to a particular population and could have been members of the more abundant SE Atlantic DPS, which is known to occupy the same waters.

While there is no specific information on effects to leatherback turtles of this DPS, pollution from various economic activities including maritime transport, tourism, and domestic and industrial waste discharges that are known to occur within their range, may also have an impact (López-Mendilaharsu *et al.* 2009; Fossette *et al.* 2014). Events such as the failure of a mining tailings dam in 2015 that resulted in the discharge of tons of mining mud contaminated with heavy metals into the Doce River, and subsequently into the waters off Espírito Santo nesting beaches, are also a concern, though no specific impacts to leatherback turtles have so far been noted from that event (Garcia *et al.* 2017).

There is also concern about the potential for increased oil and gas exploration activities (Thomé *et al.* 2007). The petroleum industry in Brazil has implemented a beach monitoring program, along large stretches of the Brazilian coast, including Espírito Santo, to monitor for potential impacts to sea turtles and their nesting beaches from industry activities (Werneck *et al.* 2018)

Assigning impacts of pollution specifically to individuals within the SW Atlantic DPS is difficult, and the best available information does not quantify such impacts. However, given its prevalence, we conclude that pollution poses a threat to the DPS.

### *Climate Change*

Climate change poses a threat to the SW Atlantic DPS. The impacts of climate change include: increases in temperatures (air, sand, and sea surface); sea level rise; increased coastal erosion; more frequent and intense storm events; and changes in ocean currents.

Because leatherback turtles nest lower on the beach than other sea turtles, their eggs are more at risk of being exposed and destroyed by increases in sea level and coastal erosion (Boyes *et al.* 2010). Additionally, given the limited availability of suitable nesting habitat, the loss of the current nesting habitat with no buffer area to move into would pose a major problem for the DPS. Thus, rising sea level and beach erosion are potential threats to the DPS.

While we do not have specific information on pivotal temperatures and temperature thresholds for egg mortality for this DPS, sand temperatures influence egg viability and sex determination. Given the potential lack of suitable nesting habitat outside the area currently being utilized, there is little opportunity for a spatial shift in

nesting in response to changing temperatures. This DPS exhibits some year-round nesting, which provides a small measure of resilience to counteract increasing temperatures. However, it is not likely to be sufficient to make up for the loss of nesting habitat and opportunity resulting from sea level rise and temperature increases. The impacts on productivity and survivorship for such shifts in nesting are unknown.

The threat of climate change is likely to modify the nesting conditions for the DPS. Adverse impacts on turtles of the SW Atlantic DPS would be inescapable because the entire DPS is confined to a limited nesting area. Impacts are likely to range from small, temporal changes in nesting season to large losses of productivity. Therefore, we conclude that climate change is a threat to the DPS.

#### *Channel Dredging*

There is evidence of interactions with hopper dredges associated with channel dredging and maintenance. Between 2008 and 2014, four leatherback turtles were killed by hopper dredges in Rio de Janeiro (Goldberg *et al.* 2015).

#### *Conservation Efforts*

There are numerous efforts to conserve the leatherback turtle. The following conservation efforts apply turtles of the SW Atlantic DPS (for a description of each effort, please see the section on conservation efforts for the overall species): Southwest Atlantic Sea Turtle Network, Convention on the Conservation of Migratory Species of Wild Animals, Convention on Biological Diversity, Convention on International Trade in Endangered Species of Wild Fauna and Flora, Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention), FAO Technical Consultation on Sea Turtle-Fishery Interactions, IAC, MARPOL, IUCN, Ramsar

Convention on Wetlands, RFMOs, South Atlantic Association, UNCLOS, and UN Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing. Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce its risk of extinction.

### *Extinction Risk Analysis*

After reviewing the best available information, the Team concluded that the SW Atlantic DPS is at “high” risk of extinction. The DPS exhibits a total index of nesting female abundance of 27 females at the index beach. Such a nesting population size places this DPS at risk of stochastic or catastrophic events that increase its extinction risk. Although there has been substantial variability in nesting at the index nesting beach since 1986, the nest trend shows a strong, nearly five percent annual increase through 2017, with the largest increase occurring in the past decade. However, nesting has declined in the past 3 years. There is only one nesting aggregation, limited to a relatively small stretch (47 km) of beach along a single coast. Some nesting also occurs outside that area, but is mostly sporadic and limited by sand and temperatures unsuited for nesting. Thus, stochastic events have the potential to have catastrophic effects on the entire DPS, with no distant subpopulations serving as a buffer or source of additional individuals or diversity. Based on these factors, we find the DPS to be at risk of extinction as a result of its limited abundance, spatial structure, and resilience.

Current threats place this DPS at further risk of extinction. The primary threat to this DPS is bycatch in commercial and artisanal, pelagic and coastal fisheries, especially gillnet and longline fisheries. Fisheries bycatch reduces abundance by removing individuals from the population. Because several fisheries operate near nesting beaches,

productivity is also reduced when nesting females are prevented from returning to nesting beaches. Exposure to and impact of this threat are high. Additional threats include: habitat modification, overutilization, predation, pollution, vessel strikes, and climate change. Habitat modification includes incidents such as the mining dam breach upstream of the Doce River, which flows into the ocean through the middle of the primary nesting beach. Overutilization and predation are threats for this DPS as well, though some protective measures exist. While many laws are in place to protect sea turtles from fishery impacts, the continued impact of bycatch indicates that regulatory mechanisms are inadequate to sufficiently address the threat. Pollution and vessel strikes are potentially increasing threats to the DPS. Climate change is another threat that is likely to increase, resulting in reduced productivity due to greater rates of coastal erosion and nest inundation, and in some areas, nest failure or skewed sex ratios due to increased sand temperatures.

We conclude, consistent with the Team's findings, that the SW Atlantic DPS is currently in danger of extinction. The total index of nesting female abundance make the DPS highly vulnerable to threats despite the apparent increasing nesting trend. In addition, this DPS consists of only one small nesting aggregation with limited potential nesting beaches to the north and south for expansion. The limited nesting range and small size makes the DPS highly vulnerable to stochastic impacts in the natural environment as well as singular, large-scale, anthropogenic events such as oil spills. Some degree of resilience is provided by the use of multiple foraging areas across a vast geographic area. However, that expansive foraging range also exposes the DPS to numerous fisheries (which are coastal and on the high seas, artisanal and commercial, off both South

America and western Africa), making fisheries bycatch by far the biggest threat to the DPS. Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce the risk of extinction. We conclude that the SW Atlantic DPS is currently in danger of extinction throughout its range and thus meets the definition of an endangered species. The threatened species definition does not apply because the DPS is at risk of extinction now (*i.e.*, at present), rather than on a trajectory to become so within the foreseeable future.

### **SE Atlantic DPS**

The Team defined the SE Atlantic DPS as leatherback turtles originating from the SE Atlantic Ocean, north of 47° S, east of 20° W, and west of 20° E; the NW boundary is a diagonal line between 12.084620° N, 20° W and 16.063° N, 16.51° W. The eastern boundary occurs at the southern tip of Africa, where the Agulhas and Benguela Currents meet. Along with the cold waters of the Antarctic Circumpolar Current, these currents likely restrict the nesting range of this DPS. We placed the western boundary at the 20° W meridian as an approximate midpoint between SE Atlantic and SW Atlantic (*i.e.*, turtles that nest in Brazil) nesting beaches and to reflect the DPS's wide foraging range throughout the South Atlantic Ocean; this DPS is more likely to be encountered in these waters compared to individuals from the less abundant SW Atlantic DPS. The northern boundary is a diagonal line between the elbow of Brazil and the northern boundary of Senegal because the SE Atlantic DPS does not appear to nest above this boundary (Fretey *et al.* 2007).

The range of the SE Atlantic DPS is extensive, mirroring that of the SW Atlantic DPS. While nesting occurs along the western coast of Africa, data indicate that foraging

areas and migratory paths stretch along the Atlantic coast of Africa from Senegal to South Africa, across the South Atlantic Ocean, and into the coastal waters of Brazil, Uruguay, and Argentina. As with the SW Atlantic DPS, this DPS does not appear to forage in northern latitudes.

All nesting for the SE Atlantic DPS occurs along the Atlantic coast of western Africa, from Senegal to Angola, a nesting range of over 7,500 km. However, the vast majority of nesting occurs in Gabon, Equatorial Guinea (including Bioko Island), and the Republic of Congo (TEWG 2007; Fretey *et al.* 2007, Witt *et al.* 2009; Tiwari *et al.* 2013). Gabon may have once hosted the largest nesting aggregation in the world when it was discovered in the early 2000s (Witt *et al.* 2009), but current data indicate much lower levels of nesting (Formia *et al.* in prep) compared to those described in Witt *et al.* (2009).

While nesting occurs along the western coast of Africa, foraging grounds and migratory paths stretch across the South Atlantic Ocean to the coastal waters of Brazil, Uruguay, and Argentina. Because of the greater abundance of this DPS, most individuals found in the western South Atlantic along the coast of South America, and on the high seas, belong to the SE Atlantic DPS. Prosdocimi *et al.* (2014) found 84 to 86 percent of leatherback turtles sampled from the foraging grounds off Argentina and Elevação do Rio Grande (an elevated offshore area across from Brazil) to originate from western African beaches.

### *Abundance*

The total index of nesting female abundance for the SE Atlantic DPS is 9,198 females. We based this total index on nine nesting aggregations in Gabon (n = 8,495 nesting females), Equatorial Guinea (n = 457), Republic of Congo (n = 69), Sierra Leone

(n = 39), Liberia (n = 45), Ivory Coast (n = 40), Ghana (n = 4), Cameroon (n = 3), and Sao Tome and Principe (n = 46). Our total index does not include 10 unquantified nesting aggregations in Guinea-Bissau, Angola, and other nations. For more information on data sources and calculations, please see the Status Review Report.

Our total index of nesting female abundance is an index because we do not have consistent data from much of the nesting range of the DPS, which extends from Senegal to Angola. However, the largest nesting aggregations occur in Gabon, Equatorial Guinea (including Bioko Island), and the Republic of Congo (TEWG 2007; Fretey *et al.* 2007; Witt *et al.* 2009; Tiwari *et al.* 2013), which are represented in our total index. The IUCN Red List assessment did not provide an estimate of population size but instead concluded that the subpopulation was “data deficient” (Tiwari *et al.* 2013).

To calculate the index of nesting female abundance in Gabon, where annual aerial surveys of 600 km of nesting beaches gather emergence data, we used a remigration interval of 3 years, a clutch frequency of 7.8 clutches per season per female, and estimated that 95 percent of emergences resulted in nesting (Formia *et al.* in prep). Our index of nesting female abundance for Gabon (*i.e.*, 8,495 nesting females) is lower than previous estimates. According to Witt *et al.* (2009), Gabon once hosted the largest leatherback nesting aggregation in the world, with an estimated 36,185 to 126,480 clutches per year (approximately 15,730 to 41,373 nesting females). These estimates were based on a combination of aerial surveys and ground-truthing surveys, conducted during the 2002/2003, 2005/2006, and 2006/2007 nesting seasons. More recent aerial surveys indicate a steep decline in nesting since the early 2000s, with a high of 108,588

estimated nests in 2002/03, a low of 4,275 estimated nests in 2009/10, and fewer than 25,000 nests in the final year of available data (2015/16; Formia *et al.* in prep).

Nesting is scattered on continental Equatorial Guinea (Fretey 2001), but it occurs on several beaches of Bioko Island and is monitored at the Gran Caldera Scientific Reserve (n = 457 nesting females, based on body pit data from the 2000/2001 through 2017/2018 nesting seasons; D. Venditti *et al.*, Drexel University, pers. comm., 2018). Rader *et al.* (2006) documented an average of 3,896 nests annually between the 2000/2001 to 2004/2005 nesting seasons, which equates to approximately 2,338 nesting females (*i.e.*, using a 3-year remigration interval and a clutch frequency of 5 nests annually). Based on the data available on nesting in the Republic of Congo from the 2003/2004 to 2016/2017 nesting seasons (N. Breheret, SWOT, pers. comm., 2018), we estimated 69 nesting females. In an analysis of older data (1999 to 2008), Girard *et al.* (2016) estimated 933 nests per year on the monitored beaches, which equates to approximately 560 nesting females.

In Guinea-Bissau, only one beach is monitored regularly, in Orango National Park, Bijagos Archipelago, where occasional leatherback nesting tracks are recorded. Each season, a few nests are reported elsewhere throughout the nation (Barbosa *et al.* 1998; Fretey *et al.* 2007).

In the Ivory Coast (n = 40 nesting females), Gomez (2005) counted 218 nests over 41 km of beach in February 2001. Peñate *et al.* (2007) reported 189 nests reported from non-exhaustive surveys of 27 km of coastline during the 2001/2002 nesting season.

In Ghana, nest monitoring occurs on three beaches: Mankoadze (n = 4 nesting females), Ada, and Keta. We were unable to calculate the index for Ada and Keta

beaches because we only received information on nest averages. From 2000 to 2017, an annual average of 34 nests were observed on Ada Beach (D. Agyeman, pers. comm., 2018). During the 2006/2007 nesting season, 481 leatherback nests were counted on Ada Beach (Allman and Armah 2010). Over an unspecified time frame, an annual average of 80 nests were observed on Keta Beach (A. Fuseini, pers. comm., 2018).

In Cameroon ( $n = 3$  nesting females; Fretey and Nibam unpublished data 2018), Girard *et al.* (2016) estimated an average of 43 leatherback nests annually, which would equate to 26 nesting females, from 1999 to 2008. In São Tomé and Príncipe ( $n = 46$  nesting females), Girard *et al.* (2016) estimated an average of 78 nests annually from 1999 to 2008, which is similar to our estimate.

Nesting occurs on other beaches throughout western Africa. However, recent consistent and standardized monitoring data are not available. Sporadic nesting occurs in Senegal (Maigret 1978; Dupuy 1986), Republic of The Gambia (Barnett *et al.* 2004, Hawkes *et al.* 2006), Togo (Segniagbeto 2004), Nigeria (Fretey 2001; Mojisola *et al.* 2015), Democratic Republic of Congo, (OCPE-ONG 2006), and Angola (Carr and Carr 1991; Weir *et al.* 2007).

The total index of nesting female abundance of the SE Atlantic DPS (9,198 females) does not reduce the risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017). Such abundance provides little resilience to buffer losses of individuals. We conclude that the nesting female abundance, as estimated, does not reduce the extinction risk of this DPS.

*Productivity*

Based on data collected from the largest nesting aggregation (*i.e.*, Gabon), the SE Atlantic DPS exhibits a declining nesting trend. Data collected between the 2002/2003 and 2015/2016 nesting seasons (with two years of missing data) indicated a median trend in nesting activity of -8.6 percent annually (sd = 21.9 percent; 95 percent CI = -52.6 to 36.9 percent;  $f = 0.676$ ; mean annual nesting activities = 35,204). The trend in Gabon is likely representative of the entire DPS, because the majority of nesting occurs there. Additional nest trend data are available from the Gran Caldera Scientific Reserve of Bioko Island, where the number of body pits increased 2.8 percent annually (sd = 15.6 percent; 95 percent CI = -27.2 to 36.0 percent) from 1996/1997 to 2017/2018.

Regarding productivity parameters, available information is often from a limited area and may not be representative of the entire DPS. However, based on available data, the size of nesting females, clutch size, hatching success, and incubation period appear to be similar to the species' averages. We conclude that the declining nesting trend contributes to the extinction risk of this DPS.

### *Spatial Distribution*

The SE Atlantic DPS has a broad spatial distribution. The nesting range is centered on Gabon, with nesting occurring from Senegal to Angola. Genetic data available for Gabon and Ghana indicate significant genetic differentiation based on mtDNA data, but weak differentiation based on analysis of nuclear DNA, likely indicating demographically independent subpopulations connected by limited gene flow (Dutton *et al.* 2013).

In addition to the extensive nesting range, this DPS also has an expansive foraging and migratory range, from the coastal waters of Atlantic Africa, across the

pelagic waters of the South Atlantic, and along the South American coast from Brazil to Argentina. While nesting along the coast of Africa extends only to Angola, recent tag returns and satellite telemetry indicate that turtles utilize the waters in Namibia as well (Almeida *et al.* 2014). Transatlantic movements were first recorded from tag returns of four leatherback turtles tagged on the nesting beaches of Gabon and recaptured in the waters of Argentina and Brazil (Billes *et al.* 2006). Satellite telemetry confirmed that nesting females from Gabon follow three different post-nesting movement trajectories towards the equatorial Atlantic Ocean, South America, or southern Africa (Witt *et al.* 2011). For combined foraging areas off Argentina and Elevação do Rio Grande (an elevated offshore area across from Brazil), the mean estimate from western Africa was 84 to 86 percent (45 percent Gabon, 41 percent Ghana; Prosdocimi *et al.* 2014).

The wide distribution of foraging areas likely buffers the DPS against local catastrophes or environmental changes that could limit prey availability. The expansive nesting range may buffer the DPS from acute environmental impacts (*e.g.*, storms and singular events) and to some degree, chronic impacts (*e.g.*, sea level rise and temperature changes). Thus, the combination of extensive nesting range, widely distributed foraging areas, and population structure reduces the extinction risk of the SE Atlantic DPS.

### *Diversity*

Genetic analyses for the SE Atlantic DPS are limited, but Dutton *et al.* (2013) found moderate genetic diversity in samples from Gabon and Ghana, including four new haplotypes unique to western African nesting females. Nesting occurs on continental and insular beaches. There are multiple foraging strategies, including pelagic and coastal, along either side of the Atlantic Ocean. The genetic diversity, along with multiple and

diverse foraging sites (*i.e.*, coastal and pelagic), and combination of insular and mainland nesting provide diversity and resilience that may reduce the extinction risk of this DPS.

*Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

Modification and loss of habitat is a threat to the SE Atlantic DPS. Present threats include obstructions, erosion, and light pollution at nesting beaches. Future threats include coastal construction and development in the region.

Nesting beach obstruction due to logs is a problem in Gabon, Equatorial Guinea, and Cameroon (Formia *et al.* 2003). Logs that have broken loose from timber rafts of industrial logging operations wash up on the beaches of Gabon at densities of up to 247 logs/km; logs blocked 30.5 percent of the beach in Pongara, Gabon, resulting in an estimated 2,111 disrupted or aborted nesting attempts (Laurance *et al.* 2008). In addition, several leatherback turtles have died as result of being trapped by logs (Laurance *et al.* 2008). Pikesley *et al.* (2013) determined that between 1.6 percent and 4.4 percent of nesting females could be trapped at beaches with high log- and turtle-densities. However, Gabon has since banned the export of whole logs. The Gabon Sea Turtle Partnership has carried out log removal efforts for at least one high-density nesting beach in Pongara National Park (Kingere Beach), and a 3 km stretch of nesting beach is now virtually free of logs; at the other main monitored beaches in Gabon, such as Mayumba and Gamba, logs are not a major threat (A. Formia, WCS, pers. comm. 2019).

Habitat loss from coastal erosion due to sand mining, harbor building, and irregular current flows has compromised the suitability of long stretches of coastal areas as nesting sites. This issue is especially prevalent between Ghana and Nigeria (Formia *et al.* 2003). Ikaran (2010) found low hatching/emergence success rates at three nesting sites

in Gabon: Pointe Denis (17/16 percent), Mayumba (43/40 percent), and Kingere (16/16 percent). In addition to predation, the main identified sources of egg mortality were beach erosion and inundation (Ikaran 2010).

Light pollution modifies nesting beach habitat, deterring nesting females and disorienting both hatchlings and nesting females. Bourgeois (2009) found that artificial lighting disoriented leatherback hatchlings in Pongara National Park, Gabon: hatchlings in 27 of the 41 nests (66 percent) studied crawled towards artificial lights. Deem *et al.* (2007) documented 71 disoriented females that crawled directly into the savannah behind the beach and towards the artificial lights. Bourgeois *et al.* (2009) concluded that light pollution from Libreville and Pointe Denis, Gabon is a major threat to nesting females and hatchlings, which become disoriented and die in the surrounding savannah.

Urbanization and coastal development are rapidly growing threats at some nesting beaches (Girard and Honarvar 2017). There is a high potential for coastal development in Gabon, including the beaches near Pointe Denis, an important and growing tourist area (Ikaran 2010). Along with direct habitat loss from coastal development and urbanization, impacts from pollution and litter are expected to increase.

In Gabon, a network of marine protected areas was created by decree 00161/PR in 2017, covering 26 percent of Gabon's territorial seas, including a vast area in front of the most important nesting beach in Gabon (Mayumba National Park) that stretches to the outer limits of the EEZ.

We conclude that a large portion of nesting females, hatchlings, and eggs are exposed to the reduction and modification of nesting habitat, as a result of logging,

erosion, coastal development, and artificial lighting. These threats impact the DPS by reducing nesting and hatching success, thus lowering the productivity of the DPS.

Logging also results in the death of nesting females, reducing the abundance of the population by removing its most reproductively important individuals. Based on the information presented above, we conclude that habitat loss and modification are major and increasing threats to the DPS.

#### *Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

Overutilization is a threat to the SE Atlantic DPS. Although receiving some legal protections, eggs and turtles nevertheless are poached for consumption, traditional medicine, and religious practices.

In Gabon, poaching is limited because 78 percent of nesting occurs within national parks and human population density along the coast is low (A. Formia, Gabon Sea Turtle Partnership, pers. comm., 2018). However, elsewhere in the region, poaching occurs at a high rate, or would be reasonably expected to return to high levels, if not limited by activities funded through the USFWS' Marine Turtle Conservation Fund enacted under the MTCA. These activities reduce poaching through increased presence on nesting beaches, beach monitoring, hiring of local citizens for participation in the projects, and raising awareness and providing education to local communities (M. Tiwari, NMFS, pers. comm. 2018).

Conflicting beliefs about sea turtles exist throughout the region. In some communities sea turtles are considered divinely provided food, while in others they have been historically protected by indigenous custom, often based on stories passed down by ancestors (Barbosa and Regalla 2016; Alexander *et al.* 2017). In general, however,

poaching is a significant problem throughout the region. Catry *et al.* (2009) concluded that, in addition to fisheries bycatch, poaching of eggs and nesting females is the main threat to sea turtles, including leatherback turtles, in Guinea-Bissau. In many cases “few if any turtles or nests are left alone when found by locals” (Catry *et al.* 2009). The fat of leatherback turtles is often used for various purported medicinal applications, including: treatment of convulsions and malaria (Togo), fever, fainting spells, liver problems, tetanus (Benin), and to induce vomiting (Togo, Benin). In one community in the Ivory Coast and parts of Cameroon, leatherback turtle fat is applied to wounds in the mouth and is used to massage into painful joints. In northwestern and southern Cameroon, it is applied to bruises (Fretey *et al.* 1999). In Togo, some mothers add turtle bones daily to the baby’s bath water; some believe that the power of the turtle (especially the leatherback) will be transmitted to the child through this practice (Segniagbeto 2004).

Turtles and eggs are poached throughout the nesting range of the DPS. Though most nesting females and eggs are protected in Gabon, poaching is widespread in other areas. Poaching of nesting females reduces both abundance (through loss of nesting females) and productivity (through loss of reproductive potential). Such impacts are high because they directly remove the most productive individuals from DPS, reducing current and/or future reproductive potential. Egg poaching reduces productivity. Given the moderate exposure and high impact, we conclude that the poaching of turtles and eggs poses a threat to the DPS.

#### *Disease or Predation*

Information on diseases among leatherback turtles originating in the SE Atlantic is minimal, but an analysis of samples from nesting females in Gabon indicated normal

blood chemistry parameters (Deem *et al.* 2006). Predation may occur at high rates in some areas, but information is limited.

Predation of leatherback eggs and/or hatchlings has been documented for a variety of predators, including: various ants, ghost crabs, monitor lizards (*Varanus niloticus*), crows (*Corvus albus*), mongoose, porcupine (*Atherurus africanus*), domestic dogs, African civet cat (*Civettictis civetta and Viverra civetta*), and drills (*Mandrillus leucophaeus*) (summarized from Eckert *et al.* 2012). In Kingere, Gabon, Ikaran (2010) noted high predation rates of eggs by crabs, lizards, mongooses, small cat species, and ants. Predation was the main source of egg mortality at three nesting sites in Gabon: Pointe Denis (43 percent), Mayumba (44 percent), and Kingere (51 to 56 percent; Ikaran 2010).

As is common for all sea turtle species, leatherback hatchlings likely experience predation from various fish species as they enter the water and swim towards the open ocean. In-water predation of juveniles and adults is not well-documented, but there is evidence of shark and killer whale predation. Shark predation was determined to be the cause of one leatherback stranding reported from Central Africa (Parnell *et al.* 2007), while interactions between killer whales and leatherback turtles resulting in possible predation has been observed in Namibian waters (Elwen and Leeney 2011).

While all eggs and hatchlings have some exposure to predation, the species compensates for a certain level of natural predation by producing a large number of eggs and hatchlings. For this DPS, the primary impact is to productivity (*i.e.*, reduced egg and hatching success). We conclude that predation poses a threat to the SE Atlantic DPS.

*Inadequacy of Existing Regulatory Mechanisms*

The SE Atlantic DPS is protected by various regulatory mechanisms. For each, the Team reviewed the objectives of the regulation and to what extent it adequately addresses the targeted threat.

The harvest of turtles and eggs is illegal in most of the nations where the DPS nests. In some cases, however, these protective mechanisms are inadequate. In addition, many nesting beaches are not protected.

In Gabon, the harvest of turtles and eggs is illegal (2011 decree 0164/PR/MEF) and much of the nesting beach habitat (and turtles utilizing that habitat) is protected because of inclusion in parks as well as being far from any city or town. However, low levels of poaching occurs, and the threats from encroaching development and associated impacts are increasing.

In Congo, wildlife laws prohibit the hunting and collection of wildlife and their products, including eggs, between November 1 and April 31. Turtles are also protected in the Conkaouati-Douli National Park. However, in areas without permanent beach monitoring, almost all eggs and nesting individuals are collected and eaten (Bal *et al.* 2007).

In the Democratic Republic of Congo, leatherback turtles are cited under the 1982 Hunting Act for protection. However, there is no post-independence legislation protecting sea turtles, and there is little commitment to the legislated protections (Fretey 2001).

Since 1988, Equatorial Guinea has protected all sea turtles under Law 8/1988 and Decree 183/87 on fishing (Tomás *et al.* 2010). However, the poaching of eggs and females for local consumption and sale has occurred (Castroviejo *et al.* 1994).

In Ghana, the Wildlife Regulations Act of 1974 prohibits all harvest of eggs and turtles. However, poverty is prevalent, and eggs and sea turtles are poached at nesting beaches (Tanner 2013). Enforcement is likely inadequate because of funding issues, the remoteness of some nesting beaches, and cultural practices.

Fishery bycatch is the primary threat to this DPS. While most nations in the region have some form of legal protection for sea turtles, many leatherback turtles die from fisheries bycatch throughout the range of the DPS. Examples of fisheries legislation include Brazil's gear restrictions and Nigeria's requirement to use TEDs in bottom trawls.

In summary, numerous regulatory mechanisms provide some protection to leatherback turtles, their eggs, and nesting habitat throughout the range of this DPS. Though the regulatory mechanisms provide some protection to the turtles, many do not adequately reduce the threat that they were designed to address, generally as a result of limited implementation or enforcement. Fisheries bycatch, poaching, and habitat loss remain major threats to the DPS despite regulatory mechanisms. We conclude that inadequacy of the regulatory mechanisms are a threat to the SE Atlantic DPS.

#### *Fisheries Bycatch*

Fisheries bycatch is the primary threat to the SE Atlantic DPS. Leatherback turtles are captured as bycatch in commercial and artisanal fisheries along coastal foraging and breeding areas as well as on the high seas. Because of the overlapping range with the SW Atlantic DPS, this DPS is vulnerable to interactions with fisheries off the coasts of Brazil, Uruguay, and Argentina, in the pelagic waters of the South Atlantic Ocean, and along the

coastal waters off western Africa. Therefore, the information presented on the fisheries bycatch for the SW Atlantic is applicable to this DPS.

One of the biggest threats for leatherback turtles in Atlantic waters is bycatch in artisanal and commercial fisheries (Wallace *et al.* 2010; Riskas and Tiwari 2013; ). Lewison *et al.* (2004) estimated that 30,000 to 60,000 leatherback turtles were taken as longline fisheries bycatch in the entire Atlantic Ocean in 2000. Stewart *et al.* (2010) estimated that in West Africa, Benin, Togo, and Cameroon had the highest average fishing densities, ranging from 11.1 to 6.5 boat-meters/km<sup>2</sup>, and gillnet densities ranked among the highest on a global scale. Despite very active artisanal and industrial fisheries in the region, overall bycatch data are quite sparse or qualitative (rather than quantitative) in nature, and Africa still represents a significant gap in bycatch evaluation studies (Wallace *et al.* 2010, 2013). Accurate and reliable bycatch data are difficult to achieve, as direct observation rates are low (<1 percent of total fleets) and statistics from the region's many small-scale fisheries are largely incomplete (Kelleher 2005; Moore *et al.* 2010; Wallace *et al.* 2010). However, several studies have concluded that bycatch rates in the region are high, given the degree of fishing activity near nesting and foraging areas (Lewison *et al.* 2004; Moore *et al.* 2010; Wallace *et al.* 2010).

Along the coasts of Angola, Namibia, and South Africa, Honig *et al.* (2007) evaluated turtle bycatch by longline fisheries in the Benguela Large Marine Ecosystem by using data from observer reports, surveys, and specialized trips from the coastal nations of South Africa, Namibia and Angola. They estimated bycatch at 672 leatherback turtles annually (based on an annual bycatch estimate of 4,200 turtles, of which approximately 16 percent are leatherback turtles) in the southern and central regions and

as many as 5,600 leatherback turtles (based on an annual bycatch estimate of 35,000 turtles) for the entire Benguela Large Marine Ecosystem (Honig *et al.* 2007). Mortality rates were not provided in this study but may range from 25 to 75 percent (Aguilar *et al.* 1995). The estimates mostly include turtles from the SE Atlantic DPS, but telemetry studies indicate that the turtles of the much smaller SW Indian DPS also use this foraging area (Luschi *et al.* 2006; Robinson *et al.* 2016). Evaluating ICCAT data, Angel *et al.* (2014) confirm exposure to high longline fishing effort and some purse seine effort for the population originating from the SE Atlantic Ocean.

The limited bycatch data available for waters of the western coast of Africa show that other fisheries interact with leatherback turtles. Between 2005 and 2015, artisanal fishing nets in Loango Bay in the Republic of Congo killed a total of 45 leatherback turtles; 0 to 628 leatherback turtles were captured or recaptured annually over that time period (Bréheret *et al.* 2017). An assessment of bycatch in the trawling fisheries in Gabon found that leatherback turtles represented only 2 percent of the bycatch despite being the most abundant sea turtle species in Gabonese waters; the low rate is possibly because leatherback turtles do not occur in the section of the water column where the trawl net is towed (Casale *et al.* 2017). Trawl bycatch in the waters around São Tomé and Príncipe included 4 juvenile leatherback turtles (17 to 21 cm in carapace length) in March 1994 (Fretey *et al.* 1999).

While specific information to estimate overall capture and mortality rates of SE Atlantic leatherback turtles in fisheries is not available, it is clear that bycatch in fisheries, especially gillnets and longlines, are a threat to the DPS across its range. Immature and mature individuals are exposed to high fishing effort throughout their foraging range and

in coastal waters near nesting beaches. Mortality is also high. Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when nesting females are incidentally captured and killed. We conclude that fisheries bycatch is a major, and the primary, threat to the SE Atlantic DPS.

#### *Vessel Strikes*

There is little information regarding vessel strikes for the SE Atlantic DPS, but such interactions are a potential, and possibly increasing, threat across at least a portion of this DPS's range. In the western South Atlantic foraging grounds off Brazil, Uruguay, and Argentina, increasing vessel traffic from fishing vessels, cargo transport, and tourism has been noted (López-Mendilaharsu *et al.* 2009; Fossette *et al.* 2014), potentially increasing the likelihood of vessel strikes on leatherback turtles. Although no specific information is available for the waters off western Africa, any economic development along the coast is likely to result in an increase in vessel traffic. We conclude that vessel strikes are a threat to the SE Atlantic DPS.

#### *Pollution*

The SE Atlantic DPS faces the threat of pollution across its extensive range throughout the South Atlantic Ocean, from Africa to South America. As the ranges of the SW Atlantic and SE Atlantic DPSs overlap, they are exposed to the same pollutants, which include contaminants, marine debris, and ghost fishing gear. Throughout Africa, marine and coastal pollution is widespread in industrial and urban areas, and garbage litters many developed beaches (Formia *et al.* 2003; Agyekumhene *et al.* 2017). Off the coast of South America, the Argentine and Brazilian coastal waters are increasingly impacted by economic activities, such as maritime cargo transport, tourism, and the

discharge of domestic and industrial waste (López-Mendilaharsu *et al.* 2009; Fossette *et al.* 2014).

The Gulf of Guinea has increasingly been the focus of extensive oil exploitation activities, following the discovery of large oil reserves. Drilling activities by large oil corporations, with associated pollution and habitat destruction, are threats to nesting aggregations in the area (Formia *et al.* 2003; Agyekumhene *et al.* 2017). In 2012/2013, oil spills following the dredging of the Port of Pointe-Noire in the Republic of Congo significantly degraded the fauna and flora of Loango Bay, where leatherback turtles occur. However, the ecosystem is believed to be slowly recovering (Bréheret *et al.* 2017). In 2005, a moderate slick of oil on the beaches of Mayumba National Park in Gabon was observed, although its impacts on turtles are unknown (Parnell *et al.* 2007).

In Nigeria, the main sources of pollution include industrial waste, raw/untreated sewage, and pesticides. Oil exploration, exploitation, and transportation have a significant effect on the environment. Spills of crude and refined oil are frequent in the coastal and marine environment, especially during periods of very strong ocean currents, when they can spread to cover the entire 853 km coastline of Nigeria.

It is clear that individuals from the SE Atlantic DPS have a high probability of encountering pollution across their range and throughout their lifecycle. Although the best available information does not quantify such impacts, ample information demonstrates that these threats are ongoing. We conclude that pollution is a threat to the DPS.

*Climate Change*

Climate change is a threat to the SE Atlantic DPS. The impacts of climate change include: increases in temperatures (air, sand, and sea surface); sea level rise; increased coastal erosion; more frequent and intense storm events; and changes in ocean currents.

Sea level rise resulting from climate change negatively impacts sea turtle nesting. Erosion of important nesting beaches in Gabon may be at least partially attributable to sea level rise. From 1983 through the 2000s, some areas have lost up to 100 m of beach width, reducing the availability of suitable nesting beach (Gabon Sea Turtle Partnership 2018; <http://www.seaturtle.org/groups/gabon/erosion.html>). Because leatherback turtles nest lower on the beach than other sea turtles, their eggs are more at risk of being inundated and destroyed by increases in sea level and coastal erosion (Boyes *et al.* 2010).

Changes in sand temperatures are likely to impact egg viability and sex determination. Ikaran (2010) found the thermal range of sand over the nesting season to be adequate for hatchling sex ratios to be mixed or even male dominated. In Gabon, the early rainy months tend to produce males, while the later, warmer months produce females, with a tendency towards a net higher production of males. Ikaran (2010) considered the nesting beaches of Gabon to be an important male producing area. However, based on predictions of warming trends, he found that within two decades the ratio could skew towards 100 percent female.

The threat of climate change is likely to modify the nesting conditions for turtles of the DPS, and it is unclear whether they have or can develop the ability to nest in different locations along existing beaches, or on new beaches. Impacts from climate change are likely to range from small, temporal changes in nesting season to large losses of productivity. Therefore, we conclude that climate change is a threat to the DPS.

### *Conservation Efforts*

There are numerous efforts to conserve the leatherback turtle. The following conservation efforts apply within the range of the SE Atlantic DPS (for a description of each effort, please see the section on conservation efforts for the overall species): Convention on the Conservation of Migratory Species of Wild Animals, Convention on Biological Diversity, Convention on International Trade in Endangered Species of Wild Fauna and Flora, Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention), FAO Technical Consultation on Sea Turtle-Fishery Interactions, IAC, MARPOL, IUCN, Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa, Ramsar Convention on Wetlands, South-East Atlantic Fisheries Organization, UNCLOS, and UN Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing. Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce its risk of extinction.

### *Extinction Risk Analysis*

After reviewing the best available information, the Team concluded overall that the SE Atlantic DPS is at high risk of extinction. The total index of nesting female abundance is 9,198 females. Since 2002, the first year that aerial survey data was collected, nesting activity has declined by –8.6 percent annually in Gabon, the largest nesting aggregation of the DPS, and what was, in 2002, the largest nesting aggregation in the world. This declining trend has the potential to further lower abundance and increase the risk of extinction. Nesting and foraging is broadly distributed; thus, the population is somewhat buffered from stochastic events that could otherwise have catastrophic effects

on the entire DPS. There is a metapopulation structure within this DPS, with fine-scale genetic differentiation between Gabon and Ghana. Genetic diversity also appears to be moderate. Based on the reduced nesting female abundance and declining nest trend, we find the DPS to be at risk of extinction, likely as a result of past threats.

Current threats place the DPS at further risk of extinction. The primary threat to this DPS is bycatch in commercial and artisanal, pelagic and coastal, fisheries, especially coastal gillnet and pelagic longline fisheries. Fisheries bycatch reduces abundance by removing individuals from the population. Because several fisheries operate near nesting beaches, productivity is also reduced when nesting females are prevented from returning to nesting beaches. Thus, exposure and impact of this threat are high. Habitat loss or modification is a threat that reduces abundance and productivity and includes the impacts of logs, which block access to the beaches or trap nesting females and hatchlings. Poaching of turtles and eggs is also a threat to this DPS, although most nesting beaches in Gabon are somewhat protected because they occur in parks or are far from any towns. Many of the beaches outside Gabon (*e.g.*, Guinea-Bissau) have limited or no protection. The degree of overutilization is highly varied across locations, but quite extensive in some areas. Funding from the MTCA has resulted in some reduction of this threat as conservation activities, research, and community involvement results in lower poaching on those beaches. However, poaching continues at high levels in other areas. Additional threats include: predation and disease, inadequate regulatory mechanisms, pollution, and climate change. Predation can be extensive at some specific beaches, but overall it does not occur at a high level. Pollution is a persistent and potentially increasing threat. Ingestion of plastics and entanglement in marine debris result in injury and reduced

health, and sometimes mortality. Climate change is likely to result in reduced productivity due to greater rates of coastal erosion and nest inundation, and in some areas, nest failure or skewed sex ratios due to increased sand temperatures. Vessel strikes are a threat that is likely to increase over time as recreational and commercial vessel activity increases, resulting in more opportunity for interactions. Though many regulatory mechanisms are in place, they do not adequately reduce the impact of logs, poaching, and fisheries. Additionally, many areas in the region have little or no enforcement of laws protecting turtles or nests on the beach.

The DPS is relatively data-poor, reducing our ability to quantify threats for more than a small portion of the population. For this reason, the Status Review Team did not come to consensus regarding the extinction risk analysis for the SE Atlantic DPS. All Team members were present to vote on the level of extinction risk. Nine Team members concluded with moderate confidence that the DPS is at high extinction risk due to threats and loss of abundance; their confidence was moderate due to the lack of data on this DPS. Two team members concluded with low confidence that the DPS is at moderate extinction risk; their confidence in this conclusion is low due to the lack of data on this DPS.

We conclude, consistent with the Team's overall conclusion, that the SE Atlantic DPS is currently in danger of extinction. The decreasing nesting trend (*i.e.*, 8.6 percent annually since 2002) is at or near a level that make the DPS highly vulnerable to threats, given the total index of nesting female abundance of 9,198 females. It faces present, ongoing threats that are likely to create imminent and substantial demographic risks (*i.e.*, declining trends and reduced abundance). Though numerous conservation efforts apply

within the range of this DPS, they do not adequately reduce the risk of extinction. We conclude that the SE Atlantic DPS is currently in danger of extinction throughout its range and therefore meets the definition of an endangered species. The threatened species definition does not apply because the DPS is at risk of extinction currently (*i.e.*, at present), rather than on a trajectory to become so in the foreseeable future.

### **SW Indian DPS**

The Team defined the SW Indian DPS as leatherback turtles originating from the SW Indian Ocean, north of 47° S, east of 20° E, and west of 61.577° E. The western boundary occurs at the southern tip of Africa, approximately where the Agulhas and Benguela Currents meet. The eastern boundary occurs at the border between Iran and Pakistan, where the Somali Current begins. These currents, and the cold waters of the Antarctic Circumpolar Current, likely restrict the nesting range of this DPS.

The range of the DPS (*i.e.*, all documented areas of occurrence) extends into the SE Atlantic Ocean, where leatherback turtles forage in the highly productive Benguela Current Large Marine Ecosystem, which occurs along the western coast of Africa, from Angola to South Africa. Leatherback turtles also range throughout the waters of eastern Africa (Ross 1985) and possibly into the Red Sea (Gasparetti *et al.* 1993). Records indicate that the species has been observed in the waters of the following nations: Djibouti; Eritrea; French Territories (Reunion Island, Mayotte, and Iles Eparses); Kenya; Madagascar; Mozambique; Seychelles; Somalia; South Africa; Tanzania; and Yemen (Hamann *et al.* 2006). Leatherback turtles may occur in the waters of the following nations: Bahrain, Kuwait; United Arab Emirates; Oman; and Sudan (Hamann *et al.* 2006).

Leatherback turtles of the SW Indian DPS nest over a distance of approximately 900 km, from Cape Vidal, South Africa to Bazaruto Islands, Mozambique (Videira *et al.* 2011; Nel *et al.* 2015). The vast majority of nesting (80 to 90 percent) occurs in South Africa, between Bhanga Nek and Leifeld's Rock (Nel *et al.* 2015). In Mozambique, most nesting occurs from the southern border to Inhaca Island, Mozambique, with low levels of nesting farther north at Bilene Beach and Bazaruto Islands (Nel *et al.* 2015). This DPS nests at the highest latitude (and southernmost location) of all leatherback turtles (Saba *et al.* 2015).

Nesting occurs on long (5 to 15 km), broad (50 to 100 m), silica sand beaches with little vegetation (Botha 2010; Nel *et al.* 2015; Robinson *et al.* 2017). The beaches are characterized by pristine, intact dunes that rise up to 100 m above sea level, interspersed with a few dynamic dunes and small, primary dunes (Nel *et al.* 2015). The beaches are separated by short rocky headlands (Robinson *et al.* 2017). Subtidal rock formations are dispersed throughout the high energy coastline. Nesting females approach the beach using strong rip-currents through obstruction-free areas (Hughes 1974; Hughes 1996; Botha 2010; Nel *et al.* 2015).

Foraging areas of the SW Indian DPS include coastal and pelagic waters of the SW Indian Ocean and the SE Atlantic Ocean. The DPS is somewhat unique in that turtles forage in two ocean basins and do not need to undergo long migrations between nesting and foraging areas because highly productive foraging areas are available adjacent to nesting beaches or connected to nesting beaches via fast-moving currents. For example, the warm, fast-flowing Agulhas Current (Lutjeharms 2001; Nel *et al.* 2015) results in high productivity foraging areas near nesting beaches and provides a migratory corridor

to distant foraging areas. As a result, the SW Indian turtles have the largest body size, largest clutch size, and highest reproductive output of all leatherback turtles (Saba *et al.* 2015).

Satellite tracking of post-nesting females (n = 27) reveals the use of one of three post-nesting migratory corridors: north into the nearby coastal waters of the Mozambique channel; south and west (via the Agulhas and Benguela Currents) into the pelagic waters of the South Atlantic Ocean; or south and east (via the Agulhas Current and Retroflection) into the oceanic eddies in the SW Indian Ocean (Luschi *et al.* 2006; Robinson *et al.* 2016; Harris *et al.* 2018). Luschi *et al.* (2006) reviewed satellite telemetry data of 11 post-nesting females tagged between 1996 and 2003 (Hughes *et al.* 1998; Luschi *et al.* 2003; Sale *et al.* 2006); and Robinson *et al.* (2016) satellite tracked 16 post-nesting females tagged between 2011 and 2013. Evaluating tracking data for 14 post-nesting females between 2006 and 2014, Harris *et al.* (2018) found that leatherback turtles equally used all three migration corridors. In the other studies, a total of 11 post-nesting females migrated a relatively short distance (approximately 500 km) to the shallow (less than 50 m depth), coastal waters of the Sofala Banks (*i.e.*, the Mozambique Channel), where net primary productivity and sea surface temperatures remain elevated year-round (n = 4, Sale *et al.* 2006; n = 7, Robinson *et al.* 2016). One post-nesting female migrated to the similarly hospitable coastal waters of Madagascar (Robinson *et al.* 2016). Ten post-nesting females tracked to pelagic waters of the Atlantic Ocean (n = 6, Sale *et al.* 2006; n = 4, Robinson *et al.* 2016). These waters are among the most productive in the world, as a result of strong upwelling (caused by the southeast trade winds) and the area's unique bathymetry, hydrography, chemistry, and trophodynamics (Honig *et al.* 2007).

Five post-nesting females appeared to track oceanic eddies into the SW Indian Ocean (n = 1, Sale *et al.* 2006; n = 4, Robinson *et al.* 2016). Luschi *et al.* (2003 and 2006) characterized leatherback turtles using this latter strategy as “wanderers, ranging over vast oceanic areas while searching for their planktonic prey.” Opportunistically encountered and highly productive eddies likely shaped the circuitous routes of these foraging turtles, which resemble drifters more than active swimmers (Luschi *et al.* 2006; Robinson *et al.* 2016; Harris *et al.* 2018). Thus, this DPS benefits from the use of three migratory corridors that all provide highly productive foraging opportunities, with minimal energetic cost required to return to waters off nesting beaches.

### *Abundance*

The total index of nesting female abundance of the SW Indian DPS is 149 females. We based this index on two nesting aggregations: South Africa (Ezemvelo KwaZulu-Natal Wildlife (Ezemvelo), unpublished data, 2018) and Mozambique (Centro Terra Viva Estudos e Advocacia Ambiental (CTV), unpublished data, 2018). Our total index does not include two unquantified nesting aggregations in Mozambique. To calculate the index of nesting female abundance (*i.e.*, 134 females) for the South Africa “monitoring area” (*i.e.*, a 52.8 km stretch of beach that has been monitored for decades), we divided the total number of nests between the 2014/2015 and 2016/2017 nesting seasons (*i.e.*, a 3-year remigration interval; Hughes 1996; Lambardi *et al.* 2008; Nel *et al.* 2013; Saba *et al.* 2015) by the clutch frequency (7 clutches/season; Nel *et al.* 2013; Saba *et al.* 2015). To calculate the index of nesting female abundance in Mozambique (*i.e.*, 15 females), we divided the total number of nests between the 2015/2016 and 2017/2018

nesting seasons (*i.e.*, a 3-year remigration interval) by the clutch frequency for South Africa (7 clutches/season; Nel *et al.* 2013; Saba *et al.* 2015).

This is an index for the DPS because it only includes available data from recently and consistently monitored nesting beaches. While nesting occurs on beaches that stretch across 900 km of South Africa and Mozambique, consistent and standardized monitoring occurs only across approximately 300 km of beaches across the two nations (Nel *et al.* 2013; Nel *et al.* 2015). Furthermore, while nesting is known to occur at low levels at Inhaca Island and Bazaruto Archipelago in Mozambique, we did not include these sites because we did not have data from the most recent 3 years.

Other estimates of total or annual nesting female abundance have been published. The IUCN Red List assessment estimated the total number of mature individuals (males and females) at 148 individuals, based on an average of 259 annual nests (Nel *et al.* 2013), a 3-year remigration interval (Nel *et al.* 2013), and a 3:1 sex ratio (Wallace *et al.* 2013). Their estimates are based on nesting surveys conducted in South Africa, which hosts approximately 80 to 90 percent of nesting, and Mozambique (Wallace *et al.* 2013; Nel *et al.* 2015). Their estimate is less than our index, despite including mature males and females. The reason for this difference is because they used an average annual number of nests that was lower than recent nest counts over the 3-year remigration interval. Nel *et al.* (2015) estimated the size of the total nesting population at approximately 100 females per season (Nel *et al.* 2015), based on 2010 data: 375 emergences and 336 nests in South Africa; and 61 emergences in Mozambique (Videira *et al.* 2011). This estimate ( $n = 300$ , based on a 3 year remigration interval) is greater than our index because there were more nests in 2010 compared to more recent years (2014 to 2016). Hamann *et al.* (2006)

estimated approximately 20 to 40 nesting females annually in South Africa and approximately 10 nesting females annually in southern Mozambique. This estimate ( $n = 90$  to 150, based on a 3 year remigration interval) is less than our index, likely as a result of using data collected over a different time-frame. The difference in estimates likely results from using different methods of calculation and different time frames and reflects some uncertainty in the precise number of nesting females. Our total index of nesting female abundance falls within the range of other estimates and is based on the best available data for the DPS at this time.

There are additional published estimates for the South Africa monitoring area. Nel *et al.* (2013) identified 2,578 nesting females over 45 years (1965 to 2009), with a mean of  $69.4 \pm 38.1$  nesting females per season (or 209 total nesting females) in the monitoring area. Hughes (1996) reported an annual average of 24 nesting females in the first decade (1976 to 1985) and an annual average of 86 nesting females in the second decade (1986 to 1995) in the monitoring area. Hughes (1996) also reported an annual average of 113 nesting females from 1986 to 1995 in an extended protected area that includes the monitoring area plus another 93 km in the St. Lucia Marine Reserve, which is surveyed periodically. The difference between these two averages reflects that most estimates of nesting female abundance in South Africa are minimum estimates because nesting occurs outside the monitoring area. Thorson *et al.* (2012) found that annual resightings for leatherback turtles decreased from the 1960s to 2009, and their modeling indicated that this decline was due to decreased detection probabilities (*i.e.*, decreased probability of returning to the monitored portion of the KwaZulu-Natal nesting beach), rather than decreased survival. Based on satellite tracking of 17 post-nesting females, Harris *et al.*

(2015) estimates that approximately 66 percent of leatherback nesting activity occurs outside the monitoring area. However, considerable inter-annual variability exists, ranging from less than 30 percent to over 80 percent, with a median of approximately 49 percent (Harris *et al.* 2015). Thus, incomplete beach monitoring is a source of uncertainty for this DPS and for our total index of nesting female abundance.

For Mozambique, our index of nesting females is similar to other published estimates, which are generally less than 20 nesting females (Hamann *et al.* 2006; Louro 2014; Pereira *et al.* 2014; Fernandes *et al.* 2018). If we use the clutch frequency for Ponta Malongane (2.25 clutches per season; Louro *et al.* 2006), which is low for the species, our index of nesting female abundance is 45 females. This clutch frequency may be underestimated due to females nesting in distant areas where monitoring does not regularly occur. If we use the clutch frequency for South Africa, (7 clutches/season; Nel *et al.* 2013; Saba *et al.* 2015), the resulting index of nesting female abundance for Mozambique (*i.e.*, 15 nesting females) is closer to published estimates.

The total index of nesting female abundance of 149 females places the DPS at risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb losses in individuals. Due to its small size, the DPS has restricted capacity to buffer such losses. Given the intrinsic problems of small population size, we conclude that the limited nesting female abundance is a major factor in the extinction risk of this DPS.

*Productivity*

The SW Indian DPS exhibits a slightly decreasing nesting trend. We base our conclusion on data consistently collected in a standardized approach in the 56 km South African monitoring area (Ezemvelo, unpublished data, 2018), where nest counts decreased by  $-0.3$  percent annually (sd = 2.1 percent; 95 percent CI =  $-4.5$  to 4.1 percent;  $f = 0.557$ ; mean annual nests = 301) between the 1973/1974 and 2016/2017 nesting seasons. The trend in South Africa is likely representative of the entire DPS, as 80 to 90 percent of nesting is estimated to occur there (Wallace *et al.* 2013; Nel *et al.* 2015) and the 44-year time series is quite robust.

Our trend estimates yield similar results to other published findings for the population. The IUCN concluded that this population has declined slightly, by 5.6 percent over the past three generations, with an annual decline of  $-0.1$  percent in South Africa and  $-0.7$  percent in Mozambique (Wallace *et al.* 2013). Hamann *et al.* (2006) also identified a declining trend in the nesting population of the SW Indian Ocean. Studies focused on the South African monitoring area (*i.e.*, the source of data for our trend analysis), however, disagree on whether the trend has declined recently (Hamann *et al.* 2006; Nel *et al.* 2013) or is stable (Nel *et al.* 2015; Saba *et al.* 2015). The nest trend may be stable if nesting in unmonitored areas has increased over time (Thorson *et al.* 2012; Harris *et al.* 2015). Different datasets lead to different conclusions due to different methods of calculation, different time frames, incomplete monitoring of all nesting areas, and therefore uncertainty in the precise number of nesting females. We find that Nel *et al.* (2013) provide the best available published data, which are based on the most recent, primary data, and we agree with their characterization of the trend as declining or recently declining.

Despite the recent decline in nesting, productivity parameters remain relatively high for the SW Indian DPS, which has the largest body size, largest clutch size, and highest reproductive output of all leatherback turtles, likely due to the close proximity between their nesting beaches and highly productive foraging areas (Saba *et al.* 2015). Nel *et al.* (2015) reports that most metrics (*i.e.*, female size, egg size, incubation time, and hatching success) are above average for this DPS. Nesting females produced 1,171 to 53,139 hatchlings each season in the South Africa monitoring area between 1965 and 2009, with an average of 36,583 to 51,610 hatchlings per season, which was calculated by multiplying 480 hatchlings per nesting female by  $69.4 \pm 38.1$  nesting females per season (Nel *et al.* 2013).

The recent nesting decline may reflect the effects of past and current threats that overwhelm the population's high productivity metrics. We conclude that the slightly declining nest trend places the DPS at risk of extinction, which is further exacerbated by the limited nesting female abundance.

#### *Spatial Distribution*

The SW Indian DPS comprises, in essence, a single nesting aggregation, with nesting females moving freely between South African and Mozambican beaches (Hughes 1996; Luschi *et al.* 2006; Nel *et al.* 2015). Nesting is limited to a total distance of approximately 900 km along South African and Mozambican coasts (Nel *et al.* 2015). While 80 to 90 percent of nesting is concentrated in South Africa, nesting is somewhat concentrated in the southern section of the South African monitoring area, although most characterize nesting as low density throughout South Africa (Hughes 1974; Lambardi *et al.* 2008; Botha 2010; Nel *et al.* 2013; Harris *et al.* 2015; Nel *et al.* 2015).

The DPS exhibits a broad foraging range that extends into coastal and pelagic waters of the eastern Atlantic and western Indian Oceans (Luschi *et al.* 2006; Lambardi *et al.* 2008; Girondot 2015). There is limited evidence that leatherback turtles may remain in South African waters throughout the year, as suggested by year-round fisheries bycatch records (Luschi *et al.* 2003, 2006; Petersen *et al.* 2009). Some forage off the coast of Madagascar (Robinson *et al.* 2016; Harris *et al.* 2018). Some turtles follow the Agulhas and Benguela Currents into foraging areas in the southeast Atlantic Ocean, off the coasts of Angola and Namibia (Girondot 2015; Robinson *et al.* 2016; Harris *et al.* 2018). Others follow the Agulhas Retroflexion and deep-sea eddies into the SW Indian Ocean (Luschi *et al.* 2006; Lambardi *et al.* 2008; Robinson *et al.* 2016; Harris *et al.* 2018). Leatherback turtles, possibly from this DPS, have also been observed in the Red Sea, presumably foraging (Hamann *et al.* 2006). The use of various foraging areas may be influenced by the prevalent currents encountered off the nesting beaches (Luschi *et al.* 2006; Lambardi *et al.* 2008; Robinson *et al.* 2016).

The wide distribution of foraging areas likely buffers the DPS somewhat against local catastrophes or environmental changes that would limit prey availability. Nesting occurs along one coastline, which is 3,000 km in length and may be similarly affected by environmental variation and directional changes (*e.g.*, sea level rise). Because the DPS is essentially a single nesting aggregation, it has limited capacity to withstand other catastrophic events. Thus, spatial distribution likely has little net effect on the extinction risk of the SW Indian DPS.

*Diversity*

Within the SW Indian DPS, genetic diversity is low, with only two mtDNA haplotypes found in 41 nesting females in South Africa (haplotype diversity =  $0.298 \pm 0.078$  and nucleotide diversity =  $0.0004 \pm 0.0004$ ; Dutton *et al.* 2013). Nesting habitat is mainly restricted to beaches along the same coast, with a few nests on Mozambican islands. The DPS does not exhibit temporal or seasonal nesting diversity, with most nesting occurring between October and March. The foraging strategies are diverse, however, with turtles using coastal and pelagic waters in the Atlantic and Indian Oceans. Diverse foraging strategies may provide some resilience against local reductions in prey availability or catastrophic events, such as oil spills, by limiting exposure. Low genetic diversity indicates the DPS may lack the raw material necessary for adapting to long-term environmental changes, such as cyclic or directional changes in ocean environments due to natural and human causes (McElhany *et al.* 2000; NMFS 2017). We conclude that limited overall diversity increases the extinction risk of this DPS by reducing its resilience to threats.

*Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

Coastal erosion, foot and vehicle traffic, and artificial lighting modify the available, suitable nesting habitat and thus are threats to the SW Indian DPS. Angel *et al.* (2014) identifies coastal erosion as the main beach-based threat to this population and one that is likely to increase with climate change.

Coastal erosion removes sand from nesting beaches, inundating nests and destroying eggs. Because leatherback turtles nest lower on the beach than other sea turtles, they have greater exposure to tidal erosion and deposition (Boyes *et al.* 2010). At South African nesting beaches over a duration of 70 days, Boyes *et al.* (2010) found an

average of 0.62 m deposition (S.D. 0.15 m; range 0.34–0.85 m) and 0.42 m erosion (S.D. 0.17 m; range 0.14–0.71 m). Because the average depth of leatherback nests was 0.66 m (S.D. 0.19 m; range 0.15–1.07 m), eggs are at some risk of being exposed and destroyed (Boyes *et al.* 2010). Nel *et al.* (2006) concludes that coastal erosion is a threat in South Africa, where the high-energy coastline varies seasonally. During two nesting seasons (2009/2010 and 2010/2011), de Wet (2012) found that 6.3 percent of nests in the South African monitoring area were destroyed by erosion. In Bazaruto Archipelago, Mozambique, coastal erosion and rising sea levels destroyed approximately 12 percent of nests over 10 seasons of monitoring (Videira and Louro 2005; Louro 2006). Despite nest loss due to erosion, hatching success remains high in South Africa (70 to 80 percent; Nel *et al.* 2015; Santidrián Tomillo *et al.* 2015). Though the introduction of *Casuarina* trees do not necessarily increase the risk of erosion, they obstruct nesting females' access to and from beaches and alter nest incubation environments (de Vos *et al.* 2019). Evolving in a high-energy coastline environment with seasonal variation has likely provided the DPS with some resilience to nesting losses due to coastal erosion. Sea level rise as a result of climate change, however, is likely to increase the rate and magnitude of this natural process.

In Mozambique, Louro (2006) describes beach driving as a “very serious problem.” Tourism and beach driving are increasing in Ponta Malongane and Bazaruto Island, nesting areas in Mozambique, where there is no legislation regarding beach driving (Louro 2006). Foot and vehicular traffic, for tourism and recreational purposes, have been found to impact nesting beach habitat and turtles in several ways. Beach activities can deter females from using a nesting beach. Beach driving causes sand

compaction, which may lower nest success. It also creates ruts that slow hatchlings' crawl to the surf, increasing their vulnerability to predators. Beach driving occurs to a lesser extent in South Africa. Recreational beach driving is allowed on a 1.5 km stretch of beach, and tourism driving (for concession, management, and media) involves a maximum of 10 vehicles per night across 40 km of beach (Nel 2006).

Artificial lighting modifies the quality of nesting beaches because lights over land disorient nesting females and hatchlings. Instead of crawling toward the surf and their marine habitat, they crawl further inland, where they may become dehydrated and die or become susceptible to predation. Within the 280 km of coastline within the iSimangaliso Wetland Park, South Africa, there are only four areas of less than 100 m each that contain artificial lighting (Nel 2006). We were unable to find data on artificial lighting in Mozambique.

The majority of nesting habitat occurs within the 280 km coastline of the iSimangaliso Wetland Park in South Africa, which has been a World Heritage Site since 1999 (UN Educational, Scientific and Cultural Organization 1999; Hughes 2010; Robinson *et al.* 2016). From 1979 to 1999, much of the nesting habitat and nearshore marine habitat was protected, first as the St. Lucia Marine Reserve, then the Maputaland Marine Reserve (Hughes 1996). Such protections contributed to the prevention of dredging a deep water harbor through turtle nesting beaches and mining heavy minerals in the adjacent dunes (Hughes 2009, 2010). In Mozambique, the Ponta do Ouro Partial Marine Reserve has provided beach and marine habitat protection since 2009. Additional protection is provided to Mozambican nesting beaches in: the Ponta do Ouro – Kosi Bay Transfrontier Marine Conservation Area; the Maputo Special Reserve; the Bazaruto

Archipelago National Park; and the Quirimbas Archipelago National Park. However, nest protection only occurs over nine percent of the Mozambique coastline (Videira *et al.* 2008; Garnier *et al.* 2012). Such protections have minimized vehicular traffic at nesting beaches in South Africa, but beach driving remains a threat in Mozambique. Erosion is a threat to nesting beaches in both South Africa and Mozambique. Thus, we conclude that the present modification of nesting habitat is a threat to the SW Indian DPS.

#### *Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

Overutilization is a threat to the SW Indian DPS (Bourjea 2015; Williams *et al.* 2016; Williams 2017). Two of nine leatherback turtles equipped with satellite tags between 1996 and 2006 were incidentally or intentionally captured in Mozambique and Madagascar and likely retained for food or sale (de Wet 2012). In Mozambique, eggs and turtles were once legally harvested and are now illegally poached for consumption (Nel 2012; Wallace *et al.* 2013; Fernandes *et al.* 2018). Turtle poaching includes turtles taken on the beaches and at sea (Williams *et al.* 2016; Williams 2017). We do not have recent, quantitative estimates of egg or turtle poaching in Mozambique. However, significant usage has been documented at various points in time. Hughes (1995) reported that nearly every nesting female was killed during the civil war (1977 to 1992). An estimated 32 loggerhead and leatherback turtles were killed at Ponta Malongane in 11 years (Louro 2006). Recent egg and turtle poaching rates in Mozambique have been qualitatively described as “alarming,” “significant,” “widespread,” “prominent,” and “prevalent” (Fernandes *et al.* 2015; Williams *et al.* 2016; Williams 2017; Pereira and Louro 2017; Fernandes *et al.* 2017; Fernandes *et al.* 2018). Nest monitoring programs in Mozambique have provided some protection since the 1990s (Garnier *et al.* 2012). Pereira *et al.* (2014)

reports that as a result of the monitoring program at the Ponta do Ouro Partial Marine Reserve, where the majority of nesting in Mozambique occurs, turtle mortalities are very rare. Egg poaching has been reduced in the Bazaruto Archipelago, where it was previously prevalent (Louro 2006). National legislation in Mozambique include: Diploma Legislativo 2627 (7 August 1965), Forest and Wildlife Regulation (Decree 12/2002 of 6 June 2002) and Conservation Law (Law 5/2017 of 11 May). These laws protect turtles and eggs and impose fines for poaching or possession. However, the laws are poorly implemented and enforced (Costa *et al.* 2007; Louro 2006; Williams *et al.* 2016; Fernandes *et al.* 2018). We conclude that the poaching of turtles and eggs remains a significant threat in Mozambique.

Poaching of turtles is also a threat in Madagascar, where leatherback turtles caught in gillnets are taken back to local villages and consumed, which is documented to have occurred twice in 2016 (Williams 2017). Leatherback turtles were caught and consumed or sold in five of eight Malagasy villages surveyed between October 2004 and March 2004. Fishers reported that leatherback turtles were uncommon but large, possibly indicative of mature individuals (Walker and Roberts 2005). No leatherback turtles were reported caught during a 2007 Malagasy village survey (Humber *et al.* 2010). Although protected by Presidential Decree (2006–400), fishers target turtles at sea for consumption (Ratsimbazafy 2003; Epps 2006; Humber *et al.* 2010). Humber *et al.* (2010) report that the Malagasy law is not adequately implemented due to lack of enforcement, a reluctance to manage the local, cultural fishery, and the size of the coastline (Rakotonirina and Cooke 1994; Okemwa *et al.* 2005). We conclude that the poaching of turtles remains a significant threat in Madagascar.

Egg and turtle poaching does not appear to be a significant threat in South Africa. Prior to the ban on egg harvest in 1963, substantial numbers of leatherback eggs in South Africa were harvested, likely contributing to the critically low number of nesting females at that time (Nel *et al.* 2015). Hughes *et al.* (1996) concluded that nesting females were not harvested. As a result of the ban, and with a lucrative tourism industry centered on the nesting turtles, egg and turtle harvest has been nearly eliminated (Hughes *et al.* 1996). Nesting females and hatchlings receive “intensive and effective” protection, as most nesting beaches fall within the iSimangaliso Wetland Park (Nel *et al.* 2015). Such beach protections have been key to recovering the number of nesting females to current levels (Hughes *et al.* 1996; Saba *et al.* 2015; Nel *et al.* 2015). We conclude that the poaching of turtles and eggs is not a significant threat in South Africa.

Exposure to poaching is low in South Africa, where the majority of females nest. Few females nest in Mozambique, reducing the DPS’s overall exposure to egg and nesting female poaching during nesting. However, turtles regularly forage in the Mozambique Channel, where they may be poached along the coasts of Mozambique and Madagascar. Poaching of nesting females or post-nesting females (*i.e.*, on land or at sea) reduces both abundance (through loss of nesting females) and productivity (through loss of reproductive potential). Such impacts are high because they directly remove the most productive individuals from DPS, reducing current and/or future reproductive potential. Egg poaching reduces productivity. We conclude that overutilization, as a result of poaching of turtles and eggs, poses a threat to the DPS.

*Disease or Predation*

While we could not find any information on disease for this DPS, predation is a threat to the SW Indian DPS. In South Africa, nest predators include feral dogs, side-striped jackals, honey badgers, and ghost crabs (Hughes 1996; Nel 2006). In the 1960s, the removal of feral dogs greatly reduced nest predation. Similarly, jackals were once a threat (Hughes 1996). However, nest predation by jackals has not been observed for 17 years (R. Nel, pers. comm. April 15, 2019). Nel (2006) reports current rates of predation as relatively low. Nel *et al.* (2013) reports that there is no evidence for significant beach predation on South African beaches. Describing nest predation as minimal in South Africa, de Wet (2012) found that 15.7 percent of nests were depredated in the 2009/2010 and 2010/2011 nesting seasons; ants and ghost crabs were the main cause of egg mortality. During the two seasons, ghost crabs consumed 3.2 percent of hatchlings as they made their way to the sea (de Wet 2012).

While all eggs and hatchlings have some exposure to predation, the species compensated for a certain level of natural predation by producing a large number of eggs and hatchlings. For this DPS, the primary impact is to productivity (*i.e.*, reduced egg and hatching success). We conclude that, though much reduced, predation still poses a threat to the SW Indian DPS.

#### *Inadequacy of Existing Regulatory Mechanisms*

The SW Indian DPS is protected to some degree by several regulatory mechanisms. For each, we review the objectives of the regulation and to what extent it adequately addresses the targeted threat.

Despite efforts to reduce impacts, fisheries bycatch continues to be the primary threat to this DPS (Petersen *et al.* 2009; Nel *et al.* 2013; Wallace *et al.* 2013; Fossette *et*

*al.* 2014; Angel *et al.* 2014; Nel *et al.* 2015; Harris *et al.* 2018). To minimize the impacts from longline fisheries, the FAO published guidelines for sea turtle protection, entitled Technical Consultation on Sea Turtle-Fishery Interactions (FAO 2004; Huang and Liu 2010). The UN 1995 Code of Conduct for Responsible Fisheries (FAO 2004) provides guidelines for the development and implementation of national fisheries policies, including gear modification (*e.g.*, circle hooks, fish bait, deeper sets, and reduced soak time), new technologies, and management of areas where fishery and sea turtle interactions are more severe. The guidelines stress the need for mitigation measures, data on all fisheries, fishing industry involvement, and education for fishers, observers, managers, and compliance officers (FAO 2004; Honig *et al.* 2007). These guidelines, however, are rarely enacted in full. The ICCAT has adopted a resolution for the reduction of sea turtle mortality (Resolution 03-11), encouraging States to submit data on sea turtle interactions, release sea turtles alive wherever possible, and conduct research on mitigation measures. The responsibility to implement mitigation measures remains within each nation, and many nations have not implemented such measures (Honig *et al.* 2007). South Africa, Namibia, and Angola signed the Memoranda of Understanding concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa. Though South African vessels are required to carry a dehooker and line-cutter (Honig *et al.* 2007) and has instituted an observer program (Petersen *et al.* 2009), few other at-sea conservation measures have been implemented (Honig *et al.* 2007). For Taiwanese fishing vessels operating within the range of this DPS, Taiwan has regulations to limit the number of vessels in the area and to require vessels to carry de-hookers. However, bycatch and mortality remain high (Huang and Liu 2010). Similarly, though the extent of shark nets

off South African beaches has been reduced from 44 km in the early 1990s to 23 km in 2007, bycatch and mortality continue to occur (Brazier *et al.* 2012), and Nel *et al.* (2015) identify bather protection nets, together with boat strikes, as the second greatest threat to the DPS, after longline fisheries. Regarding shark nets, Brazier *et al.* (2012) concludes that bycatch is low and rates are stable, but because the leatherback population is small, a further reduction in bycatch is desirable. Because the offshore longline fishery contributes more than the shark nets to leatherback mortality, Brazier *et al.* (2012) also recommends further introduction of bycatch reduction techniques in the longline fishery. Because longline threats are proportionally large and possibly increasing, Harris *et al.* (2018) concludes that bycatch mitigation measures in this industry remain first and most important management action. Thus, existing regulations have been inadequate to meet their objectives.

Beach habitat is protected throughout a portion of the nesting range of this DPS. In South Africa, approximately 280 km of nesting beaches benefit from intensive and effective protection as part of the iSimangaliso Wetland Park, a World Heritage Site since 1999 (UN Educational, Scientific and Cultural Organization 1999; Nel *et al.* 2015). iSimangaliso includes 280 km of beaches, rocky shores, mangroves, lakes, estuaries, and coastal waters out to three nautical miles (5 km) and 200 m depth. Regulations prevent coastal development and commercial fishing within this area. However, Harris *et al.* (2015) estimated that 66 percent of leatherback turtles nest outside of the protected monitoring area (*i.e.*, only 300 km of the 900 km nesting area is monitored and protected). In addition, leatherback turtles use coastal waters that are not protected under the marine reserve. In Mozambique, much of the nesting habitat is protected, including:

the Ponto du Ouro – Kosi Bay Transfrontier Marine Conservation Area; the Maputo Special Reserve; the Bazaruto Archipelago National Park; and the Quirimbas Archipelago National Park. However, nest protection only occurs over nine percent of the Mozambique coastline (Videira *et al.* 2008; Garnier *et al.* 2012). Thus, regulations to protect the nesting habitat of the DPS have been successful. However, leatherback turtles nesting outside these areas receive no protection.

In addition, South Africa hosts several marine protected areas and has proposed to add 20 new marine protected areas to expand protection to five percent of its EEZ (<https://www.marineprotectedareas.org.za/>). Two of these were proposed in order to protect leatherback marine habitat: the 1200 km<sup>2</sup> iSimangaliso Marine Protected Area (off nesting beaches); and the 6200 km<sup>2</sup> Agulhas Front Marine Protected Area (encompassing core foraging habitat). These initiatives are likely to protect leatherback turtles within the proposed areas. However, the DPS has a large range that extends well beyond protected areas. Harris *et al.* (2018) identifies the Mozambique Channel as an additional key priority area to protect.

In South Africa, a 1963 ban on egg and turtle harvest has been effective in virtually eliminating overutilization (Hughes 1996). The current law, Regulation 58(7) of the MLRA (1998), provides full protection to sea turtles and their products. In Mozambique, national legislation includes: Diploma Legislativo 2627 (7 August 1965), Forest and Wildlife Regulation (Decree 12/2002 of 6 June 2002) and Conservation Law (Law 5/2017 of 11 May). These laws protect turtles and eggs and impose fines for poaching or possession. For example, the Forest and Wildlife regulation prohibits the killing of turtles and the possession of their eggs, with fines up to US\$1,000 (Decree

12/2002 of 6 June 2002; Costa *et al.* 2007). In 2008, there were at least 13 conservation programs focusing on protection and education. Despite these efforts, illegal poaching of eggs and turtles remains prevalent in Mozambique (Fernandes *et al.* 2014) due to limited implementation and enforcement of the environmental legislation (Costa *et al.* 2007; Louro 2006; Williams *et al.* 2016; Fernandes *et al.* 2018). In Madagascar, all sea turtles are protected from exploitation by Presidential Decree (2006–400). However, fishers continue to target and consume turtles captured at sea (Ratsimbazafy 2003; Epps 2006; Humber *et al.* 2010). The effectiveness of the Malagasy law is limited due to lack of enforcement, a reluctance to manage the local, cultural fishery, and the size of the coastline (Rakotonirina and Cooke 1994; Okemwa *et al.* 2005; Humber *et al.* 2010). Thus, while regulations to prevent the harvest of turtles and eggs have been adequate in South Africa, regulatory protections in Mozambique and Madagascar are inadequate.

In summary, numerous regulatory mechanisms protect leatherback turtles, eggs, and nesting habitat throughout the range of this DPS. Though the regulatory mechanisms provide some protection to the species, many do not adequately reduce the threat that they were designed to address, generally as a result of limited implementation or enforcement. As a result, bycatch, incomplete nesting habitat protection, and poaching in Mozambique and Madagascar remain threats to the DPS. In summary, we consider the inadequacy of the regulatory mechanisms to be a threat to the SW Indian DPS.

### *Fisheries Bycatch*

Fisheries bycatch is the primary threat to the SW Indian DPS (Wallace *et al.* 2013; Fossette *et al.* 2014; Angel *et al.* 2014; Nel *et al.* 2015; Harris *et al.* 2018). Bycatch occurs in commercial and artisanal, coastal and pelagic fisheries. Gear types include:

longline, purse seine, pelagic trawl, shrimp trawl, gillnets, and beach seines (Honig *et al.* 2007; Petersen *et al.* 2009; Nel *et al.* 2013; Nel *et al.* 2015).

Of all gear types, longline fisheries likely have the largest impact on the DPS (Petersen *et al.* 2009; Nel *et al.* 2013; Angel *et al.* 2014; Nel *et al.* 2015; Harris *et al.* 2018). Leatherback turtles are exposed to longline fisheries throughout their foraging range, including the Benguela Current in the Atlantic Ocean, the Agulhas Current in the Indian Ocean, and coastal waters off South Africa, Mozambique, and Madagascar (Honig *et al.* 2007; Peterson *et al.* 2009; Huang and Liu 2010; Harris *et al.* 2018). Flag states include: South Africa, Mozambique, Japan, and Taiwan (Honig *et al.* 2007; Peterson *et al.* 2009; Huang and Liu 2010).

Harris *et al.* (2018) found a positive, significant relationship between the longline fisheries' extent of overlap with leatherback migratory corridors and threat intensity ( $F_{1,8} = 184.7, P < 0.001, R^2 = 0.95$ ), which was defined as a product of the turtles utilization distribution and the normalized fishing effort. They concluded that incidental capture in longline fisheries was the most important offshore threat to leatherbacks and supports the hypothesis that longlining is suppressing growth of this DPS (Nel *et al.* 2013; Harris *et al.* 2018). Harris *et al.* (2018) calculated longline bycatch rates, around Southern Africa, to be 1,500 leatherback turtles annually. Though this estimate likely includes turtles from other DPSs (SE Atlantic and NE Indian), the authors concluded that even low absolute bycatch has a disproportionately large effect in slowing population growth rates, due to the small nesting female abundance of the SW Indian DPS (Harris *et al.* 2018).

Additional reason for concern is that the threat intensity of longlining was especially high in the last 5 years of the study (ICCAT and IOTC data from 2004 to 2013), suggesting

that the threat and its impacts on the DPS are increasing (Harris *et al.* 2018). Throughout the SE Atlantic and SW Indian Oceans, Harris *et al.* (2018), Wallace *et al.* (2013), deWet (2012), Thorson *et al.* (2012), and Peterson *et al.* (2009) analyze longline bycatch over a large portion of the DPS's foraging range. Wallace *et al.* (2013) categorize the longline fishing effort as medium to high and conclude that such effort leads to a high risk and high bycatch impact for the SW Indian DPS. Thorson *et al.* (2012) used data from the IOTC (1954 to 2009) and South African fishery (2006 to 2009) in a model of leatherback turtle survival and availability. Their model did not find that leatherback survival declined during the period when longline fishing effort increase. However, the authors state that their null result could be explained by an imprecise index of longline effort or using newer bycatch rates for the South African longline fishery (*i.e.*, Petersen *et al.* 2009). For example, based on fisheries data from 30 South African and Asian pelagic longline vessels operating in the South African EEZ between 2006 and 2010, De Wet (2012) estimates the mean annual bycatch to be 7.8 ( $\pm$  7.8 S.D.) leatherback turtles, based on 39 leatherback turtle captures reported over 5 years. Other studies estimate bycatch to be higher. Based on extrapolations from independent observer bycatch reports from 1998 to 2005 (n = 2,256 sets), Peterson *et al.* (2009) estimates that the South African pelagic longline fishery for tunas and swordfish captures 50 leatherback turtles annually, many of which likely belong to the SW Indian DPS (the remainder belong to the SE Atlantic DPS). Though most (84 percent) were caught alive, Peterson *et al.* (2009) estimates the long-term survival of affected turtles at 50 percent (based on an estimated range of 25 to 75 percent; Aguilar *et al.* 1995). Peterson *et al.* (2009) thus estimates total mortality from the South African pelagic longline fishery to be 25 turtles annually, or around two

percent of the total population (based on a total population size of 1,200 leatherback turtles), which they conclude is enough to hamper recovery of the SW Indian population. Nel *et al.* (2013) agrees with this conclusion, citing a 30 year (1965 to 1995) increasing trend in nesting female abundance that stalled as the longline fishery expanded from 1990 to 1995. Huang and Liu (2010) come to a similar conclusion. They report that the longline fishery operated at a relatively low level until 1995, when South Africa, Japan, and Taiwan started a joint venture fishing program.

In the Indian Ocean, Huang and Liu (2010) evaluated the Taiwanese longline fishery bycatch, and Louro (2006) described illegal longlining in Mozambique waters. Huang and Liu (2010) evaluated observer data from 77 trips (4,409 sets) on Taiwanese large-scale longline fishing vessels. They identified 84 leatherback turtles captured from 2004 to 2008, with 48 mortalities (57 percent; Huang and Liu 2010). Extrapolating to the entire Taiwanese longline fishery in the Indian Ocean, they estimated an average bycatch of 173 leatherback turtles between 2004 and 2007. This number likely included individuals from the SW and NE Indian DPSs. In addition to commercial longlining, artisanal longlining also occurs in the SW Indian Ocean. Illegal longlining off Mozambique targets sharks and leatherback turtles. The level of take and mortality is unknown. A program called Eyes on the Horizon reports such events, when observed (Louro 2006).

In the SE Atlantic Ocean, Honig *et al.* (2007) and Angel *et al.* (2014) evaluate longline bycatch. Honig *et al.* (2007) evaluated turtle bycatch by longline fisheries in the Benguela Large Marine Ecosystem by using data from observer reports, surveys, and specialized trips from the coastal nations of South Africa, Namibia and Angola. They estimated bycatch at 672 leatherback turtles annually (based on an annual bycatch

estimate of 4,200 turtles, of which approximately 16 percent are leatherback turtles) in the southern and central regions and as many as 5,600 leatherback turtles (based on an annual bycatch estimate of 35,000 turtles) for the entire Benguela Large Marine Ecosystem (Honig *et al.* 2007). These estimates likely include many leatherback turtles from the much larger SE Atlantic DPS, but telemetry studies indicate that the turtles of the SW Indian DPS use this foraging area too (Luschi *et al.* 2006; Robinson *et al.* 2016). Evaluating ICCAT data, Angel *et al.* (2014) confirms exposure to high longline fishing effort but reports that bycatch of this population is low relative to other leatherback populations. Although Thorson *et al.* (2012) found that increased fishing effort had no explanatory power regarding changes in leatherback survival, other studies identify longline fisheries as the primary threat to the DPS (Petersen *et al.* 2009; Nel *et al.* 2013; Angel *et al.* 2014; Nel *et al.* 2015; Harris *et al.* 2018). Based on the weight of evidence, we agree with the latter and conclude that longline fisheries pose a major threat to the DPS throughout its foraging range.

Other fisheries also impact the SW Indian DPS, possibly resulting in substantial mortalities. However, these fisheries are not as well studied, and mortality estimates are not available (Honig *et al.* 2007; Nel *et al.* 2013). Leatherback turtles are caught in artisanal and commercial shrimp trawl, pelagic trawl, gillnet, purse seine, and beach seine fisheries (Honig *et al.* 2007; Petersen *et al.* 2009; Nel *et al.* 2013). Citing Walker (2005) and Rakotonirina (1994), Nel (2013) reports that the number of sea turtles (all species) caught in artisanal fisheries of the Mozambique Channel could exceed commercial fishery catches. Honig *et al.* (2007) echoes this concern for the Benguela Current Large Marine Ecosystem, citing high mortality rates for these fisheries in other regions. The

Mozambican shrimp trawl fishery operates in the Sofala Bank of the Mozambique Channel, near leatherback nesting, migrating, and foraging areas (Luschi *et al.* 2006; Robinson *et al.* 2016). The fishery supports 50 to 96 vessels that employ standard otter trawl nets in a single or quad-net configuration with an average tow-time of three hours (Brito 2012). It does not employ TEDs and incidentally captures several (*i.e.*, at least two to six but possibly many more) leatherback turtles annually (Louro 2006; Videira *et al.* 2010; SWOT 2017). In 2001, one shrimp trawler captain reported capturing more than six leatherback turtles since fishing season opened; all were captured alive (Gove *et al.* 2001). Based on 39 interviews with observers, enforcement officers, and vessel operators, the fleet ( $n = 50$ ) captures approximately 56 ( $\pm 40$ ) leatherback turtles; the overall estimated mortality rate for bycaught turtles is 14 percent (Brito 2012). Given the overlap between the fishery and an important foraging area, M. Pereira (CTV, pers. comm., 2019) concludes that the Mozambican shrimp trawl fishery may be one of the main threats to this DPS. The South African shrimp trawl fishery has been reduced to two vessels, with an average annual bycatch of less than one leatherback (Honig *et al.* 2007; Petersen *et al.* 2009; Nel *et al.* 2013). Domestic shrimp trawling in Eritrea is considered a major threat to sea turtles, and bycatch is underreported. However, leatherback turtles are relatively rare in these waters, as demonstrated by the foreign trawl fleet, which has 100 percent observer coverage and bycatch records indicating 39 leatherback turtles between 1996 and 2005 (Pilcher *et al.* 2006).

During a small random sampling exercise in 2013 by onboard observers from the Research Division of Eritrea, one leatherback turtle (of 48 sea turtles total) was captured and released (Mebrahtu 2015). On June 20, 2019, the European Union passed a

regulation (PE-CONS 59/1/19 Rev 1) that requires shrimp trawl fisheries to use a turtle excluder device in European Union waters of the Indian and West Atlantic Oceans.

Gillnets in Macaneta, Mozambique, killed two leatherback turtles during the 2010 nesting season (Videira *et al.* 2010) and captured one in the 2003 nesting season (Louro 2006). In Madagascar, leatherback turtles are a “common” bycatch of the set gillnet shark fishery (Robinson and Sauer 2013); mortality is likely high given the 24-hour soak time and propensity for consuming turtle meat. Purse seine fisheries have a much lower impact than longline fisheries (Angel *et al.* 2014); two leatherback turtles were captured (alive) between 1995 and 2010 in the Indian Ocean (Clermont *et al.* 2012). In the EEZ of all Indian Ocean French Territories (mostly from the Mozambique Channel), 40 leatherback turtles were captured in unspecified fisheries from 1996 to 1999; 92 percent were released alive (Ciccione 2006).

Shark or bather nets, which are gillnets installed off beaches in South Africa to limit human-shark interactions, incidentally capture leatherback turtles. According to Nel *et al.* (2015), bather protection nets and boat strikes together present the second greatest threat to the DPS, after fisheries. Three of nine leatherback turtles equipped with satellite tags between 1996 and 2006 were caught in shark nets (de Wet 2012). Between 1981 and 2008, 150 leatherback turtles were captured (mean = 5.36; SE = 0.60), of which 20 were mature females and 39 were mature males (Brazier *et al.* 2012). Total mortality was 62.7 percent, with an annual range of 1 to 12 mortalities (mean = 3.4; SE = 0.47; Brazier *et al.* 2012). Most turtles were captured in December, the peak month for nesting, which together with the prevalence of mature individuals, suggests that bycatch is dominated by adults from nearby nesting and breeding areas (Brazier *et al.* 2012). Analyzing these data

over an additional 2 years (1981 to 2010), de Wet (2012) found that 157 leatherback turtles (mean = 5.26; SD = 2.7) were captured in the nets, with a 62.4 percent mortality rate (mean = 3.3; SD = 1.8).

To reduce bycatch mortality in longlines, South African regulations require vessels to carry a dehooker and line cutter (Honig *et al.* 2007). To reduce bycatch in the shark nets, effort was reduced from 44 km of nets in the early 1990s to 23 km in 2007 (Brazier *et al.* 2012). Despite these efforts, a previously increasing trend in nesting female abundance has stalled and “declined recently” (Nel *et al.* 2013).

Individuals (immature and adult turtles) of this DPS are exposed to high fishing effort throughout their foraging range. Estimates of bycatch rates, when available, range considerably. For example, Harris *et al.* (2018) estimated the annual longline bycatch rates around Southern Africa to be 1,500 leatherback turtles annually; whereas, de Wet (2012) estimated the mean annual bycatch to be 7.8 ( $\pm$  7.8 S.D.) leatherback turtles. We have annual mortality estimates for few individual fisheries: n = 25 for South African longline (Peterson *et al.* 2009); n = 12 for Taiwanese longline (Huang and Liu 2010); n = 1 to 12 for shark nets (Brazier *et al.* 2012). Adding in other longline fisheries and additional gear types may result in more than 100 mortalities annually. These estimates likely include individuals from other DPSs (*i.e.*, the SE Atlantic and NE Indian). However, because of the small nesting population, even small levels of mortality have the potential to slow population growth (Harris *et al.* 2018). Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when potential nesting females are killed. Several studies conclude that bycatch has prevented continued population growth and/or contributed to the recent slight decline in nesting (Petersen *et*

*al.* 2009; Huang and Liu 2010; Brazier *et al.* 2012; Nel *et al.* 2013; Harris *et al.* 2018).

We conclude that fisheries bycatch is the primary threat to the SW Indian DPS.

### *Vessel Strikes*

Vessel strikes are a threat to the SW Indian DPS. According to Nel *et al.* (2015), vessel strikes and bather protection nets together present the second greatest threat to the DPS, after fisheries. Together these threats kill up to 10 leatherback turtles annually (Nel *et al.* 2015). One of 24 leatherback turtles stranded along the South African coastline between 1972 and 2010 was struck by a boat propeller (Nel 2008). However, additional mortalities or injuries may go unnoticed or unreported. Vessel strikes affect adult females returning to nest, removing individuals and their future reproductive potential. Thus, this threat reduces the abundance and productivity of the DPS. We conclude that vessel strikes pose a threat to the DPS.

### *Pollution*

Pollution includes contaminants, marine debris, and ghost fishing gear. As with all leatherback turtles, entanglement in and ingestion of marine debris and plastics are threats that likely kill several individuals a year. For six stranded hatchlings and 24 stranded adults over the past 40 years, the cause of death was generally unknown. However, fishery-related injuries, ghost-fishing (*i.e.*, entanglement in discarded fishing gear), disease, or pollution may be responsible (de Wet 2012). Plastic pollution may be a main threat in the waters off Mozambique (M. Pereira, pers. comm., 2019). Outer accumulation of the Indian Ocean “garbage patch” (Cozar *et al.* 2014) overlaps with foraging areas in the Mozambique Channel and occurs in waters offshore from nesting areas in South Africa and Mozambique. Though we were unable to find ingestion or

entanglement data for SW Indian leatherback turtles, 51.4 percent of gut and fecal samples from loggerhead turtles (n = 74) captured as bycatch in the Reunion Island longline fishery contained marine debris, of which plastic comprised 96.2 percent (Hoarau *et al.* 2014). Ryan *et al.* (2016) found that 24 of 40 loggerhead turtle post-hatchlings had ingested plastics or other anthropogenic debris. Based on the foraging behavior of leatherback turtles and the proximity of the “garbage patch,” we conclude that the ingestion and entanglement of marine debris are threats to this DPS.

In addition, State of the World’s Sea Turtles (SWOT 2017) identifies hydrocarbon extraction along the eastern African seaboard, including northern Mozambique, as the greatest emerging concern for this DPS. They report that the impact of such activities remain to be seen (SWOT 2017). However associated oil spills are likely to modify habitat off nesting beaches and reduce prey availability for all life stages. Harris *et al.* (2018) found that the hydrocarbon industry poses a moderate threat to the DPS because of its spatial overlap with migratory corridors (second in extent, after longline fisheries). They expressed concern over the expansion of the hydrocarbon extraction along the coasts of southern Mozambique and northeastern South African and the possibility of an oil spill in these areas (Harris *et al.* 2018). Pretorius (2018) identified 28 significant impacts to sea turtles as a result of hydrocarbon exploration and production; these included: potential water pollution, light pollution, noise pollution, and habitat destruction. However, Du Preez *et al.* (2018) reports that metal and metalloid contaminants do not appear to be a problem for this DPS. We conclude that pollution poses a threat to the DPS.

*Climate Change*

Climate change is a threat to the SW Indian DPS. The impacts of climate change include: increases in temperatures (air, sand, and sea surface); sea level rise; increased coastal erosion; more frequent and intense storm events; and changes in ocean currents.

Angel *et al.* (2014) identifies coastal erosion as the main beach-based threat to this population and one that is likely to increase with climate change. Though coastal erosion is a natural process, sea level rise (as a result of climate change) increases the rate of erosion and the amount of beach affected. In Bazaruto Archipelago, Mozambique, coastal erosion and rising sea levels destroyed approximately 12 percent of nests over 10 seasons of monitoring (Videira and Louro 2005; Louro 2006). Because leatherback turtles nest lower on the beach than other sea turtles, their eggs are more at risk of being exposed and destroyed by increases in sea level and coastal erosion (Boyes *et al.* 2010). Thus, erosion and rising sea level as a result of climate change are a threat to the DPS.

Sand temperatures influence leatherbacks' egg viability and sex determination. Temperatures over 32°C result in death and temperatures below 29.2°C produce only males (Rimblot *et al.* 1985; Rimblot-Baly *et al.* 1986). Temperature probes on South African beaches reveal that nests are already close to pivotal temperatures, with an average of 29.04°C (S.D. 0.86°C; range 27.62 to 29.69°C; Boyes *et al.* 2010). A modeling study suggests that even if South African beaches experience a temperature increase of 5°C, hatching success and emergence success may not be significantly reduced (Santidrián Tomillo *et al.* 2015). Instead, nesting females may shift their nesting season to months (*e.g.*, July through October) when temperature and precipitation would be similar to current conditions of the current nesting season (*i.e.*, October through January). However, the authors cautioned that because nesting females do not change

their nesting habits in response to oceanographic conditions, they may not change their nesting habits in response to climate change either (Santidrián Tomillo *et al.* 2015). In addition, a shift in the nesting season could have impacts beyond hatching success, such as reduced post-hatchling survival and suboptimal foraging conditions for post-nesting females. We therefore conclude that increased temperatures may be a threat to the DPS, and will likely result in impacts ranging from nesting season shifts to significant nest losses.

The threat of climate change may modify the nesting conditions for the entire DPS. Impacts likely range from small, temporal changes in nesting season to large losses of productivity. Because we are already seeing small impacts due to coastal erosion and sea level rise, we conclude that climate change is a threat to the SW Indian DPS.

#### *Conservation Efforts*

There are numerous efforts to conserve the leatherback turtle. The following conservation efforts apply to the SW Indian DPS (for a description of each effort, please see the section on conservation efforts for the overall taxonomic species): African Convention on the Conservation of Nature and Natural Resources (Algiers Convention), Convention on the Conservation of Migratory Species of Wild Animals, Convention on Biological Diversity, Convention on International Trade in Endangered Species of Wild Fauna and Flora, Convention on the Conservation of European Wildlife and Natural Habitats, Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region (Abidjan Convention) and Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum), Convention

Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention), FAO Technical Consultation on Sea Turtle-Fishery Interactions, Indian Ocean Tuna Commission, The Indian Ocean Tuna Commission, Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding, MARPOL, IUCN, Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region, Ramsar Convention on Wetlands, UNCLOS, and UN Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing. Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce its risk of extinction.

#### *Extinction Risk Analysis*

After reviewing the best available information, the Team concluded that the SW Indian DPS is at high risk of extinction. The DPS exhibits a total index of nesting female abundance of 149 females. Such a limited nesting population size places this DPS in danger of stochastic or catastrophic events that increase its extinction risk. This DPS exhibits a slightly decreasing nest trend at monitored nesting beaches in South Africa. This declining trend has the potential to further lower abundance and thereby increase the risk of extinction. With only one nesting aggregation, the DPS lacks spatial structure, and its genetic diversity is low. Thus, stochastic events could have catastrophic effects on nesting for the entire DPS, with no potential source subpopulations to buffer losses or provide additional diversity. However, the DPS uses multiple, distant, and diverse foraging areas, providing some resilience against reduced prey availability. Based on these factors, we find the DPS to be at risk of extinction, likely as a result of past threats.

Current (ongoing) threats further contribute the risk of extinction of this DPS. The primary threat to this DPS is bycatch in commercial and artisanal, pelagic and coastal, fisheries. Longline fisheries constitute the greatest threat. Though poorly studied, other fisheries together may have overall mortality rates for affected turtles from this DPS that rival those from longline fisheries. Fisheries bycatch reduces abundance by removing individuals from the population. Because several fisheries operate near nesting beaches, productivity is also reduced when nesting females are prevented from returning to nesting beaches. Exposure and impact of this threat are high. Poaching is also a threat to the DPS. Egg and turtle poaching, while no longer a threat in South Africa, likely continues in Mozambique. In Madagascar, turtles are illegally captured at sea and consumed in local villages. Vessel strikes also pose a threat. Vessel strikes kill several leatherback turtles each year, including females returning to beaches to nest. While exposure is low, impacts are high, affecting both abundance and productivity. Coastal erosion and beach driving in Mozambique modify nesting habitat and are believed to result in minor reductions in productivity currently. However, these threats are likely to increase over time as climate change and tourism increases. Climate change is likely to result in reduced productivity due to greater rates of coastal erosion and nest inundation. Predation of eggs and hatchlings is also a threat. However, although predation has the potential to reduce productivity, the DPS has likely adapted to predation by native species, which account for most of the predation at present. Ingestion of plastics and entanglement in marine debris are threats to all leatherback turtles, most likely resulting in injury and reduced health, though sometimes mortality. Though many regulatory mechanisms are in place,

they do not reduce the impact of these threats to levels that allow the DPS to continue its previous increasing nesting trend.

Thus, the Team unanimously concluded, that the SW Indian DPS is at high risk of extinction. The total index of nesting female abundance of 149 females makes the DPS highly vulnerable to threats. We determine, consistent with the team's findings, that the DPS is currently "in danger of extinction." The slightly declining nest trend and lack of spatial structure and diversity further contribute to its risk of extinction. While this small population had an increasing or stable nesting trend for decades, the lack of continued population growth and recent decline may indicate that threats have outpaced productivity. Past egg and turtle harvest initially reduced the nesting female abundance of this DPS and likely confined its nesting habitat to a relatively small geographic area, with little diversity or spatial structure. Currently, fisheries bycatch is the primary present, ongoing threat. It reduces abundance and productivity (*i.e.*, imminent and substantial demographic risks) by removing mature and immature individuals from the population at rates exceeding replacement. Though numerous conservation efforts apply to this DPS, they do not adequately reduce the risk of extinction. We conclude that the SW Indian DPS is in danger of extinction throughout its range and therefore meets the definition of an endangered species. The threatened species definition does not apply because the DPS is at risk of extinction currently (*i.e.*, at present), rather than on a trajectory to become so within the foreseeable future.

### **NE Indian DPS**

The Team defined the NE Indian DPS as leatherback turtles originating from the NE Indian Ocean, south of 71° N, east of 61.577° E, and west of 120° E. The western

boundary occurs at the border between Iran and Pakistan, where the Somali Current begins. This current, and the cold waters of the Antarctic Circumpolar Current, likely restrict the nesting range of this DPS. We placed the eastern boundary at 120° E to approximate the Wallace and Huxley lines, which are established biogeographic barriers to gene flow between Indian and Pacific Ocean populations of numerous species. While the genetic differences between the NE Indian and West Pacific DPSs demonstrate discreteness, genetic sampling is unavailable from areas where the nesting range of the DPSs likely meet, preventing us from defining the boundary more specifically.

The range of the DPS (*i.e.*, all areas of documented occurrence) extends throughout the Indian Ocean and possibly into the Pacific Ocean. Records indicate that the species occurs in the waters of the following nations: India, Sri Lanka, Bangladesh, Myanmar, Thailand, Malaysia, Indonesia, Vietnam, China, and Philippines (Hamann *et al.* 2006). Given the range of the DPS, leatherbacks may also occur in the waters of Pakistan, Australia, Brunei, Cambodia, Philippines, and Taiwan.

Leatherback turtles of the NE Indian DPS nest on beaches scattered throughout the NE Indian Ocean. The largest abundance of nesting occurs on beaches of the Andaman and Nicobar Islands in India. The sandy beaches of the Andaman and Nicobar Islands consist of soft limestone formed of coral and shell (Lal 1976; Bandopadhyay and Carter 2017). A moderate amount of nesting occurs in Sri Lanka, and even less occurs in Thailand and Sumatra, Indonesia (Hamann *et al.* 2006; Nel 2015).

Information on this DPS is limited, but foraging appears to occur throughout the Indian Ocean (Andrews *et al.* 2006; Hamann *et al.* 2006). The foraging range extends throughout the Bay of Bengal, south of Sri Lanka, and along the west coast of Sumatra,

Indonesia, as indicated by satellite telemetry data and fisheries reports (NMFS and FWS 2013). Nesting females at Little Andaman Island likely use a variety of foraging areas and have been tracked to: south and east of the Andaman and Nicobar Islands; along the coast of Sumatra; beyond Cocos (Keeling) Island towards Western Australia; and across the Indian Ocean towards Madagascar and the African continent (Namboothri *et al.* 2012; Swaminathan *et al.* 2017; Swaminathan *et al.* 2019). Stranding data also indicate the use of diverse foraging areas: 15 individuals stranded or were caught in fishing gear along the mainland coast of India (Shanker 2013). Leatherback turtles have also stranded along the coasts of Mindanao, Philippines and Pakistan (Firdous 2006; Lucero *et al.* 2011).

### *Abundance*

The total index of nesting female abundance of the NE Indian DPS is 109 females. We based this total index on the nesting aggregations at South and West Bays, Little Andaman Island, India (K. Shanker pers. comm., 2018). Our total index does not include 14 unquantified nesting aggregations in Bangladesh, India, Indonesia, Malaysia, Myanmar, Sri Lanka, Thailand, Philippines, and Vietnam. To calculate the index of nesting female abundance, we divided the total number of nests at South and West Bays, Little Andaman Island between the 2015/2016 and 2017/2018 nesting seasons (*i.e.*, a 3-year remigration interval; Andrews 2002) by the clutch frequency (3.8 clutches/season; Andrews 2002; Eckert *et al.* 2015). This number represents an index of abundance for this DPS, and is likely to be an underestimate, because it only includes available data from recently and consistently monitored nesting beaches. Additional nesting occurs at other locations but is unquantified.

Published estimates of total nesting female abundance are not available for this DPS. The IUCN Red List assessment did not provide an estimate of the total number of mature individuals because monitoring was not sufficient (Tiwari *et al.* 2013). Currently, the largest nesting aggregations occur in the Andaman and Nicobar Islands of India. Nesting in Sri Lanka may consist of about 100 to 200 nesting females per year, and low levels of nesting occur in Thailand and Sumatra, Indonesia (Hamann *et al.* 2006; Nel 2012). Low and scattered nesting occurs in Indonesia: 1 to 14 nesting females annually at Alas Purwo in East Java; and one to three nesting females annually on three beaches in Bali. There are also rare reports of nesting in the Philippines (Lucero *et al.* 2011; Arguelles 2013), Vietnam, and Malaysia. In Myanmar, nesting is rare, and only one confirmed nesting event has been recorded in recent years (*i.e.*, December 2016; Platt *et al.* 2017). Historically, there may have been nesting in Bangladesh, but no current reports exist (Hamann *et al.* 2006).

Malaysia once hosted the DPS's largest nesting aggregation (Chan and Liew 1996). It is now considered functionally extinct or extirpated (Pilcher *et al.* 2013), as a result of continuous, large-scale egg harvest and fisheries bycatch (Chan and Liew 1996; Eckert *et al.* 2012). The major nesting site in Malaysia, Rantau Bang in Terengganu, decreased drastically from 10,000 nests in the 1950s to 10 or fewer nests in the 2010s (reviewed by Eckert *et al.* 2012), and to one or no nests annually, more recently. The number of nesting females in Vietnam has also decreased dramatically, from approximately 500 nesting females in the 1960s to two to three nests in 2005 and 2007 (The Chu and Nguyen 2015). In the late 1970s, females nested in multiple locations of Thailand, including: along the airport beach in Changwat Phuket; in the Laem Phan Wa

marine reserve; and in coastal Changwan Phangnga (Bain and Humphry 1980). Settle (1995) recorded about 30 nests on the Phuket and Phangnga coastlines from 1992 to 1993. Aureggi *et al.* (1999) found nine nests between 1997 and 1998, during a survey of Phra Thong Island in the south.

Our total index of nesting female abundance (109 females) places the DPS at risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb losses in individuals. Due to its small size, the DPS has restricted capacity to buffer such losses. Historic abundance estimates indicate that the DPS was once much larger. The current abundance is likely a result of past and current threats, which we describe below. Given the intrinsic problems of small population size, we conclude that the limited nesting female abundance is a major factor in the extinction risk of this DPS.

### *Productivity*

The NE Indian DPS has exhibited a drastic population decline with extirpation of its largest nesting aggregation in Malaysia. Nest counts from Malaysia exhibited a steep decline of 17.9 percent annually (sd = 4.2 percent; 95 percent CI = -25.5 to -8.4 percent;  $f = 0.998$ ; mean annual nests = 1,166) over the 44-year period of data collection (1967 to 2010). The drastic decline of nests observed in Malaysia is likely representative of the overall trend for the DPS given the magnitude of historical abundance for that site and the high confidence in the trend estimate.

Despite the dramatic population decline, driven by the extirpation of the largest nesting aggregation (*i.e.*, Malaysia), productivity parameters are similar to the species averages. However, we have a low degree of confidence in these estimates due to limited monitoring of existing nesting aggregations. We conclude that the NE Indian DPS exhibits a declining nesting trend, which increases its extinction risk.

### *Spatial Distribution*

For the NE Indian DPS, nesting is limited to a few, scattered nesting beaches. Currently, the majority of the nesting occurs on beaches of the Andaman and Nicobar Islands and Sri Lanka, with small numbers of nests on the western coast of Thailand, Sumatra, and Java (Nel *et al.* 2015).

Spatial structure is unknown but presumed to be low. There are no estimates of genetic population structure within this DPS because published genotypes only exist for Malaysia (Dutton *et al.* 1999, 2007). Genetic samples were taken from nesting females at Little Andaman Island from 2008 through 2010, but the results are not yet available (Namboothri *et al.* 2010).

The wide distribution of foraging areas likely buffers the DPS somewhat against local catastrophes or environmental changes that would limit prey availability. Remaining nesting is limited to a few, scattered but broadly distributed nesting sites. The largest nesting aggregations are clustered, thus rendering the DPS susceptible to environmental catastrophes (*e.g.*, tsunamis), and directional changes (*e.g.*, sea level rise). Thus, despite widely distributed foraging areas, the somewhat limited nesting distribution increases the extinction risk of the NE Indian DPS.

### *Diversity*

Genetic diversity of the NE Indian DPS is potentially relatively high, based on analyses of samples collected from the previously large, but now functionally extinct, nesting aggregation in Malaysia (Dutton *et al.* 1999, 2007); genetic diversity has not been assessed at other nesting sites. The diversity of nesting sites is low, given that the majority of the nesting currently occurs on islands (Sivasundar and Prasad 1996). We conclude that existing diversity provides little resilience to the DPS.

*Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

Erosion, coastal development, and artificial lighting have destroyed or modified the available, suitable nesting habitat and thus are threats to the NE Indian DPS.

Erosion reduces the available nesting habitat for the DPS. Some erosion occurs as a result of natural disasters. In 2004, a major earthquake occurred off the west coast of Sumatra, Indonesia, resulting in the 2004 tsunami, which destroyed many of the beaches that once hosted over 1,000 nests (Subramaniam *et al.* 2009). As a result of the tsunami, the width of the coastline was reduced by one meter, severely modifying the beaches of South Bay, Little Andaman Island, and resulting in very low leatherback nesting in 2005 and 2006 (Namboothri 2010). The tsunami also caused drastic changes at other leatherback nesting beaches (Alfred *et al.* 2005; Ramachandran *et al.* 2005; Murugan 2005; Andrews *et al.* 2006). It caused erosion at some beaches and accretion at others, especially in the Andaman and Nicobar Islands, which lie closest to the epi-center of the earthquake and host the largest numbers of nesting females in the DPS (Subramaniam *et al.* 2009). In addition, the beaches in Indonesia are being lost due to erosion from high tides and monsoons (Obermeier 2002).

Sand mining and tourism-related development are the main threats to nesting habitat (Fatima *et al.* 2011). While we were unable to find specific information regarding sand mining, coastal development is increasing in Sri Lanka, India, and Bangladesh. The beaches of Sri Lanka are under high threat from tourism development (*e.g.*, large hotels and restaurants), urban and industrial development, and artificial lighting (Kapurisinghe 2006). Along the mainland of India, granite blocks and embankments prevent turtles from approaching many beaches (Andrews *et al.* 2006). Intense coastal development, stemming from the tourism industry, occurs in Bangladesh without environmental review (Pilcher 2006), resulting in the alteration of sand dunes and other activities that reduce the quality of nesting habitat (Islam 2002; Islam *et al.* 2011). In Vietnam, increasing tourism is expected to result in coastal development on the beaches of Son Tra Peninsula, QuanLan, and Minh Chau (Ministry of Fisheries 2003). In addition, most Vietnam beaches are affected by a large amount of marine debris (*e.g.*, glass, plastics, polystyrenes, floats, nets, and light bulbs), which can entrap turtles and impede nesting activity.

Artificial lighting modifies the quality of nesting beaches because lights over land disorient nesting females and hatchlings. Instead of crawling toward the surf and their marine habitat, they crawl further inland, where they may become dehydrated and die or are susceptible to predation. Nests moved to hatcheries as part of conservation efforts may be subject to inadequate hatchery practices, which have resulted in skewed sex ratios and low hatching success (Chan and Liew 1996; Kapurisinghe 2006; Rajakaruna *et al.* 2013; Phillott *et al.* 2018).

Some areas are protected. Of the 306 islands in the Andaman and Nicobar Islands of India, 94 are designated as wildlife sanctuaries, six of which are national parks, and two of which are marine national parks (Andrews *et al.* 2006). In Sri Lanka, in 2006, sea turtle sanctuaries were established at Rekawa (4.5 km stretch) and Godawaya (3.8 km stretch), where high frequency leatherback nesting is observed; the area is bounded 500 meters towards the sea and 100 meters towards the land from the high tide level in both sites (Phillott *et al.* 2018). Although laws protect sea turtles throughout Sri Lanka, most nesting areas are not protected and hence, local communities can disturb nesting beaches by removing sand, lighting the beaches, and cutting the beach vegetation (Phillott *et al.* 2018). In Malaysia, turtle sanctuaries have been established in Terengganu, Sabah, and Sarawak. However, nesting habitat modification and destruction continue in many areas.

We conclude that nesting females, hatchlings, and eggs are exposed to the reduction and modification of nesting habitat, as a result of erosion, coastal development, and artificial lighting. These threats impact the DPS by reducing nesting and hatching success, thus lowering its productivity. The most abundant remaining nesting aggregations are protected from development, but they experience high rates of erosion; other nesting beaches are subject to anthropogenic threats. Thus, we conclude that habitat loss and modification pose a threat to the NE Indian DPS.

#### *Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

Overutilization is a threat to the NE Indian DPS. The harvest of turtles and eggs led to the historical decline of the DPS, and poaching continues in several areas (Phillott *et al.* 2018).

Regular, nearly complete egg harvest caused the functional extinction of the once large nesting aggregation in Malaysia (Chan and Liew 1996). In the early 1960s, the Terengganu, Malaysia nesting beaches were leased to the highest bidder, and nearly all leatherback eggs were harvested. In the 1980s, the State Fisheries Department tried to buy back about 10 percent of the harvested eggs to be incubated in a hatchery (Siow and Moll 1982; Chan and Liew 1996; Stiles 2009). However, such efforts could not prevent the extirpation. Excessive egg harvest, both legal and illegal, also caused declines in India, Sri Lanka, and Thailand (Ross 1982).

The harvest of turtles and eggs continues but has not been quantified (Nel 2012). In Sri Lanka, almost all eggs are taken from the beach and sold at markets or to hatcheries for ecotourism (Kapurusinghe 2000, 2006; Rajakaruna *et al.* 2013; Phillott *et al.* 2018). The conservation benefit provided by hatcheries in Sri Lanka is debatable (Phillott *et al.* 2018) because they do not follow the hatchery practices established by the IUCN (Hewavisenthi 1994; IUCN 2005; Namboothri *et al.* 2012; Rajakaruna *et al.* 2013; Phillott *et al.* 2018). Egg harvest also continues in Thailand. Commercial egg harvest is illegal in the Andaman and Nicobar Islands, and in the Andaman Islands, a ban on hunting and harvesting of turtles came into force in 1977. However, the original inhabitants of the Andaman and Nicobar Islands are exempt from the Indian Wildlife Protection Act (Shanker and Andrews 2004), and Namboothri *et al.* (2012) observed intense egg harvest at Delgarno, Trilby, and East Turtle Islands. In Myanmar, despite regulations prohibiting the consumption of turtle meat and eggs (Hamann *et al.* 2006), there is illegal trade of turtles caught at sea, including leatherback turtles (Murugan 2007). In Sri Lanka, the historically high direct take of turtles at sea has been reduced

(Kapurushinghe 2006). Records indicate that turtle meat and parts were once regularly exported from Tamil Nadu, India, to Sri Lanka, and then to other nations such as the United States, Singapore, and Belgium (Kuriyan 1950; Chari 1964; Shanmughasundarun 1968 as cited in Agastheesapillai and Thiagarajan 1979).

Exposure to egg and turtle poaching remains high throughout the range of the DPS. Poaching of nesting females or post-nesting females at sea reduces both abundance (through loss of nesting females) and productivity (through loss of reproductive potential). Such impacts are high because they directly remove the most productive individuals from the DPS, reducing current and future reproductive potential. Egg harvest reduces productivity only, but, as previously demonstrated within this DPS, can have devastating population-level impacts. We conclude that overutilization, as a result of egg and turtle harvest, poses a major threat to the NE Indian DPS.

#### *Disease or Predation*

While we could not find any information on disease for this DPS, the best available data indicate that predation is a threat to the NE Indian DPS. Multiple predators prey on eggs and hatchlings at several nesting beaches (Andrews 2000). During a 2016 survey of the Nicobar Islands, approximately 57 percent (n = 1,223) of leatherback nests were lost to depredation by feral dogs, water monitor lizards, or feral pigs (*Sus domesticus*; Swaminathan *et al.* 2017). In the South Bay of Great Nicobar Island, wild boars and dogs prey on eggs, while fiddler crabs, dogs, and raptors prey on hatchlings (Sivakumar 2002). Sivasundar and Prasad (1996) documented that Asian water monitor lizards took 68.6 percent of leatherback nests in the Andaman Islands. In Sri Lanka, egg predators include feral dogs, water and land monitor lizards, jackals, wild boars,

mongooses, and ants. Egg predation by feral pigs is a major threat in Indonesia (Maturbongs *et al.* 1993; Maturbongs 1995, 1996; Sivasundar and Prasad 1996).

A large number of eggs and hatchlings are exposed to predation. Though leatherback turtles produce a large number of eggs and hatchlings, published rates of predation (57 to 69 percent) are high. The predation of eggs and hatchlings mainly impacts productivity. We conclude that predation poses a threat to the NE Indian DPS.

#### *Inadequacy of Existing Regulatory Mechanisms*

Turtles of the NE Indian DPS are protected by several regulatory mechanisms. For each, we review the objectives of the regulation and to what extent it adequately addresses the targeted threat. Nearly all nations that host nesting aggregations have legislation to protect sea turtles.

In India, the leatherback turtle is included on Schedule I, Part II of the Wildlife (Protection) Act, 1972 (Entry No. 11) updated by Wild Life (Protection) Amendment Act, 2002 (No. 16 of 2003). India also bans the hunting and trade of wild animals (India National Report to CMS, 1991 and 1994). However, the indigenous people of the Andaman and Nicobar Islands are exempt from these laws. India has regulations to require TEDs and minimize fisheries interactions; and much of the Andaman and Nicobar Islands are protected as wildlife sanctuaries, including two marine national parks (Andrews *et al.* 2006).

In Indonesia, Order No.301/1991 lists leatherback turtles as a protected species. Pursuant to the Act of 10 August 1990 on the Conservation of Living Resources and Their Ecosystems, it is prohibited to kill, capture, possess, transport, trade in or export protected animals whether alive or dead, or parts of such animals. The taking,

destruction, trade or possession of the eggs or nests of protected animals are also prohibited (ECOLEX 2003). There are no habitat protections and no regulations to minimize fisheries interactions or require TEDs in Indonesia.

In Sabah, Malaysia, the leatherback turtle is not listed as a totally protected or partially protected species in the Wildlife Conservation Enactment (No. 6 of 1997). In Sarawak, Malaysia, leatherback turtles have been fully protected since 1958. Under the Wildlife Protection Ordinance 1998, all marine turtles in Malaysia are protected from hunting, killing, capture, sale, import, export, possession of any animal, recognizable part or derivative or any nest, except in accordance with the permission in writing of the Controller of Wildlife for scientific or educational purposes or for the protection or conservation of a species (Tisen and Bali 2002). The nesting beach at Rantau Abang, Terengganu is protected. However, the nesting aggregation that once used this beach has been extirpated. In 1994, the waters surrounding 38 offshore islands of Peninsular Malaysia and Labuan became protected as marine parks. In addition, one national park in Sarawak, three in Sabah, and one state park in Terengganu protect coastal and marine ecosystems (Malaysia National Biodiversity Policy 1998). Additional habitat protections include: The Turtle Trust Ordinance 1957; Land Code 1958; Turtle Protection Rules 1962; Fisheries Prohibited Areas under section 61 of the Fisheries Act 1985; and the Wildlife Protection Ordinance 1998 (Tisen and Bali 2002). The use of TEDs will be required in Malaysia by 2020.

In Myanmar, the Burma Wildlife Protection Act 1936 (Act No. VII of 1936) requires licenses to hunt, possess, sell, or buy wild animals with closed hunting seasons

(FAOLEX 2003). The Burma Wildlife Protection Rules of 1941 states that the import or export of any reptile (including parts or products) into or from Myanmar is prohibited.

In Pakistan, the leatherback turtle is protected in Baluchistan, Azad Kashmir and Sind (Baluchistan Wildlife Protection Act 1974 No.19/1974; The Azad Jammu and Kashmir Wildlife Act 1975 No.23/1975; The Sindh Wildlife Protection Ordinance 1972 No.5/1972). Possession, transport, and/or national trade are prohibited or regulated (ECOLEX 2003).

In Sri Lanka, the leatherback turtle is protected under the Fauna and Flora Protection Ordinance (Sri Lanka National Report to CMS 1994), which makes it an offense to kill, wound, harm or take a turtle, or to use a noose, net, trap, explosive or any other device for those purposes, to keep in possession a turtle (dead or alive) or any part of a turtle, to sell or expose for sale a turtle or part of a turtle, or to destroy or take turtle eggs. The minister of Fisheries and Aquatic Resources may also prohibit or regulate the import and export of turtles or their derivatives (Parliament of the Democratic Socialist Republic of Sri Lanka 1993). The nesting beach in Yala Reserve is also protected.

In Thailand, the Leatherback Turtle is protected under the Animals Protection Act B.D 2535 (The Zoological Park Organization 2003).

In summary, numerous regulatory mechanisms protect leatherback turtles, their eggs, and nesting habitat throughout the range of this DPS. Although these regulatory mechanisms provide some protection, many do not adequately reduce the threat that they were designed to address, generally as a result of limited implementation or enforcement. As a result, bycatch, nesting habitat protection, and legal and illegal harvest remain

threats to the DPS. We conclude that the inadequacy of the regulatory mechanisms is a threat to the NE Indian DPS.

### *Fisheries Bycatch*

Fisheries bycatch is a threat to the NE Indian DPS. Capture in gillnet, trawl, purse seine, and longline fisheries is a significant cause of leatherback mortality for this DPS (Wright and Mohanty 2002; Hamann *et al.* 2006; Project GloBAL 2007; Bourjea *et al.* 2008; Abdulqader 2010; Wallace *et al.* 2010).

Gillnet fisheries pose a major threat to the DPS. A survey conducted at 16 main fishing ports in Sri Lanka estimated that 431 leatherback turtles were caught in gillnets between 1999 and 2000 (Kapurusinghe and Cooray 2002). In Malaysia, Chan *et al.* (1988) reported an average of 742 and 422 sea turtles, most of which were leatherback turtles, caught in drift gillnets and bottom longlines, respectively. In Bangladesh, gillnets, set bag nets, trawl nets, seine nets, hook and line and other net types of gear capture turtles (Hossain and Hoq 2010). Gillnet and purse seine fisheries are common off the coasts of the Andaman and Nicobar Islands, where the largest nesting aggregations occur (Shanker and Pilcher 2003; Chandi *et al.* 2012).

Trawl fisheries also pose a large threat to the DPS. In India, TEDs are required for trawl nets. However, fishers are reluctant to use them (Murugan 2007). Trawl fishing is also common in Bangladesh, and the use of TEDs is not required (Ahmed *et al.* 2006).

Longline fisheries occur in coastal and pelagic waters. Huang and Liu (2010) evaluated observer data from 77 trips (4,409 sets) on Taiwanese large-scale longline fishing vessels in the Indian Ocean. They identified 84 leatherback turtles captured from 2004 to 2008, with 48 mortalities (57 percent; Huang and Liu 2010). Extrapolating to the

entire Taiwanese longline fishery in the Indian Ocean, they estimated an average bycatch of 173 leatherback turtles between 2004 and 2007. This number likely includes individuals from both the SW and NE Indian DPSs (Louro 2006). In Vietnam, longline fisheries continue to capture leatherback turtles. However, a circle hook program has been implemented to minimize the impact (WWF 2013).

Purse seine fisheries have a much lower impact than longline fisheries (Angel *et al.* 2014); two leatherback turtles were captured (alive) between 1995 and 2010 in the Indian Ocean (Clermont *et al.* 2012). In the EEZ of all Indian Ocean French Territories (mostly from the Mozambique Channel), 40 leatherback turtles were captured in unspecified fisheries from 1996 to 1999; 92 percent were released alive (Ciccione 2006).

In Thailand, one of the main causes of decline in the turtle population is bycatch in trawl, drift gillnet, and purse seine fisheries. The rapid expansion of fishing operations is largely responsible for the increase in adult turtle mortality due to bycatch (Settle 1995).

In Malaysia, the Fisheries Act of 1985 prohibited capture of sea turtles by any type of fishery. However, this merely reduced the reporting of interactions (Yeo *et al.* 2011 in Dutton *et al.* 2011). The 1991 Regulations prohibit fishing in waters adjacent to Rantau Abang during the leatherback nesting season (Chan 1993).

We conclude that juveniles and adults are exposed to high fishing effort throughout their foraging range and in coastal waters near nesting beaches. Mortality rates are likely high, especially in areas where turtle meat is consumed. Mortality reduces abundance, by removing individuals from the population. It also reduces productivity,

when nesting females are incidentally captured and killed. We conclude that fisheries bycatch is a major threat to the NE Indian DPS.

### *Pollution*

Pollution includes contaminants, marine debris, and ghost fishing gear. Ghost fishing gear can drift in the ocean and fish unattended for decades and kill numerous individuals (Wilcox *et al.* 2013). The main sources of ghost fishing gear are gillnet, purse seine, and trawl fisheries (Stelfox *et al.* 2016). In one collection event, volunteers collected over 600 nets, ropes, and buoys from India, Maldives, Oman, Pakistan, Sri Lanka, and Thailand (Stelfox *et al.* 2016). Though educational programs created in 2014 focus on reusing and recycling fishing gear, the threat continues throughout the range of the DPS. Ghost nets in the Maldives primarily drift from fisheries in the Bay of Bengal (*e.g.*, Sri Lanka and India; Stelfox *et al.* 2016). Around the Andaman and Nicobar Islands and Sri Lanka, plastics and other garbage are washed from polluted beaches and inland waters to the sea, where they can kill or harm sea turtles through ingestion or entanglement (Kapurusinghe 2006; Das *et al.* 2016). Pollution has been identified as a main threat to sea turtles in Iran (Mobaraki 2007) and Pakistan (Firdous 2001). However, no specific information about the type of pollution was provided. In Gujarat, India, increased port and shipping traffic have resulted in oil spills and the release of other pollutants, such as fertilizers and cement (Sunderraj *et al.* 2006). Heavy metals and *E. coli* were found at relatively high levels in the waters of Malaysia (including Terengganu) and in the pancreases and livers of leatherback turtles (Caurant *et al.* 1999; Ngah *et al.* 2012). It is not known how these pollutants affect leatherback physiology (Jakimska *et al.* 2011).

As with all leatherback turtles, entanglement in and ingestion of marine debris and plastics are threats that likely kill several individuals a year. However, data specific to this DPS were not available. We conclude that pollution is a threat to the NE Indian DPS, albeit with effects that are unquantifiable on the basis of the best available information.

### *Climate Change*

Climate change is a threat to the NE Indian DPS. A significant rise in sea level would further reduce nesting habitat, which is already affected by erosion. The DPS is also likely to be affected by increases in sand temperatures (Hawkes *et al.* 2009; Poloczanska *et al.* 2009). Sand temperatures prevailing during the middle third of the incubation period determine the sex of hatchling sea turtles (Mrosovsky and Yntema 1980). Incubation temperatures near the upper end of the tolerable range produce only female hatchlings, while incubation temperatures near the lower end of the tolerable range produce only males. As temperatures increase, incubation temperatures may exceed the thermal tolerance for embryonic development, thus increasing embryo and hatchling mortality.

In addition, the frequency and intensity of severe storm events and cyclones in the Bay of Bengal are predicted to increase with climate change (Balaguru *et al.* 2014).

Climate change is likely to modify nesting conditions for the entire DPS. Impacts likely range from small changes in nesting metrics to large losses of productivity. As the DPS is already experiencing nesting habitat loss due to coastal erosion, we conclude that climate change is a threat to the NE Indian DPS.

### *Conservation Efforts*

There are numerous efforts to conserve the leatherback turtle. The following conservation efforts apply to the NE Indian DPS (for a description of each effort, please see the section on conservation efforts for the overall species): Association of Southeast Asian Nations Ministers on Agriculture and Forestry, Andaman and Nicobar Island Environmental Team, The Centre for Herpetology/ Madras Crocodile Bank Trust, Convention on the Conservation of Migratory Species of Wild Animals, Convention on Biological Diversity, Convention on International Trade in Endangered Species of Wild Fauna and Flora, Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention), FAO Technical Consultation on Sea Turtle-Fishery Interactions, The Indian Ocean Tuna Commission, Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding, MARPOL, IUCN, Memorandum of Agreement between the Government of the Republic of the Philippines and the Government of Malaysia on the Establishment of the Turtle Island Heritage Protected Area, Memorandum of Understanding on Association of South East Asian Nations Sea Turtle Conservation and Protection, The Memorandum of Understanding of a Tri-National Partnership between the Government of the Republic of Indonesia, the Independent State of Papua New Guinea and the Government of Solomon Islands, National Sea Turtle Conservation Project in India, Ramsar Convention on Wetlands, UNCLOS, and UN Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing. Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce its risk of extinction.

*Extinction Risk Analysis*

After reviewing the best available information, the Team concluded that the NE Indian DPS is at high risk of extinction. The once large nesting aggregation in Malaysia is now functionally extirpated. The total index of nesting female abundance is 109 females at all monitored beaches. This estimate is likely low because several nesting sites were not included in the calculation due to lack of consistent, standardized monitoring over multiple and entire nesting seasons. Still, the low nesting female abundance places this DPS at risk of stochastic or catastrophic events that increase its extinction risk. The DPS once exhibited much greater nesting female abundance, which has dramatically declined in recent decades. It currently exhibits a slightly declining nest trend at monitored nesting beaches in India. The DPS exhibits average productivity metrics, such as body size, clutch size and frequency. Though it exhibits some spatial distribution and diversity, with multiple foraging sites and relatively high genetic diversity at the sampled locations, nesting only occurs on islands. Based on these factors, we find the DPS to be at risk of extinction as a result of past threats.

Current threats further contribute to the risk of extinction of this DPS. Major threats to the DPS include fisheries bycatch and the harvest of turtles and eggs. There are not many nests to exploit, but evidence suggests that if such nests are found by humans, the eggs are at risk of being harvested. Egg harvest led to the extirpation of the largest nesting aggregation (*i.e.*, Malaysia), and current overexploitation occurs in Thailand, Vietnam, and Sri Lanka. The poaching of turtles is also a threat in Myanmar. Fisheries bycatch is a major threat, with turtles being captured in trawl and gillnet fisheries in Malaysia, India, Thailand, Sri Lanka, Bangladesh, and Indonesia. Erosion on the Andaman and Nicobar Islands, as a result of tsunami damage, has significantly reduced

available nesting habitat. Additional habitat modifications include coastal development and artificial lighting, as a result of increases in tourism. Pollution and climate change are threats that likely affect the DPS by reducing abundance and productivity, though the best available data do not allow for quantification of those effects. Though many regulatory mechanisms are in place, they do not reduce the impact of threats to levels that ensure the continued existence of the DPS.

We conclude, consistent with the team's findings, that the NE Indian DPS is currently in danger of extinction. Its low nesting female abundance makes the DPS highly vulnerable to threats. Dramatic declines in over the past several decades contribute to our concern over the continued persistence of the DPS. Past egg and turtle harvest initially reduced the nesting female abundance of this DPS and likely confined its nesting habitat to a few island beaches, with little diversity and reduced spatial distribution. The present, ongoing threats include: overutilization (*i.e.*, turtle and egg harvest); fisheries bycatch; loss of habitat; and predation. Overutilization and fisheries bycatch reduces abundance and productivity (*i.e.*, imminent and substantial demographic risks) by removing mature and immature individuals from the population at rates exceeding replacement. The loss of nesting habitat and predation (of eggs) reduces productivity and the DPS's ability to recover to its previous abundance. Though numerous conservation efforts apply to this DPS, they do not adequately reduce the risk of extinction. We conclude that the NE Indian DPS is in danger of extinction throughout its range and therefore meets the definition of an endangered species. The threatened species definition does not apply because the DPS is at risk of extinction currently (*i.e.*, at present), rather than on a trajectory to become so within the foreseeable future.

## West Pacific DPS

The Team defined the West Pacific DPS as leatherback turtles originating from the West Pacific Ocean, south of 71° N, north of 47° S, east of 120° E, and west of 117.124° W. The northern and southern boundaries reflect the highest latitude occurrences of leatherback turtles in each hemisphere (Goff and Lien 1988; Carriol and Vader 2002; McMahon and Hayes 2006; Shillinger *et al.* 2008; Benson *et al.* 2011; Eckert *et al.* 2012). We placed the western boundary at 120° E to approximate the Wallace and Huxley lines, which are established biogeographic barriers to gene flow between Indian and Pacific Ocean populations of numerous species. While the genetic differences between the Northeast Indian and West Pacific DPSs demonstrate discreteness, genetic sampling is unavailable from areas where the nesting ranges of the DPSs likely meet, preventing us from defining the boundary more specifically. We placed the eastern boundary at the border between the United States and Mexico to reflect the DPS's wide foraging range throughout the Pacific Ocean. We chose this border because the West Pacific DPS crosses the ocean to forage in the eastern Pacific Ocean, including in waters of the United States, whereas the East Pacific DPS forages primarily off the coasts of Central and South America. The two DPSs overlap in foraging habitats off waters of Chile and Peru (Donoso and Dutton 2010).

The range of the DPS (*i.e.*, all areas of occurrence) extends throughout the Pacific Ocean with specific coastal and pelagic areas in the Indo-Pacific basin providing important foraging and migratory habitats. Documented nesting occurs on beaches of the following nations: Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu. Leatherback turtles of the West Pacific DPS migrate through the EEZs of at least 32

nations including in the U.S. EEZs of California and Hawaii, spending between 45 and 78 percent of the year on the high seas (Harrison *et al.* 2018). Of the 32 nations, the West Pacific DPS migrates through at least 18 nations or territories of the western and southwestern Pacific Ocean: Indonesia, Papua New Guinea, Solomon Islands, Philippines, Malaysia, Vietnam, Japan, Palau, Micronesia, Marshall Islands, Northern Mariana Islands and Guam, Fiji, Vanuatu, Australia, New Caledonia, New Zealand, Line Islands, and Kiribati (Harrison *et al.* 2018). Foraging occurs in seven ecoregions: South China/Sulu and Sulawesi Seas, Indonesian Seas, East Australian Current Extension, Tasman Front, Kuroshio Extension of the Central North Pacific, equatorial Eastern Pacific, and California Current Extension (Benson *et al.* 2011). Individuals demonstrate fidelity to these foraging areas, likely as a result of their post-hatchling dispersal patterns and nesting season (Benson *et al.* 2011; Gaspar *et al.* 2012; Gaspar and Lalire 2017; Harrison *et al.* 2018).

Leatherback turtles of the West Pacific DPS nest in tropical and subtropical latitudes primarily in Indonesia, Papua New Guinea, and Solomon Islands, and a lesser extent in Vanuatu (Dutton *et al.* 2007; Benson *et al.* 2007a; Benson *et al.* 2007b; Benson *et al.* 2011). The majority of nesting occurs along the north coast of the Bird's Head Peninsula, Papua Barat, Indonesia at Jamursba-Medi and Wermon Beaches (Dutton *et al.* 2007). A recent discovery of a previously undocumented nesting area on Buru Island, Maluku Province, Indonesia (WWF 2018) suggests that additional undocumented nesting habitats may exist on other remote or infrequently surveyed islands of the western Pacific Ocean. This DPS nests year round, and exhibits a bimodal nesting strategy whereby a proportion of females nest during November through February (*i.e.*, "winter" nesting

females) and other females nest May through September (*i.e.*, “summer” nesting females; Benson *et al.* 2007a; Benson *et al.* 2007b; Dutton *et al.* 2007; Tapilatu and Tiwari 2007; Benson *et al.* 2011).

Nesting beach habitats throughout the West Pacific are generally dynamic, high profile beaches associated with deep water approaches and strong waves. Beaches can be quite narrow as in parts of the Solomon Islands or Papua New Guinea, or broad as in the case of Jamursba-Medi, Indonesia during the summer months. Nesting females appear to prefer coarse-grained sand free of rocks, coral, or other abrasive substrates (reviewed by Eckert *et al.* 2012).

While West Pacific leatherback turtles do not have distinct “migratory corridors,” several areas are considered “areas of passage” used by turtles traveling between nesting and foraging locations, and there is clear separation of migratory and foraging destinations based on nesting season (Benson *et al.* 2007a, b; Benson *et al.* 2011; Harrison *et al.* 2018). Post-nesting, winter nesting females from Papua New Guinea, Indonesia, and Solomon Islands migrate through the Halmahera, Bismarck, Solomon, and Coral Seas, towards Southern Hemisphere temperate and tropical foraging areas in the Tasman Sea, East Australian Current, and western South Pacific Ocean (Benson *et al.* 2011; Harrison *et al.* 2018; Jino *et al.* 2018). Genetic analyses of leatherback turtles caught in fisheries off Peru and Chile indicates that approximately 15 percent of sampled individuals originate from the West Pacific DPS, likely winter nesting females that have migrated across the Southern Hemisphere to the productive waters off South America (Donoso and Dutton 2010; NMFS unpublished data 2018). It is unclear what proportion

of the West Pacific DPS might utilize this area and how important it might be to this DPS.

Leatherback turtles migrate through and forage in the waters of the Philippines (Benson *et al.* 2007a, 2011; MRF 2010, 2014). In 2005, Salinas *et al.* (2009) found a female in San Fernando (close to El Nido) that had been previously tagged at Jamursba-Medi in July 2003. The Marine Research Foundation (MRF) utilized aerial transects to assess leatherback foraging area use in Palawan waters and off the coast of Borneo (MRF 2010, 2014). They found leatherback turtles (n = 28 in 2010 and 2013/2014) foraging in nearshore waters around the NE and SE coasts of Palawan, potentially linked to large jellyfish aggregations from February to May, and overlapping with high density fishing activity in Taytay Bay, off NE Palawan (MRF 2010, 2014). Additionally, numerous leatherback turtle marine sightings, strandings, and fishery bycatch (typically entangled in gillnet gear) exist for locations throughout the Philippines including Marine Wildlife Watch of the local NGO, Marine Wildlife Watch of the Philippines, from 2010 to 2018 (Bagarinao 2011; Cruz 2006; MRF 2010; MWWP unpublished data 2018).

### *Abundance*

The total index of nesting female abundance of the West Pacific DPS is 1,277 females. We based this total index on two nesting aggregations in Jamursba-Medi and Wermon, Indonesia (Tapilatu *et al.* 2013; Tiwari *et al.* in prep). Our total index does not include 18 unquantified nesting aggregations in Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu. To calculate the index of nesting female abundance (723 females) for Jamursba-Medi (*i.e.*, a 18 km stretch of beach that has been monitored since 1981), we divided the total number of nests between the 2015/2016 and 2017/2018 nesting

seasons (*i.e.*, a 3-year remigration interval) by the clutch frequency (5.5 clutches per season; Tapilatu *et al.* 2013). We performed a similar analysis for data from Wermon (index = 554 females), a 6 km beach that has been monitored since 2002.

Based on the Tapilatu *et al.* (2013) study, the IUCN Red List assessment estimated the total number of mature individuals (including females and males) utilizing Jamursba-Medi and Wermon beaches to be 1,438 leatherback turtles (Tiwari *et al.* 2013). The IUCN estimate includes males and thus is higher than ours. Curtis *et al.* (2015) provided a minimum annual nesting female estimate of 318 females (or 954 total nesting female abundance over a 3-year remigration interval). Dutton *et al.* (2007) estimated that 1,113 females may have nested annually, or conservatively 2,700 total nesting females, in the entire western Pacific population. At that time, they estimated 75 percent of the population originated from Bird's Head Peninsula (or approximately 2,025 females; Dutton *et al.* 2007). Our total index is within the range of published estimates of abundance for this DPS, taking into account differences in survey methods over time, and is based on the best available data for the DPS at this time.

Within the nesting range of this DPS, nest monitoring activities have occurred relatively recently, with standardized methods in Papua Barat first implemented in 2002 (Hitipeuw *et al.* 2007; Tapilatu *et al.* 2013). Outside the Bird's Head Peninsula, monitoring has been sporadic, opportunistic, and spatially limited because the region is vast, remote, and logistically challenging to access. Often nesting beaches are located far from towns or cities, where there are no roads to, or electricity in, adjacent villages. Cultural and socio-economic dynamics confound monitoring programs, which are dependent upon fiscal sponsorship, incentives, community buy-in, and the degree of

familiarity of local communities with concepts of sustainability or conservation (Kinch 2006; Gjersten and Pakiding 2012). While Jamursba-Medi and Wermon beaches have been monitored fairly consistently over time, less is known about the status and trends of nesting beaches in Papua New Guinea, Solomon Islands, and Vanuatu. Records are further confounded by changes in place names and jurisdictional boundaries over recent decades (*e.g.* the Indonesian province formerly known as Irian Jaya is currently two provinces of Papua and Papua Barat). Village names or location descriptions have also changed over time, and geographic coordinates were not recorded historically. Therefore, all estimates of abundance in this DPS carry substantial uncertainty.

In Indonesia, aerial surveys provided the first indication of leatherback nesting in Papua (*i.e.*, Irian Jaya; Salm 1982). At that time, Salm (1982) did not provide location details out of concern that public disclosure prior to protection would be detrimental. Follow-up studies during the 1980s and 1990s indicated that a large nesting population was located along the coastal beaches of northern Papua or Papua Barat, Bird's Head Peninsula (Bhaskar 1985). Systematic monitoring of leatherback turtles began during the early 1990s, primarily in the form of annual nest counts (Hitipeuw *et al.* 2007). On the Bird's Head Peninsula of Papua Barat, nesting occurs mainly at Jamursba-Medi and Wermon, where a total of 1,371 nesting females were tagged between 2002 and 2011 (Tapilatu *et al.* 2013). The primary nesting season at Jamursba-Medi occurs during the summer (May to September), while nesting occurs year round at Wermon with a small peak in July and primary nesting activity during the winter, between November and February (Hitipeuw *et al.* 2007). Historically, approximately 60 percent of nesting activity occurs at Jamursba-Medi with 40 percent of activity at Wermon (Tapilatu *et al.*

2013). While a few females have been documented nesting at both beaches during a nesting season (Tapilatu *et al.* 2013), the vast majority of females do not appear to utilize both Jamursba-Medi and Wermon Beaches during a single nesting season (Tapilatu and Tiwari 2007; Tapilatu *et al.* 2013; Lontoh 2014). Based on nest counts and clutch frequency per season (mean = 5.5 +/- 1.6 nests per female), approximately 464 to 612 females nested at Jamursba-Medi and Wermon in 2011 (Tapilatu *et al.* 2013). Additional low-level nesting activity in Indonesia occurs in the Manokawari region of the Bird's Head Peninsula to the east of the Jamursba-Medi and Wermon Beaches (Suganuma *et al.* 2012). Between 2008 and 2011, 84 to 135 nests were recorded, or a mean of about 117 nests annually (Suganuma *et al.* 2012). However, survey effort was limited and not consistent across years and may underestimate total nesting activity. Further it is unknown whether interchange occurs between turtles nesting in the Manokawari region and those of the Bird's Head Peninsula index beaches. In 2016, nesting activity was identified in Central Maluku at Buru Island, west of Bird's Head Peninsula. In 2017, a monitoring program to quantify nesting activity was initiated on three north coast beaches of Buru Island (totaling 10 km) which documented 203 nests, and preliminary data indicates that there might be two nesting peaks: May through July and November through February (WWF 2018). Nesting activity in other areas of Indonesia are known or suspected, but unquantified (Dutton *et al.* 2007; Tapilatu 2017).

In Papua New Guinea, the majority of known nesting activity occurs during the winter months (November to February) along the Huon Coast on the northeastern coast of the Morobe Province, where 576 females were tagged between 1999 and 2013 (Pilcher 2006, 2008, 2009, 2010, 2011, 2012, 2013; Pilcher and Chaloupka 2013). Aerial surveys

along the Huon Coast in January and December between 2004 and 2006 documented 276 nests, with an estimate of 500 nests per season (Benson *et al.* 2007b; Dutton *et al.* 2007). During the Huon Coast Leatherback Turtle Project, which took place between 2005 and 2012, an average of 258 nests were laid per season (range: 193 to 527) at seven beaches which comprised approximately 35 km of nesting habitat along the Huon Coast (Pilcher 2013; WPRFMC 2015). One challenge in estimating nesting activity in Papua New Guinea is that leatherback site fidelity appears to be variable, with some satellite tagged animals seen visiting a number of areas during one nesting season (Benson *et al.* 2007b). For example, a number of Huon Coast nesting females visited other nearby beaches and east-facing beaches of the Huon Peninsula, including Bougainville and Woodlark Islands during a single nesting season (Benson *et al.* 2007b). Therefore, for assessment purposes, we consider the Huon Coast to be one nesting beach complex.

Additional nesting activity occurs in other areas of Papua New Guinea, such as along the north coast of the Madang Province and on several islands including Manus, Long, New Britain, Bougainville, New Ireland, and Normanby (Prichard 1982; Spring 1982; Benson *et al.* 2007b; Dutton *et al.* 2007). In these areas nesting activity has not been quantified via standardized or consistent methods, but information has been obtained via community surveys, aerial surveys, or rapid assessments. Nesting occurs primarily in the winter months, although low-level year-round nesting may also occur (Spring 1982; Dutton *et al.* 2007). Approximately 50 nests may be laid annually along the north coast of the Madang Province (Benson *et al.* 2007b; TIRN 2017). The Islands of New Britain and Bougainville may host approximately 140 to 160 nests per year, respectively (Benson *et al.* 2007b; Dutton *et al.* 2007; Kinch *et al.* 2009). On

Bougainville Island, aerial surveys conducted during the 2005 and 2007 nesting seasons documented a mean of 68 nests (range: 41 to 107 nests) or an extrapolated estimate of 160 to 415 nests per year (Dutton *et al.* 2007; Benson *et al.* 2007b). In 2009, a one week full-island ground survey (conducted by boat and foot) recorded 46 leatherback nests (Kinch *et al.* 2009).

In the Solomon Islands nesting activity is distributed throughout the country with the majority of nesting activity at Sasakolo and Litogarhira beaches on Isabel Island, and on Rendova and Tetepare Islands in the Western Province (Pita 2005; Dutton *et al.* 2007; Benson *et al.* 2018a). The nesting season occurs primarily during winter (November through February), although some year-round nesting has been documented (Pilcher 2010b; Williams *et al.* 2014; Jino *et al.* 2018; TNC-Solomon Islands 2018 unpublished). Leatherback turtle monitoring was begun by the Solomon Island Department of Fisheries in 1989 (Pita 2005). Between 1999 and 2006, an estimated 640 to 700 nests were laid annually in the Solomon Islands, representing approximately eight percent of the total western Pacific leatherback nesting at that time (Dutton *et al.* 2007). At Sasokolo Beach, Isabel Island, during a 54 day monitoring period between November 28, 2000 and January 21, 2001, 132 nests were documented with an additional 35 nests present when monitoring began (Ramohia *et al.* 2001). Between December 27, 2006 and January 2, 2007, aerial surveys provided seasonal estimates of 207 nests laid on Isabel Island, and an additional 312 nests on other islands (Benson *et al.* 2018a). A January 2011 site visit resulted in 315 nests identified at Sasakolo and Litogahira (Tiwari 2011 unpublished). Recently, nesting activity has also been documented at the southeastern side of Isabel, where approximately 52 females may nest annually (TNC-Solomons 2018 unpublished).

Since 2002, the Tetepare Descendants' Association (TDA) has monitored nesting activity opportunistically in the Solomon Islands, where approximately 30 to 50 leatherback nests are laid seasonally on two beaches (Goby *et al.* 2010). Between July 1, 2012 and April 30, 2013, TDA undertook 257 beach surveys and found 44 leatherback nests (TDA 2013). While monitoring efforts may be ongoing, data management and analysis remains a key challenge for these isolated communities (TDA 2013; Pilcher 2010b). At Rendova Island during the 2003/2004 winter nesting season, 235 leatherback turtle nests were recorded, and during the 2009/2010 season, 79 nests were laid (Pilcher 2010b; Goby *et al.* 2010). Likely the most comprehensive surveys occurred from September 1, 2012 to April 30, 2013 (91 patrols, 3 days per week), which documented a total of 74 nests (TDA 2013). During the 2017/2018 winter nesting season, 29 nests were documented (Solomon Islands Community Conservation Partnership 2018 unpublished data). The community on Vangunu Island documented a total of 23 nests and 11 females between June 2011 and July 2014 (Jino *et al.* 2018). Nesting occurred during two distinct seasons from May to July and from November to January, and of the females tagged, one nested successfully six times and another nested five times (Jino *et al.* 2018). The other nine turtles were only observed nesting once or twice, and it is likely that either some nesting events were not recorded or the females nested on surrounding unmonitored beaches (Jino *et al.* 2018). On Malaita Island at Waisurione beach, nesting activity occurs during the summer (June to August), but only a few females were determined to use the area, with five and seven nests documented in 2014 and 2015, respectively (Williams *et al.* 2014).

Nesting occurs in low numbers at other islands in the western Pacific Ocean. In Vanuatu, 30 to 40 nests are laid annually on Epi and Ambrym Islands (Dutton *et al.* 2007;

Petro *et al.* 2007; WSB 2011), although fewer nests ( $n = 15$ ) were documented during the 2014/2015 nesting season (WSB 2016). Leatherback turtles have been reported in Fiji (Rupeni *et al.* 2002; NMFS and USFWS 2013; Jino *et al.* 2018), but these accounts involved foraging or in-water capture of animals, and it is unclear if historic reports included nesting activity (Guinea 1993; Benson *et al.* 2013). Historical nesting records also exist for the eastern coast of Queensland, in New South Wales, and in the Northern Territories from December to February (Dobbs 2002; Limpus 2009). However, current information was not available at the time of the study, and no nests have been observed since 1995 despite regular monitoring (Flint *et al.* 2012). Since the 1980s, there have also been reports of leatherback turtles nesting in the Philippines (Cruz 2006; MRF 2010). Of recent reports, two documented cases have been confirmed by sea turtle experts (*i.e.*, staff of the Marine Wildlife Watch of the Philippines). On July 15, 2013, at Barangay Yawah, Legazpi City, Albay, NAVFORSOL (the Philippines Naval facility) personnel observed a leatherback nesting, but the eggs failed to hatch. On August 6, 2013 at Camp Picardo beach, Barangay, Eastern Samar, a nesting event was aborted due to disturbance on the beach, but according to the social media report (*i.e.*, a Facebook post), the female was tagged and led back to sea (MWWP unpublished 2018). Given the low-site fidelity of the turtles in this DPS (Benson *et al.* 2007b), it is not surprising that leatherbacks might distribute nests among various areas throughout the region.

The total index of nesting female abundance of the West Pacific DPS (*i.e.*, 1,277 females) places it at risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017). These processes, working alone or in concert, place small populations at a

greater extinction risk than large populations, which are better able to absorb impacts to habitat or losses in individuals. Due to its small size, the DPS has restricted capacity to buffer such losses. Given the intrinsic problems of small population size, we conclude that the nesting female abundance is a major factor in the extinction risk of this DPS.

### *Productivity*

The West Pacific DPS exhibits a declining nesting trend. We conducted trend analyses for the two index beaches in Indonesia, which were the only two beaches with 9 or more recent years of standardized data, with the most recent data collection in 2014 or more recently (the standards for conducting a trend analysis in this report). The median trend in annual nest counts estimated for Jamursba-Medi (data collected from 2001 to 2017) was  $-5.7$  percent annually (sd = 5.4 percent; 95 percent CI =  $-16.2$  to 5.3 percent;  $f = 0.867$ ; mean annual nests = 2,063). While data are available for the period starting in 1999, the best available information indicates that beach monitoring and nest protection practices improved in 2001; therefore, we used the time series starting in 2001. For Wermon (data collected from 2006 to 2017, excluding 2002-2005 and 2013-2015 due to low or insufficient effort), the median trend was  $-2.3$  percent annually (sd = 8.4 percent; 95 percent CI =  $-19.8$  to 14.9 percent;  $f = 0.643$ ; mean annual nests = 1,010). As Jamursba-Medi and Wermon currently represent approximately 75 percent of nesting for this DPS, we consider these declining trends to be representative of the entire DPS.

Our trend data for Indonesia yield similar results to other published findings. The IUCN Red List assessment found a decreasing trend of  $-7$  percent annually (Tiwari *et al.* 2013). Tapilatu *et al.* (2013) identified a  $-5.5$  percent annual rate of decline at Jamursba-Medi between 1984 and 2011 and a  $-11.6$  percent annual rate of decline at Wermon

between 2002 and 2011. Between 1986 and 2010, Benson *et al.* (2013) highlighted drastic declines in the annual number of nests at Jamursba-Medi and Wermon. Additionally, a 27-year aerial survey study indicates a decline in the number of leatherback turtles foraging off central California (Benson *et al.* 2018b). From 1995 to 2003, an estimated 12 to 379 individuals (mean = 178) foraged in this area (Benson *et al.* 2007), while from 2004 to 2017, an estimated 23 to 112 individuals foraged in this area, representing a decline of 5.6 percent annually (Benson *et al.* 2018b).

At Jamursba-Medi, nesting data have been collected for some years since 1981. However, no data were collected during many years in the mid-1980s and late 1990s (Tapilatu *et al.* 2013). There is considerable uncertainty in the early estimates, with over 4,000 nests estimated in 1981, 14,522 nests in 1984, and a dramatic drop to 3,261 nests in 1985 (Tapilatu *et al.* 2013). It is unclear if there was sampling inconsistency between years or if there was an actual decline in nesting activity. However, if analyses are based on the 1984 data, during which the greatest number of nests was recorded, there was a 78.3 percent decline over the past 27 years (1984 to 2011), or 5.5 percent annual rate of decline (Tapilatu *et al.* 2013). Alternatively, if analysis is based on 2005 to 2011 when the Tapilatu *et al.* (2013) study ensued, nesting activity declined 29 percent from 2,626 nests (in 2005) to 1,596 nests (in 2011; Tapilatu *et al.* 2013). Since the Tapilatu *et al.* (2013) study, University of Papua scientists have continued to engage with local communities to monitor nesting activity. The overall nesting trend has continued to decline by 5.6 percent per year between 2003 and 2017. However, there appears to be an increase in nesting since 2013 (Tiwari *et al.* in prep).

The first comprehensive surveys at Wermon beach in 2002 found almost as many nests laid on Wermon as on Jamursba-Medi (Hitipeuw *et al.* 2007). At that time, it was hypothesized that the decline at Jamursba-Medi may have been offset by an increase at Wermon (Hitipeuw *et al.* 2007). However, Tapilatu *et al.* (2013) found a significant decline in nesting at Wermon from 2,994 nests in 2002 to 1,096 nests in 2011 (62.8 percent total or 11.6 percent annual rate of decline). Unfortunately, no monitoring activities occurred at Wermon between 2013 and 2015 due to community discord, which prevented beach access. Between 2006 and 2017, nesting has continued to decline at approximately 2.3 percent (Tiwari *et al.* in prep). However, there may have been a slight increase in recent nesting, similar to Jamursba-Medi (Tiwari *et al.* in prep).

Local residents stated that leatherback turtles were the dominant sea turtle species nesting in Maokawari prior to the 1980s, but that the population has declined significantly since the 1990s due to village development and exploitation of turtles and eggs (Tapilatu *et al.* 2017).

Data collection in Papua New Guinea spanned 8 years and ended prior to 2014. Because these data did not meet our criteria for “recent,” we did not perform a trend analysis, but included a bar graph in the Status Review Report. In Papua New Guinea, nesting activity along the Huon Coast was relatively stable between 2005 and 2013, with 193 to 527 nests per year (mean annual nests = 258) and with most nesting activity occurring at two primary areas, Busama and Kamiali (Pilcher 2013; Benson *et al.* 2015; WPRFMC 2015). Given the exchange of females and evidence of multiple beach use among females in Papua New Guinea (Benson *et al.* 2007b), we consider the Huon Coast to be one nesting area and not individual nesting beaches. Though there have been

several independent studies of abundance over time, we determined that these data are inadequate to incorporate into a trend analysis because these data do not meet our criteria (*i.e.*, nest count data consistently collected in a standardized approach for at least 9 years). For historical perspective, leatherback turtle nesting along the Huon Coast was first identified south of the city of Lae near the Buang River, at an area likely between Labu Tale and Busama villages (*i.e.*, Maus Buang or Buang-Buassi; Bedding and Lockhart 1989; Quinn and Kojis 1985; Hirth *et al.* 1993). Estimates of leatherback turtle nesting at Maus Buang during the 1980s ranged from five to 10 turtles per night from November to January (Quinn and Kojis 1985) or 300 nests laid annually (Bedding and Lockhart 1989). Quinn and Kojis (1985) estimated that 300 to 500 females may nest annually in Papua New Guinea, although it is unclear if estimates were for the Maus Buang area specifically or the Huon Coast at large. Hirth *et al.* (1993) undertook the most standardized survey at that time and recorded 76 nests and 34 females nesting at “Piguwa” (*i.e.*, Maus Buang) on 725 meters of beach during a 15-day period in December 1989. During the Huon Coast leatherback turtle nesting beach program, an average of 35 and 114 nests were laid annually during the 4-month nesting season in this similar area at Labu Tale and Busama beaches, respectively (Pilcher 2013; WPRFMC 2015). Kamiali Beach lies approximately 30 km south of the city of Lae. In 1996, the Kamiali Wildlife Management Area was declared a protected area for leatherback turtles, and the harvest of nests was prohibited along 2 km of beach. In 1999, village rangers began opportunistic tagging of nesting females at Kamiali. A community-based nesting beach monitoring program was established in 2003, which soon grew into the Huon Coast Leatherback Turtle Conservation Program (Benson *et al.* 2007b; Pilcher and Chaloupka 2013; Kinch

2006). By 2005, monitoring activities expanded from Kamiali Beach (approximately 7 km) to seven beaches encompassing approximately 35 km of nesting beaches, which included an agreement by participating villages to no longer harvest eggs (Kinch 2006; Pilcher 2013). Of these seven beaches, Kamiali was the nesting beach with the longest running, most consistent monitoring within the Huon Coast nesting beach complex. At Kamiali, 194 females were tagged between 1999 and 2012, and an average of 77 nests laid per winter nesting season between 2005/2006 and 2012/2013 (Pilcher 2010, 2011, 2012, 2013; Pilcher and Chaloupka 2013). While we are unable to interpret an overall trend from these studies, anecdotal reports from villagers and historic information indicates that leatherback nesting activity was significantly greater in past decades (Benson *et al.* 2007b, 2015; Hirth *et al.* 1993; Kinch 2006; Bellagio Sea Turtle Conservation Initiative 2008).

In the Solomon Islands, it is not possible to estimate nesting trends due to non-standardized methods and opportunistic monitoring efforts over time. Available datasets cannot be compared due to differences in methodology and do not meet our criteria (*i.e.*, nest count data consistently collected in a standardized approach for at least 9 years). Historically, nesting was reported at more than 15 beaches in the Solomon Islands, which may have totaled several hundred nests per season (McKeown 1977; Vaughan 1981). Currently, nesting activity occurs primarily in eight locations (Pita 2005; Dutton *et al.* 2007; Benson *et al.* 2018a; Jino *et al.* 2018). However, due to the remoteness of these areas and lack of systematic surveys, and likely additional undocumented nesting beaches, additional low numbers of nesting leatherback turtles are likely to exist in Solomon Islands. For example, nesting activity was recently identified on Vanugnu

Island, where 23 nests were recorded and 11 females nested between 2011 and 2014 (Jino *et al.* 2018). Additionally, it is unknown to what extent females use multiple beaches throughout the Solomon Islands, or those in Papua New Guinea, and what proportion of females nest in the summer versus winter (Benson *et al.* 2007b; Jino *et al.* 2018; TNC-Solomons 2018 unpublished). While we are unable to interpret an overall trend, local villagers indicate that leatherback nesting was greater in past decades (Bellagio Sea Turtle Conservation Initiative 2008; Benson *et al.* 2007b; Benson *et al.* 2015).

In Vanuatu, anecdotal information suggests that nesting has declined over time (Petro *et al.* 2007). During the 2010/2011 winter nesting season, 41 nests were laid at Votlo Beach, Epi Island, and, during the 2014/2015 nesting season, three females laid 15 nests (WSB 2011, 2016). It is not possible to estimate nest trends due to non-standardized methods and opportunistic monitoring efforts over time, which render existing data incomparable and do not meet our criteria (*i.e.*, nest count data consistently collected in a standardized approach for at least 9 years).

In addition to an overall declining nest trend, the West Pacific DPS exhibits low reproductive output (*i.e.*, low hatching success), due in part to a combination of past and current threats (*e.g.*, beach erosion, predation, and beach temperatures).

The DPS exhibits low productivity (*i.e.*, low hatching success), and the overall nest trend is declining, likely due to anthropogenic and environmental impacts at nesting beaches and in foraging habitats (Tiwari *et al.* 2013). We conclude that the declining nest trend and low reproductive output place the DPS at elevated extinction risk, especially given the low nesting female abundance.

### *Spatial Distribution*

The West Pacific DPS nests throughout four countries with a broad, diverse foraging range. It exhibits metapopulation dynamics and fine-scale population structure.

Aerial surveys conducted between 2004 and 2007 identified Indonesia, Papua New Guinea and Solomon Islands as the core nesting areas for the DPS (Benson *et al.* 2007a; Benson *et al.* 2007b; Benson *et al.* 2011; Benson *et al.* 2018b). During the nesting season, nesting females generally stayed within 300 km or less of these nesting beaches, although a few females were documented visiting multiple beaches during a nesting season (Benson *et al.* 2007b). Distributing nesting activity among various habitats may help to buffer some of the population from impacts at a single nesting area, but the majority of females utilize one nesting area during a nesting season (Benson *et al.* 2011).

Migration and foraging strategies vary based on nesting season, likely due to prevailing offshore currents and seasonal monsoon-related effects experienced by the turtles as hatchlings (Gaspar *et al.* 2012). The lack of crossover among seasonal nesting populations suggests that leatherback turtles develop fidelity for specific foraging regions, likely based on juvenile dispersal patterns (Benson *et al.* 2011; Gaspar *et al.* 2012; Gaspar and Lalire 2017). Oceanic currents help to structure the spatial and temporal distribution of juveniles and lead them to foraging and developmental habitats (*e.g.*, the North Pacific Transition Zone) and to undertake seasonal migrations seeking favorable oceanic habitats/temperatures and abundant foraging resources, such as the central California ecoregion (Gaspar and Lalire 2017). Inter-annual or long-term variability in dispersal patterns can influence population impacts or resilience to regional or Pacific Ocean perturbations (*e.g.*, exposure to fisheries, ENSO events, etc.). Stable isotopes, linked to particular foraging regions, confirm nesting season fidelity to specific

foraging regions (Seminoff *et al.* 2012). Size differences are also apparent, with slightly larger adults appearing to exploit distant temperate foraging habitats regardless of nesting season (Benson *et al.* 2011; Lontoh 2014).

Summer nesting females forage in Northern Hemisphere habitats in Asia and the Central North Pacific Ocean, while winter nesting females forage in tropical waters of the Southern Hemisphere in the South Pacific Ocean (Benson *et al.* 2011; Harrison *et al.* 2018). This variance in foraging strategy results in a foraging range that covers much of the Pacific Ocean: Tasman Sea; East Australian Current; eastern and western South Pacific Ocean; Indonesian, Sulu and Sulawesi, and South China Seas; North Pacific Transition Zone; equatorial currents; and central California ecoregion (Benson *et al.* 2011; Lontoh 2014; Harrison *et al.* 2018; Jino *et al.* 2018). Different strategies result in demographic differences within the DPS which may affect productivity and reproductive output. For example, leatherback turtles that exploit distant temperate foraging habitats (*e.g.*, central California) may require multiple years of seasonal foraging before returning to nesting beaches, due to greater energetic demands. In contrast, leatherback turtles exploiting geographically closer, year-round prey resources in more tropical habitats (*e.g.*, Sulu Sulawesi and South China Seas) may remigrate annually (Lontoh 2014).

The DPS also exhibits genetic population structure. While mtDNA analyses of 106 samples from Indonesia, Papua New Guinea, and Solomon Islands did not detect genetic differentiation among nesting aggregations (Dutton *et al.* 2007), microsatellite DNA analyses indicate fine-scale genetic structure (Dutton 2019; NMFS SWFSC unpublished data).

The wide distribution and variance in foraging strategies likely buffers the DPS to some degree against local catastrophes or environmental changes that would limit prey availability. The distribution of nesting beaches throughout four countries, although primarily concentrated in three, helps to buffer the entire DPS from major environmental catastrophes, because disturbances are not likely to similarly affect all countries during the same seasons. Additionally, the fine-scale genetic structure among nesting aggregations is indicative of metapopulation dynamics, which may also provide the DPS with some resilience.

### *Diversity*

The West Pacific DPS exhibits genetic diversity, with six haplotypes identified in 106 samples from Solomon Islands, Papua Barat Indonesia, and Papua New Guinea (Dutton 2006; Dutton *et al.* 2007; Dutton and Squires 2008). This may provide the DPS with the raw material necessary for adapting to long-term environmental changes, such as cyclic or directional changes in ocean environments due to natural and human causes (McElhany *et al.* 2000; NMFS 2017). The population also exhibits temporal nesting diversity, with various proportions of the population nesting during different times of the year (summer versus winter) which helps to increase resilience to environmental impacts. The foraging strategies are also diverse, with turtles using seven ecoregions of the Pacific Ocean. Diverse foraging strategies likely provide some resilience against local reductions in prey availability or catastrophic events, such as oil spills or typhoons, by limiting exposure from a single event to only a portion of the DPS. We conclude that diversity within the DPS provides it with some level of resilience to threats.

### *Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

The destruction or modification of habitat is a threat to this DPS. Primary impacts to nesting beaches include erosion and ocean inundation, which may be caused by natural processes.

Nesting beaches of the West Pacific DPS are dynamic, high profile beaches that are subject to erosion, such as during King Tides (naturally occurring, predictable highest tides), which are common seasonal occurrences. In Indonesia, the Bird's Head Peninsula beaches are also subject to seasonal patterns of erosion and accretion. Changes in the currents brought on by monsoons beginning in September cause major erosion at Jamursba-Medi that often removes the entire beach, making the habitat unsuitable for nesting until accretion begins again in March (Hitipeuw *et al.* 2007). This natural erosion has been documented to impact many nests at Jamursba-Medi (Hitipeuw *et al.* 2007). Arguably, western Pacific leatherbacks have been dealing with such changes in beach habitats over time, and a turtle's long reproductive lifespan in general is designed to sustain nest loss during a few bad years or seasons. For example, during the 2003/2004 nesting season, 80 percent of marked nests at Jamursba-Medi (Warmamed beach) washed away before they hatched (Hitipeuw *et al.* 2007). However, given the low abundance of the population, the loss (or continued loss over time) of nests is a concern.

At Wermon, the inundation of nests from high tides is a threat during the winter months. During the 2008/2009 winter nesting season, 26 percent of nests laid at Wermon were inundated by tidal activity (Wurlianty and Hitipeuw 2009). During the 2004/2005 nesting season, 23 percent of nests were lost to inundation (Wurlianty and Hitipeuw 2005). During the 2003/2004 nesting season, 10.7 percent of all nests at Wermon were below the high water mark and were subsequently washed away by high tides (Hitipeuw

*et al.* 2007). Tapilatu and Tiwari (2007) stressed that any management plan developed for Papua will need to address the impact of inundation and beach erosion.

Beach erosion is also a threat to nests in Papua New Guinea, where strong storms and tidal surges result in substantial erosion and changes to beaches throughout the Huon Coast. For example, much of the Labu Tale nesting beach was lost to erosion during the 2012/2013 nesting season (Pilcher 2013). The differences in beach width along the Huon Coast place some beaches at more risk of inundation and erosion, such as Kamiali Beach, which is half the width and significantly narrower than Busama Beach (Pilcher 2008). At Kamiali, the average distance of nests to the sea was 3.2 m, compared to 6.2 m at Busama; the distances to the vegetation line were comparable across sites (1.3 m and 1.7 m, respectively; Pilcher 2013).

In Vanuatu, there has been low hatching success in some years due to storms, floods, and high water (Petro *et al.* 2007; WSB 2016).

In recent years, management and conservation practices have included relocating erosion-prone nests to bolster hatchling production. However, these projects are funding-dependent throughout the range of the West Pacific DPS. At Jamursba-Medi, “doomed” nests (*i.e.*, those that are likely to be lost to erosion or inundation) are sometimes relocated to a more stable section of beach; 15 nests were relocated during the 2017 summer nesting season (Tiwari *et al.* in prep.). At Wermon, nests are relocated to avoid erosion and tidal inundation, and increasingly due to *Ipomea* root invasion (Tiwari *et al.* in prep), but beach management activities are project-dependent. At Wermon during the 2017/18 winter nesting season, nests could not be relocated because of the lack of permission from the beach owners, and all but three nests washed away (Tiwari *et al.* in

prep). In Papua New Guinea, 22 of 47 nests (47 percent) at Kamiali beach were relocated to protect them from storm surge and erosion during the 2011/2012 nesting season, and 41 percent of nests were relocated during the 2009/2010 season (Pilcher 2012). In the Solomon Islands, efforts to relocate “doomed” nests is an ongoing and necessary management strategy to help bolster hatchling production, given that a large proportion of nests are inundated or have very low hatching success (Goby *et al.* 2010; TDA 2013; Jino *et al.* 2018).

A large, significant portion of nests (*i.e.*, 10.7 percent to nearly all) are exposed to the reduction and modification of nesting habitat, as a result of erosion and inundation. This threat impacts the DPS by reducing nesting and hatching success, which has been documented throughout the nesting range of the DPS (NMFS and USFWS 2013; Bellagio Sea Turtle Conservation Initiative 2008). While West Pacific leatherback turtles have undoubtedly evolved to sustain changes in beach habitats given their proclivity to select highly dynamic and typically narrow beach habitats, and therefore at the population level can sustain some level (albeit unquantified level) of nest loss. However, the increasing frequency of storms and high water events perhaps as a result of climate change can result in increased and perhaps unnatural loss of nests. Such impacts may lower the productivity of the DPS. Based on the information presented above, we conclude that habitat loss and modification is a threat to the DPS.

#### *Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

The primary threat to the West Pacific DPS is the harvest (both legal and illegal) of leatherback turtles and their eggs. Leatherback turtles are protected by regulatory mechanisms in all four nations where the DPS nests, but laws are largely ignored and not

consistently enforced. This is due to the extreme remoteness of beaches, customary and traditional community-based ownership of natural resources (which includes sea turtles), and overall lack of institutional capacity and funding for enforcement. Furthermore, the cultural and socio-economic dynamics in these nations confound community buy-in and conservation efforts (Kinch 2006; Gjersten and Pakiding 2012; von Essen *et al.* 2014). Additionally, there are nuances related to indigenous harvest (and the definition thereof), some of which is permitted in these nations.

Turtle poaching affects both nesting females on beaches and turtles in their foraging habitats (Bellagio Sea Turtle Conservation Initiative 2008; Kinch 2009; Suarez and Starbird 1996; Tiwari *et al.* 2013; WWF 2018). Turtle poaching has been documented in all four countries where this DPS nests. Egg poaching is a well-documented threat (past and current) and is widespread throughout the range of the DPS (Bellagio Sea Turtle Conservation Initiative 2008; NMFS and USFWS 2013; Tiwari *et al.* 2013; Tapilatu *et al.* 2017).

In Indonesia, the poaching of turtles and eggs continues to occur, though egg harvest and exploitation of females has been minimized at Jamursba-Medi and Wermon beaches due to the presence of monitoring programs and educational outreach. Large-scale egg poaching occurred at Jamursba-Medi between 1980 and 1993, whereby approximately 4 to 5 boats per week (from May to August) collected 10,000 to 15,000 eggs per boat (Tapilatu *et al.* 2013). Commercial egg harvest has been effectively eliminated since beach monitoring was established at that beach in 1993 (Hitipeuw *et al.* 2007). However, recent survey efforts indicate that most, if not all, sea turtle eggs (including leatherback turtles) are poached at other Bird's Head Peninsula beaches and

sold in local markets (Tapilatu *et al.* 2017). At Buru Island, Indonesia, between 2016 and 2017, eight females were poached (WWF 2018), and over the past 20+ years, three to five nesting females have likely been taken annually (J. Wang, NMFS, pers. comm., 2018). In 2017, 114 of 203 leatherback nests were harvested at Buru Island (WWF 2018). In 2018, due to education provided by the newly established WWF program on Buru Island, local community-based efforts in four villages now prohibit female and egg harvest. While protective laws exist in Indonesia, enforcement is largely lacking in areas where monitoring programs do not exist.

In Indonesia, foraging leatherback turtles are also harvested in the waters of the Kei Islands, Maluku Province, where a recognized indigenous subsistence harvest of immature and adult turtles (average size 145 to 170 cm; range 52 to 203 cm) occurs and has likely been a key feature of the local traditional culture for centuries (Compost 1980; Hamman *et al.* 2006; Hitipeuw and Lawalata 2006, 2008). Within the Kei Islands, customary law (“*hak adat*”) authorizes a ritual leatherback turtle hunt in the nine villages of the traditional kingdom of the Nufit people. Starbird and Suarez (1994) brought attention to this hunt when they reported that approximately 200 turtles were harpooned in three months (October to December) of 1994, with as many as 13 taken in one day. Over the past three decades, sporadic monitoring efforts have estimated that up to 100 individuals are harvested annually (Suarez and Starbird 1996; Hitipeuw and Lawalata 2008; WWF 2018). At one point, it was assumed that harvest pressure had declined and was no longer an issue (NMFS and USFWS 2013). However, recent surveys indicate that harvest continues, with conservative estimates of 431 turtles killed over an 8-year period (an average of 53.9 turtles annually), typically between August to February (Hitipeuw

and Lawalata 2008), and at least 103 turtles harvested in 2017 (WWF 2018). Most concerning perhaps is that some of the turtle meat harvested may be commercially sold as dried meat (*i.e.*, leatherback “jerky” locally known as dendeng), which is illegal to sell and inconsistent with indigenous traditional practices. Of four genetic samples acquired in 1995 from turtles harvested in the Kei Islands, three were assigned to Birds Head Indonesian region and the fourth sample was not definitive (66 percent probability, with 34 percent probability to Solomon Islands), although it could also be from the Indian Ocean or from an undetermined location (NMFS SWFSC unpublished data 2018).

In Papua New Guinea, turtle and egg poaching is a major threat despite the fact that leatherback turtles have been protected since the 1976 Fauna (Protection and Control) Act. The illegal take of both eggs and turtles likely continues throughout the country due to lack of community-based awareness, reliance on traditional community-based practices, institutional capacity, and law enforcement (Bellagio Sea Turtle Conservation Initiative, 2008). The killing of nesting females has also been well documented throughout Papua New Guinea (Bellagio Sea Turtle Conservation Initiative 2008; Kinch 2009; Pilcher 2013). For example, at Bougainville Island, surveys of community members identified that 21 nesting females were poached during the last decade (Kinch 2009). However, the harvest of eggs is likely the most prolific threat in Papua New Guinea. If unprotected, egg harvest (compounded by intense dog predation described below) resulted in the loss of 70 to 100 percent of nests (Quinn and Kojis 1985; Hirth 1993; Bellagio Sea Turtle Conservation Initiative 2008; Pilcher 2013). For example, during a one week survey in January 2009 at Bougainville Island, almost 100 percent of the 46 documented nests were poached (Kinch 2009). It is likely that near total

egg collection occurred throughout the Huon Coast between World War II and the establishment of the Huon Coast Leatherback Turtle Monitoring and Conservation Program in 2003 (Bellagio Sea Turtle Conservation Initiative 2008; Pilcher and Chaloupka 2013; Pilcher 2013). The Huon Coast Project, which operated between 2003 and 2013, helped to reduce egg and turtle harvest due to program involvement and community incentive funds received in exchange for non-harvest agreements (Pilcher 2013). As a result of the program, hatchling production (*i.e.*, percent of eggs yielding hatchlings) increased from zero to approximately 60 percent (Pilcher 2009, 2011, 2013; WPRFMC 2015). The Project ended in 2013, and unfortunately egg harvest resumed since there was no incentive for communities to maintain their no-harvest agreements (John Ben, Huon Coast Leatherback Turtle Project, pers. comm., 2020).

In Vanuatu and the Solomon Islands, the poaching of females and collection of eggs is also well documented (Bellagio Sea Turtle Conservation Initiative 2008; NMFS and USFWS 2013). In Vanuatu, MacKay *et al.* (2014) reported the harvest of five nesting females between 1999 and 2008. However there is a general understanding that nesting females were typically harvested (Petro *et al.* 2007). Of the 315 nests documented on Isabel Island, Solomon Islands during a January 2011 site visit at Sasokolo and Litogahira beaches, the majority of nests had been poached (Tiwari 2011 unpublished data). Historically, nearly all nesting females and eggs were poached on Redova for consumption (Tiwari 2011 unpublished data). In response, financial incentive programs have been established to protect nests and females whereby villagers are paid a financial reward for each nest that hatches successfully (TDA 2013). On Vangunu Island, 10 to 20 nesting females were poached annually, in addition to near-total egg collection (Jino *et*

*al.* 2018). In response to declining population trends, the community declared a moratorium on the harvest of leatherback turtles in 1999 (Jino *et al.* 2018), and a community incentive program providing financial awards has helped to reduce harvest pressure (TDA 2013). Despite these efforts and protective legislation, the poaching of females and eggs likely persists throughout the Solomon Islands (TDA 2013: Tiwari 2011 unpublished; MacKay *et al.* 2014).

Within the West Pacific DPS, many nesting females, foraging turtles, and eggs are exposed to both illegal poaching and legal harvest. The taking of turtles reduces abundance. The taking of nesting females reduces both abundance and productivity. Such impacts are high because they directly remove the most productive individuals from the DPS, reducing current and/or future reproductive potential. Egg harvest reduces productivity; the persistent, and near-total (at some locations) collection of eggs guarantees that future population recruitment (*i.e.*, nesting female abundance) will be reduced or eliminated. Given the declining nesting trend and current nesting female abundance of this DPS, the continued and unregulated poaching or harvest of leatherback turtles and eggs is unsustainable. Further, the harvest of approximately 100 foraging leatherback turtles annually at the Kei Islands, Indonesia is likely an unsustainable practice given the current low abundance of the population. We conclude that overutilization is a major, and the primary, threat to the West Pacific DPS, accelerating its risk of extinction.

#### *Disease or Predation*

While we could not find any information on disease for this DPS, predation of eggs is a major and well-documented threat to the West Pacific DPS, likely second to

poaching (*i.e.*, nests not taken by humans are typically predated; Bellagio Sea Turtle Conservation Initiative 2008).

In Indonesia, predation of eggs by feral pigs, feral dogs, and monitor lizards has been documented, with feral pig predation being the most detrimental (Hitipeuw and Maturbongs 2002; Tapilatu and Tiwari 2007; Bellagio Sea Turtle Conservation Initiative 2008). Nest predation by domestic and/or feral dogs has been recorded in both Jamursba-Medi and Wermon. Predation of nesting females by crocodiles has also been documented at Wermon beach (Bellagio Sea Turtle Conservation Initiative 2008; UNIPA, pers. comm., 2018). At Jamursba-Medi, between June and July of 2005, 29.3 percent of nests were destroyed by pigs (Tapilatu and Tiwari 2007). Intensive management effort at Jamursba-Medi reduced feral pig predation of nests to five percent during the 2016 and 2017 nesting seasons (Tiwari *et al.* in prep). Feral pigs and dogs depredated 17.5 percent of all nests at Wermon during the 2003 and 2004 winter nesting season (Hitipeuw *et al.* 2007). At Wermon, 21 percent of nests were lost to predation during the 2004/2005 nesting season (Wurlianty and Hitipeuw 2005). At Buru Island in 2017, 16 nests were lost to predation by dogs, wild boar, lizards, or saltwater crocodiles (WWF 2018).

In Papua New Guinea, predators of eggs include feral dogs, monitor lizards, and ghost crabs (Kinch 2009). Depredation of nests by village dogs was determined to be an intense threat to nests, with dogs consuming all nests laid during the 2003/2004 and 2004/2005 nesting seasons at Kamiali beach (Pilcher 2006; I. Kelly, NMFS, pers. comm., 2018). Predation of nesting females by crocodiles has also been documented in a number of locations in Papua New Guinea (Bellagio Sea Turtle Conservation Initiative 2008; Kinch 2009). To protect nests, Huon Coast communities developed and placed bamboo

grids over nests to prevent dogs from preying on the eggs (Pilcher 2006; 2009). This, along with efforts to reduce egg harvest by humans, resulted in increased hatching production from zero to approximately 60 percent between 2006 and 2013, with over 2,300 nests saved producing approximately 100,000 hatchlings (Pilcher 2009; 2011; 2013; WRFMC 2015). However, this project ended in 2013, and it is unknown if egg protection continues, or if nest predation has resumed.

In this DPS, a large proportion of eggs are exposed to predation, especially by dogs and pigs. Predation primarily results in the loss of eggs, and the impact of this threat is a reduction of productivity. Though leatherback turtles generally produce a large number of eggs and hatchlings, predation is widespread throughout the range of the DPS, and in some areas, predation rates are as high as 100 percent. We conclude that predation poses a threat to the West Pacific DPS.

#### *Inadequacy of Existing Regulatory Mechanisms*

The West Pacific DPS is protected by several regulatory mechanisms. For each, we review the objectives of the regulation and to what extent it adequately addresses the targeted threat.

Leatherback turtles are protected by legislation in all four of the nations where the West Pacific DPS nests (Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu). It is generally illegal to harvest leatherback turtles and their eggs. However, laws are not typically enforced or followed given customary marine tenure systems that dictate near-shore rights. Lack of enforcement or implementation of protective laws may be due to: overall lack of in-country institutional capacity and funding for enforcement; the extreme remoteness of beaches; customary marine tenure or traditional community-based

ownership of natural resources in these nations (which includes sea turtles; Kinch 2006; McDonald 2006) and regulatory government-led legislation, which may be incompatible with traditional practices (von Essen *et al.* 2014). There are also nuances related to indigenous harvest (and the definition thereof), which is not prohibited in these nations. As a result, most leatherback nesting beaches with the exception of Jamursba-Medi and Wermon (*i.e.*, beaches with established long-term monitoring programs) are not currently protected (or only minimally protected) from harvest or poaching of eggs, nesting females, or other anthropogenic threats.

In Indonesia, all sea turtles are protected by law, but there are allowances for indigenous peoples (although indigenous provisions are not clearly defined). The 1990 Government Regulation Act number 5 concerning the Conservation of the Natural Resources and the Ecosystem, makes the trade of protected wildlife illegal, and those found liable can be punished to a maximum of 5-year prison term and fined 100 million Indonesia Rupiah (approximately 6,500 USD). The protection of all sea turtle species (Government Regulation No. 7 on Preserving Flora and Fauna Species) came into effect in 1999 (Zainudin *et al.* 2007). The use of protected wildlife is allowed for the purposes of research, science, and rescue of the wildlife itself. While the trade and exploitation of turtles is illegal in Indonesia, there still exists a documented harvest of green turtles in Bali, which contributes to public confusion regarding sea turtle protections (Westerlaken 2016).

In Papua New Guinea, the leatherback turtle is the only species protected under the 1976 Fauna (Protection and Control) Act, which makes killing of leatherback turtles or taking of leatherback turtle eggs illegal, with fines of 500 to 1000 kina (approximately

100 to 300 USD). Any person who buys or sells or offers for sale, or has in possession leatherback turtle eggs or meat can also be fined. The Act makes provisions for persons with customary rights to take turtles, but states that sea turtles cannot be taken, killed, or sold from May through July (Kinch 2006). This is typically the nesting season for hard-shelled sea turtle species, but leatherback turtles nest primarily during the winter months (November to February). As with most Melanesian countries, lands are locally-owned and managed, and the national government has little influence outside major cities (Kinch 2006).

The Solomon Islands Fisheries Act (1993) regulations protect nesting turtles and eggs during the breeding season (June to August and November to January); prohibit the sale, purchase, or export of sea turtle species or their parts; and contain specific protections for leatherback turtles. In the Solomon Islands, more than 85 percent of the land is held under customary (locally-managed) marine tenure, and the vast majority of the population still lives in rural areas making a living from the natural resources on those lands. For centuries, communities have practiced traditional models of resource stewardship, making implementation and enforcement of national regulations nearly impossible. Instead, natural resource governance must originate from chiefs and village leaders, which requires extensive educational outreach to encourage traditional approaches that may be supported by legal or 'modern' enforcement measures (McDonald 2006).

Fisheries Regulations under the Vanuatu Fisheries Act (2009) prohibit the take, harm, capture, disturbance, possession, sale, purchase of or interference with any turtle nest (or any turtle in the process of nesting) and the import, or export of green, hawksbill,

and leatherback turtles or their products (shell, eggs, or hatchlings). The Act also prohibits the possession of turtles in captivity. A person may apply in writing to the Director of Fisheries for an exemption from all or any of these provisions for the purposes of carrying out customary practices, education, and/or research. Similar to Papua New Guinea and the Solomon Islands, natural resource governance in Vanuatu is best directed, realized, and implemented at the community level and not via national legislation. Fortunately, traditional practices are experiencing a renaissance in Vanuatu and may complement current regulatory marine resource management efforts (Hickey *et al.* 2006).

Throughout the foraging range of the DPS, there are numerous regulatory mechanisms that protect turtles within the DPS. These include: RFMOs such as the Western and Central Pacific Fisheries Commission (WCPFC) and the IATTC and fisheries management regulations in 32 nations where this DPS may occur (Harrison *et al.* 2018). The WCPFC adopted a Sea Turtle Conservation and Management Measure (CMM 2008-03) to mitigate the impacts on turtles from commercial shallow-set fisheries operating in the Western and Central Pacific Ocean. The measure included the adoption of FAO (2009) guidelines to reduce sea turtle mortality through safe handling practices and to reduce bycatch by implementing one of three methods by January 2010. The three methods to choose from are: 1) use only large circle hooks with offsets of  $\leq 10^\circ$ ; 2) use whole finfish bait; or 3) use any other mitigation plan or activity that has been approved by the Commission. This sea turtle conservation measure is specific to self-identified shallow-setting, swordfish-targeting fleets. It does not apply to the international Pacific longline deep-set tuna-targeting fisheries, which comprise the majority of the longline

fisheries and are also known to interact with leatherback turtles (Lewison *et al.* 2004; Beverly and Chapman 2007; Roe *et al.* 2014; Wallace *et al.* 2013). Technical analysis of the sea turtle conservation measure found a very small percentage of shallow-set fisheries to be in compliance, with less than one percent of Western and Central Pacific Ocean longline effort implementing mitigation measures, even though approximately 20 percent of longline effort consists of shallow sets (Clarke 2017). Further, many RFMO members are not meeting the five percent observer coverage requirement resulting in limited bycatch reporting (Clarke 2017).

In summary, regulatory mechanisms exist to protect leatherback turtles and their eggs throughout the range of this DPS. However, most are inadequate to reduce the threat that they were designed to address due to a lack of implementation or enforcement or inclusion of provisions for indigenous harvest. Regulations are also misaligned with established traditional practice and management systems. As a result, poaching and bycatch remain major threats to the DPS. In summary, we consider the inadequacy of the regulatory mechanisms to be a threat to the DPS.

### *Fisheries Bycatch*

Fishery bycatch in coastal and pelagic fisheries is a major threat to the West Pacific DPS, which is exposed to domestic and international fisheries throughout its extensive foraging range. At-sea bycatch of leatherback turtles has been documented for a variety of gillnet and longline fisheries in the Pacific Ocean, but little is known about the total magnitude or full geographic extent of mortality. Satellite telemetry studies have identified movements and revealed fidelity to foraging regions of the DPS, specifically in habitats of the North Pacific Ocean, southwestern Pacific Ocean, and Indo-Pacific

tropical seas (Bailey *et al.* 2012; Benson *et al.* 2011, Seminoff *et al.* 2012; Roe *et al.* 2014). The summer nesting component of the population exhibits strong site fidelity to the central California foraging area (Benson *et al.* 2011) which puts them at risk during migrations of interacting with U.S. and international pelagic longline fleets operating throughout the Central and North Pacific Oceans. For example, several of the turtles tagged in Papua Barat, Indonesia were known or suspected to have been killed in fisheries operating off Japan, Philippines, and Malaysia (Benson *et al.* 2011).

Historically, significant leatherback bycatch was documented in the North Pacific high seas driftnet fishery, which expanded rapidly during the late 1970s but was banned in 1992 by a UN resolution (summarized in Benson *et al.* 2015). Wetherall *et al.* (1993) estimated that over 750 leatherback turtles were killed in Japanese, Korean, and Taiwanese driftnet fisheries during the 1990 to 1991 season, with potentially 5,000 to 10,000 leatherback turtles bycaught between the late 1970s and 1992. Based on current knowledge of movement patterns (Benson *et al.* 2011), the majority of these bycaught turtles would have originated from western Pacific nesting beaches after their boreal summer nesting period. Thus, high seas driftnet fishery bycatch was likely a significant contributor to the population declines observed at nesting beaches during the 1980s and 1990s (Benson *et al.* 2015).

Many nations are involved in longline fishing in the Pacific Ocean, where two types of vessels are used: (1) large distant-water freezer vessels that undertake long (months) voyages and operate over large areas of the region; and (2) smaller offshore vessels with ice or chill capacity that typically undertake trips of about one month. Target species are yellowfin, bigeye, albacore tuna, and swordfish. The total annual number of

longline vessels in the western and central Pacific region has fluctuated between 3,000 and 6,000 for the last 30 years, including the 100 to 140 vessels in the Hawaii longline fisheries (NMFS 2018).

### Pelagic Fisheries

International longline fisheries are characterized by inconsistent reporting and traditional gear configurations, including J-style hooks with squid bait, which result in higher interaction and mortality rates than for modified gear (Beverly and Chapman 2007; Lewison *et al.* 2004; Swimmer *et al.* 2017). For example, the Taiwan and China tuna longline fisheries are estimated to have bycatch rates several times higher than Hawaii longline fisheries (Bartram and Kaneko 2008; Chan and Pan 2012). Analyzing multi-national turtle bycatch data from 1990 to 2004, Molony (2005) found that the purse seine fishery and the deep, shallow, and albacore longline fisheries (operating between 15 °N and 31 °S) take an average of about 100 leatherback turtles annually. Lewison *et al.* (2004) collected fish catch data from 40 nations and turtle bycatch data from 13 international observer programs to estimate global longline bycatch of loggerhead and leatherback turtles in 2000. In the Pacific Ocean, they estimated 1,000 to 3,200 leatherback turtle (juvenile and adult) mortalities from pelagic longlining in 2000 (Lewison *et al.* 2004). Using effort data from Lewison *et al.* (2004) and bycatch data from Molony (2005), Beverly and Chapman (2007) estimated sea turtle longline bycatch to be approximately 20 percent of that estimated by Lewison *et al.* (2004), approximately 200 to 640 leatherback turtles annually. These estimates include turtles from the East and West Pacific DPS. While the results of each of these studies may be feasible, the Lewison *et al.* 2004 estimates were based on available data at that time (*i.e.*, less than 30 percent

of longline fishing effort) that was skewed toward fishing fleets with relatively better management and data reporting systems, and hence extrapolations may have overestimated interaction rates (Clarke *et al.* 2014). However, Beverly and Chapman (2007) applied different catch per unit effort (CPUE) estimates in calculations differentiated between deep-set and shallow-set fisheries which have different interaction rates and, hence, their estimates may be more realistic.

Despite scientific evidence showing that use of circle hooks and finfish bait significantly reduces leatherback turtle bycatch rates in longline fisheries (Gilman *et al.* 2007; Swimmer *et al.* 2017), nations are not required to use this hook/bait combination. The WCPFC Sea Turtle Conservation and Management Measure (CMM 2008-03) only applies to fleets using shallow-set gear targeting swordfish. Additionally, observer program coverage levels in WCPFC longline fisheries have not reached the required five percent coverage rate, resulting in limited bycatch reporting and likely underreporting (Clarke 2017). Further, existing sea turtle mitigation measures are currently only being applied to approximately one percent of shallow-set longline fisheries in the Convention Area, even though approximately 20 percent of the longline effort consists of shallow-sets (Clarke 2017).

A workshop convened to assess the effectiveness of WCPFC's Sea Turtle Conservation and Management Measure found limited reductions in interactions and mortalities (Clarke 2017). Fishery observer data collected between 1989 and 2015 of 34 purse seine and longline fleets across the Pacific documented a total of 2,323 sea turtle interactions, of which 331 were leatherback turtles (Clarke 2017). Two bycatch hotspot areas were identified: one in central North Pacific (which likely reflects the 100 percent

observer coverage in the Hawaii shallow-set longline fishery) and a second hotspot in eastern Australia (Clarke 2017). However, analysis of the data also found that overall conservation benefits would have been greater had mitigation measures also been applied to deep-set gear and not only to shallow-set swordfish fisheries (Clarke 2017).

While bycatch in pelagic shallow-set swordfish-targeting longline fisheries has received the most attention to date, comparable studies for deep-set tuna-targeting fisheries are not available due to the more complex nature of these fisheries. There may be fewer interactions because deep-set fisheries (operating at depths more than 60 m) generally have lower bycatch rates, but they also have higher mortality rates than shallow-set gear (Lewison *et al.* 2004; Kaplan 2005; Gilman *et al.* 2007). Pelagic deep-set tuna-targeting fisheries cannot be ignored because they also have the potential to interact with leatherback turtles and constitute four times greater effort than shallow-set fisheries yet do not have RFMO gear mitigation requirements (Clarke 2017).

Wallace *et al.* (2013), and a global review based on that study (FAO 2014), categorized longline and gillnet fisheries interactions with West Pacific leatherback turtles as high risk but low impact for longline and gillnet gear, likely due to insufficient data from this data-poor region. Bycatch in small-scale coastal fisheries has been a significant contributor to population declines in many regions (Kaplan 2005; Peckham *et al.* 2007; Alfaro-Shigueto *et al.* 2011), yet there is a significant lack of information from coastal and small-scale fisheries, especially from the Indian Ocean and Southeast Asian region (Lewison *et al.* 2014).

Southeast Asian Fisheries

Waters of Southeast Asia are heavily fished by a variety of gillnets, trawls, fish traps, and a range of different hook and line gears, involving hundreds of thousands of fishers (FAO 2011). The West Pacific DPS nests, migrates, and forages throughout this densely populated and heavily exploited coastal region (Bellagio Sea Turtle Conservation Initiative 2008; Benson *et al.* 2011; Lewison *et al.* 2014; Roe *et al.* 2014; Harrison *et al.* 2018).

There are few quantitative estimates of fisheries interactions near nesting beaches of this DPS, and existing reports provide only brief snapshots of impacts or are outdated. In Indonesia, between 1980 and 1993, shark gillnets off the nesting beaches of Jamursba-Medi killed two to three nesting females weekly (Tapilatu *et al.* 2013). As a member of the WCPFC and the IOTC, Indonesia must comply with reporting requirements and conservation measures as required by these RFMOs. In 2006, of the 85 sea turtle interactions observed in 539 sets on 10 tuna longline vessels, 3 were adult leatherback turtles (Zainudin *et al.* 2007). Leatherback turtles are known to migrate through and forage within Philippine waters (Benson *et al.* 2011), and in 2014, aerial surveys observed leatherback turtles foraging in high density fishing areas (130 to 381 boats; MRF 2010, 2014). Leatherback turtles have also stranded dead or injured on Philippine beaches as a result of fishery interactions, typically with gillnet gear (Bagarinao 2011; Cruz 2006; MRF 2010; MWWP 2018 unpublished). In Malaysia, bycatch studies using an interview-based approach revealed that four leatherback turtles were caught in gillnets the prior year (Pilcher *et al.* 2008).

Fisheries operating out of Australia and New Zealand may result in high bycatch and mortality rates for the winter nesting component of the DPS that migrates into the

Southern Hemisphere (MacKay *et al.* 2014; Harrison *et al.* 2018). In Australia, some bycatch records exist for pelagic longline fisheries (Robins *et al.* 2002; Stobutzki *et al.* 2006), prawn trawls off Queensland and Northern Territory, gillnet fisheries off Queensland and Tasmania, and pot gear off Tasmania (Limpus 2009). Gillnet sea turtle bycatch is reported as widespread and includes anecdotal reports of leatherback turtles taken in Tasmanian tuna gillnet fisheries (Limpus 2009).

Between 2004 and 2014, the Australian shallow-set fishery had an estimated 29 to 178 leatherback interactions, based on two to 10 observations (average = 4.6 interactions) and four to 10 percent observer coverage (MacKay *et al.* 2014). These data are similar to bycatch information extrapolated from interviews with Australian fishers (Robins *et al.* 2002) which identified 162 leatherback turtles interactions in 2001 (MacKay *et al.* 2014). Australia has a sea turtle mitigation plan for its Eastern Tuna and Billfish Fishery which sets “trigger level” interaction rates of  $\leq 0.0048$  turtles per 1,000 hooks for each turtle species or 0.0172 turtles per 1,000 hooks overall (DAFF 2009 in Clarke *et al.* 2014). In 2013, Australia reported that the trigger levels had been exceeded for the third year in a row and as a consequence the Australian Fisheries Management Authority required that shallow-set vessels in these fisheries use large circle hooks consistent with the WCPFC sea turtle measure (CMM 2008-03; Clarke *et al.* 2014).

In New Zealand, records document 288 instances of stranding or commercial and recreational bycatch of leatherback turtles from 1892 to 2015 (Godoy *et al.* 2016). New Zealand’s surface longline fishery captured 90 leatherback turtles between 2008 and 2015 (Godoy *et al.* 2016). This is likely an underestimate because data were based on low observer coverage (5.8 percent overall), with limited observer coverage during the peak

time of leatherback abundance in New Zealand waters (January to March). Strandings can also provide opportunities for researchers to identify fisheries interactions. MacKay *et al.* (2014) identified 19 mortalities in New Zealand and 29 mortalities in Australia. Although the cause of most strandings was often unknown, leatherback turtles have been found entangled in crab pot gear and monofilament fishing nets and ropes. Longline fishing is concentrated off southern Queensland and New South Wales, Australia and is the suspected cause of 41 percent of strandings ( $n = 12$ ). In Victoria, Tasmania and South Australia, 61 percent of strandings ( $n = 17$ ) involved suspected entanglement in inshore fishing gear and crab pots (MacKay *et al.* 2014).

#### U.S. Pacific Pelagic Fisheries

Detailed bycatch data are available for U.S.-managed pelagic fisheries operating in the central and eastern Pacific Ocean due to regulatory mandates and high levels of observer coverage. Longline fisheries, based out of Hawaii and American Samoa, may interact with foraging turtles of the West Pacific DPS. However, only two interactions involved individuals of the East Pacific DPS in 1995 and 2011 (P. Dutton, NMFS, pers. comm., 2018). Prior to 2001, the Hawaii longline fishery was estimated to capture about 110 leatherback turtles annually, resulting in approximately 9 annual mortalities (McCracken 2000). Since 2005, the fishery has reduced its estimated mortality to seven leatherback turtles annually, and data confidence increased significantly due to increased observer coverage (NMFS 2018). The fishery was closed in 2001 under court order and re-opened in 2004 as two separate fisheries: a shallow-set swordfish-targeting fishery and a deep-set tuna-targeting fishery. Management requirements include: gear modification (*e.g.*, circle hooks and fin-fish bait) and handling measures designed to reduce sea turtle

bycatch rates and post-hooking mortality in both fisheries; an annual hard-cap limit on the number of allowable interactions in the shallow-set fishery; 100 percent observer coverage in the shallow-set fishery; and 20 percent observer coverage in the deep-set fishery (50 CFR 665 (Subparts A-C); NMFS 2012, 2014, 2015). The shallow-set fishery has been closed three additional times since reopening in 2004: in 2006, after reaching the hard cap for loggerhead turtle interactions (n = 17); in 2011, after reaching the hard cap for leatherback turtle interactions (n = 16); and in 2018 under a stipulated settlement after the Ninth Circuit Court of Appeals held that NMFS' no jeopardy determination for loggerheads in the 2012 biological opinion (9th Circuit 2017) was arbitrary and capricious. *See Turtle Island Restoration Network v. U.S. Dep't. of Commerce*, 878 F.3d 725 (9<sup>th</sup> Cir. 2017). Since 2004, leatherback turtle interactions in the shallow-set component of the fishery have been reduced by 84 percent from 0.03 to 0.01 BPUE as a result fisheries regulations (Swimmer *et al.* 2017). Between 2004 and 2017, there have been 99 total leatherback turtle interactions in the shallow-set fishery (or approximately 8 turtles annually), based on 100 percent observer coverage (WPRFMC 2018). Between 2002 and 2016, an estimated 168 interactions may have occurred in the Hawaii deep-set fishery (or approximately 12 annually), based on an extrapolation of data collected at a level of 20 percent observer coverage (WPRFMC 2018). Observer coverage of the American Samoa longline fishery has varied over time from 5 to 40 percent and has had an estimated 59 interactions between 2010 and 2017 (WPRFMC 2018).

The U.S. tuna purse seine fishery operating in the Western and Central Pacific Ocean anticipates up to 11 leatherback turtle interactions annually (NMFS 2006). However, the fishery had fewer interactions, with approximately 16 leatherback turtle

interactions between 2008 and 2015 based on observer coverage ranging from 20 to 100 percent (NMFS unpublished data).

From 1990 to 2009, there were 24 observed leatherback turtle interactions in the California drift gillnet fishery based on 15.6 percent per year observer coverage (Martin *et al.* 2015). Genetic analyses indicated that almost all originated from the West Pacific DPS (Dutton *et al.* 1999; NMFS SWFSC unpublished). In 2001, NMFS implemented regulations (*i.e.*, a large time/area closure in Central California) that reduced interactions by approximately 80 to 90 percent, with only two leatherback turtle interactions (both alive) observed based on 20 to 30 percent observer coverage since regulations were implemented (NMFS West Coast Region unpublished). Drift gillnet fishing is prohibited annually from August 15 to November 15 within the California leatherback turtle conservation area. Currently, NMFS anticipates up to 10 interactions (or 7 mortalities) over a 5-year period (NMFS 2013).

In addition, nine fixed gear fisheries operate off the U.S. West Coast, including the Federally-managed sablefish pot fishery and the state-managed California Dungeness crab fishery. Since 2008, only one leatherback interaction has been documented in the sablefish fishery (NMFS 2013). The state-managed Dungeness crab fishery may be a newly emerging threat: two documented leatherback entanglements in pot gear (mainline or surface buoy) occurred in 2015 and 2016. Fishing effort was high, and the fishery had shifted into the Central California region, which overlaps somewhat with leatherback foraging habitat (S. Benson, NMFS, pers. comm., 2018). In 2019, the State of California settled with a non-profit organization in response to a complaint that the commercial Dungeness crab fishery was taking leatherback sea turtles (and other large whales)

without authorization under section 10 of the ESA. The California Dungeness crab fishery closed in mid-April 2019 as part of the settlement agreement and again on May 15, 2020 (just the Central Management Area), due to significant risk of marine life entanglement. The northern part of California remains open until mid-July unless CDFW decides to take further management action (*i.e.*, if risks to large whales and/or leatherbacks is elevated in that area).

#### East Pacific Pelagic Fisheries

The West Pacific DPS has a vast trans-Pacific range. Some individuals forage in the East Pacific Ocean, where leatherback turtles are caught in fisheries of Peru and Chile (Donoso and Dutton 2010; Alfaro-Shigueto *et al.* 2007, 2011, 2018). Of 59 leatherback turtles caught in East Pacific fisheries, an estimated 15 percent of individuals sampled originated from the West Pacific DPS (Dutton *et al.* 2000; Donoso and Dutton 2010). Information compiled by IATTC on sea turtle interactions with pelagic longline fisheries operating in the East Pacific is limited, given that requirements for longline observer coverage of five percent was only implemented in January 2013 (Clarke *et al.* 2014). Additional information on East Pacific fisheries are presented in the bycatch section for the East Pacific DPS.

#### Summary of Fisheries Bycatch

We conclude that individuals of this DPS are exposed to high fishing effort throughout their foraging range, in coastal waters near nesting beaches, and when migrating to and from nesting beaches, though very little fisheries data are available for coastal areas. Bycatch rates in international pelagic and coastal fisheries are high, and these fisheries have limited management regulations despite hotspots of high interactions

in Southeast Asia (Lewison *et al.* 2004, 2014; Alfaro-Shigueto *et al.* 2011; Wallace *et al.* 2013; Clarke 2017). Annual interaction and mortality estimates are only available for U.S.-managed pelagic fisheries, which operate under extensive fisheries regulations that are designed to minimize the capture and mortality of endangered and threatened sea turtles (NMFS 2013; Swimmer *et al.* 2017; NMFS 2018). Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when nesting females are killed. We conclude that fisheries bycatch is a major threat to the West Pacific DPS.

#### *Vessel Strikes*

Vessel strikes are a threat to the West Pacific DPS. Between 1981 and 2016, there were 11 documented vessel strikes in central California (NMFS West Coast Region, unpublished data 2018). Many vessel strikes are not reported, and turtles are not recovered.

The range of the DPS overlaps with many high-density vessel traffic areas. Though the potential for exposure is high, we are only aware of 11 vessel strikes in recent decades. Vessel strikes resulting in mortality would lower the abundance of the DPS. However, available data does not support characterizing this as a high or moderate impact. We conclude that vessel strikes pose a threat to the DPS, albeit of less concern than other impacts such as overutilization and fisheries interactions.

#### *Pollution*

Pollution includes contaminants, marine debris, and ghost fishing gear. Leatherback turtles can ingest small debris, causing internal damage and blockage. Larger debris can entangle animals, leading to reduced mobility, starvation, and death.

Given the amount of floating debris in the Pacific Ocean (Lebreton *et al.* 2018), marine debris has the potential to be a significant threat to the DPS. Presently available data do not allow for quantifying the precise extent of the threat.

Leatherback turtles feed exclusively on jellyfish and other gelatinous organisms and as a result may be prone to ingesting plastics resembling their food source (Schuyler *et al.* 2013). Lebreton *et al.* (2018) estimated plastic debris accumulation to be at least 79,000 (45,000 to 129,000) tonnes in the Great Pacific Garbage Patch, a 1.6 million km<sup>2</sup> of subtropical waters between California and Hawaii. This figure is four to 16 times greater than previously reported. Entanglement in ghost fishing gear is also a concern (Gilman *et al.* 2016), and derelict nets made up approximately 46 percent by piece, and 86 percent by weight, of debris floating in this area (Lebreton *et al.* 2018). The highest risk areas within the range of the West Pacific DPS where animals may encounter significant amounts of debris includes the north Pacific gyre, the South China Sea, and off of the east coast of Australia (Schuler *et al.* 2015). However, Wedemeyer-Strombel *et al.* (2015) found no plastics in the gastrointestinal tracts of two leatherback carcasses from American Samoan and Hawaiian longline fisheries from 1993 to 2011. Clukey *et al.* (2017) found no plastics in the gastrointestinal tracts of three leatherback carcasses from Pacific longline fisheries captured between 2012 and 2016. However, it is very difficult to obtain dead leatherback turtles to study these effects, and given the great amount of plastics within environment, such results may underestimate ingestion impacts.

Few studies of pollutants and their effect on leatherback turtles were available within the range of this DPS. Harris *et al.* (2011) found the heavy metal exposure in leatherback turtles foraging off the coast of California to be nine times higher than the St.

Croix nesting population, although levels were not expected to be lethal. We do not know if there were sub-lethal effects. Stewart *et al.* (2011) found that PCBs are more likely to be transferred from females to their eggs than from the environment to eggs.

Given the large amount of marine debris within the range of the DPS, we expect exposure to be high for all life stages despite low sample sizes of leatherback turtles with ingested marine debris. Potential impacts include death and injury. However, quantitative estimates of such impacts are not available. We conclude that pollution may be a threat to the DPS.

#### *Natural Disasters*

The best available scientific and commercial data indicate that natural disasters are a threat to the DPS but do not allow the impact to be quantified. Natural disasters within the range of this DPS include: tsunamis, typhoons, earthquakes, and flash floods. Such environmental events are periodic, with localized impacts that do not persist over time. These events may reduce nest incubation and hatching success in one season or at few locations. While leatherback turtles have undoubtedly evolved to sustain such natural impacts, the increasing frequency of environmental events as a result of a changing climate, which can affect the frequency and intensity of high tides and large storms, may hamper productivity and conservation activities (Goby *et al.* 2010; S. Benson, NMFS, pers. comm., 2018). Such events may pose additional threats by depositing marine debris on nesting beaches and in occupied waters. The 2011 Japan tsunami and the 2006 Indonesian earthquake and resulting tsunami likely deposited large amounts of debris (*i.e.*, millions of tons) into the foraging and migrating habitats of the DPS (Hafner *et al.*

2014; NOAA 2015). We conclude that natural disasters pose a potential threat to the West Pacific DPS.

### *Climate Change*

Climate change is a threat to the West Pacific DPS. A warming climate and rising sea levels can impact leatherback turtles through changes in beach morphology, increased sand temperatures leading to a greater incidence of lethal incubation temperatures, changes in hatchling sex ratios, and the loss of nests or nesting habitat due to beach erosion (Benson *et al.* 2015).

Elevated egg incubation temperatures can lead to mortality. During the 2009/2010 nesting season at the Huon Coast (Papua New Guinea), Pilcher (2010) found higher incubation temperatures (32 to 33°C) in exposed nests compared to shaded nests (29 to 30°C). Sea turtles exhibit temperature-dependent sex determination. The incubation temperature determines sex ratios and the duration of incubation (*i.e.*, thermosensitive period). Along the Huon Coast, incubation duration decreased during the nesting season as beach temperatures warmed. During the 2006/2007 nesting season, nests laid in November hatched in  $61.8 \pm 4.2$  days, and nests laid in February hatched in  $55.8 \pm 3.4$  days ( $n = 171$  nests; Steckenreuter *et al.* 2010). Assuming that hatchlings were male at temperatures less than 29.2°C and female at temperatures greater than 30.5°C, Steckenreuter *et al.* (2010) estimated that only 7.7 percent of the hatchlings were female, indicating a highly male-skewed sex ratio. However, given the Pilcher (2010) results, sex ratios are likely variable over time and space.

Climatic change may also alter rainfall levels, which may cool beaches and offset increases in sand temperature. At Wermon, the sand is black, yet beach temperatures are

lower, perhaps because peak nesting coincides with the monsoon season (Tapilatu and Tiwari 2007). Sand temperatures fluctuate between 28.6 and 34.9 °C at Jamursba-Medi and between 27.0 and 32.7 °C at Wermon (Tapilatu and Tiwari 2007). Hatching success of nests undisturbed by feral pig predation was significantly lower in Jamursba-Medi (25.5 percent) than Wermon (47.1 percent). Although there was significant variation between beaches, Tapilatu and Tiwari (2007) concluded that high sand temperatures may exceed the thermal tolerance of leatherback embryos, resulting in high embryo mortality and low hatching success at Jamursba-Medi. Further, Tapilatu and Tiwari (2007) concluded that high average sand temperatures may suggest a female-biased population at Jamursba-Medi. However, the mean incubation period of  $61.5 \pm 4.7$  days (Tapilatu and Tiwari 2007) was similar to the length of incubation recorded in Papua New Guinea during the cooler November period, which Steckenreuter *et al.* (2010) suggested produced a male-biased sex ratio.

Tapilatu *et al.* (2013b) found that the daily average sand temperatures during the boreal summer (from 2005 to 2012) ranged from 26.5 to 34.9°C, suggesting the production of female-biased sex ratios and potentially lower hatching success. Further, histological examination of dead hatchlings from both summer and winter nesting seasons from 2009 to 2019 produced a female-biased sex ratio, which is consistent with the relatively warm thermal profiles of the nesting beaches (Tapilatu *et al.* 2013b). Additional impacts of climate change include increased sea level rise and storm frequency, resulting in greater nest inundation and beach erosion. As sea level rises, King Tides are likely to have a greater effect on nests. Climate change may also affect prey

availability. Saba *et al.* (2007, 2012) identified a correlation between the reproductive frequency of the East Pacific DPS and ENSO events. Because the West DPS also forages in the East Pacific Ocean, it too may be exposed to variability in productivity.

The threat of climate change is likely to modify the nesting and foraging conditions for turtles of the DPS. Impacts are likely to affect productivity. Negative impacts and low hatching success due to high beach temperatures and coastal erosion have already been documented and are likely to become worse, and thus we conclude that climate change is a threat to the West Pacific DPS.

### *Conservation Efforts*

There are numerous efforts to conserve the leatherback turtle. The following conservation efforts apply to turtles of the West Pacific DPS (for a description of each effort, please see the section on conservation efforts for the overall species): Convention on the Conservation of Migratory Species of Wild Animals, Convention on Biological Diversity, Convention on International Trade in Endangered Species of Wild Fauna and Flora, Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention), Convention for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPF Convention), Convention for the Protection of the Natural Resources and Environment of the South Pacific Region, Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention), Eastern Pacific Leatherback Network, Eastern Tropical Pacific Marine Corridor Initiative, FAO Technical Consultation on Sea Turtle-Fishery Interactions, IAC, MARPOL, IUCN, The Memorandum of Understanding of a Tri-National Partnership between the Government

of the Republic of Indonesia, the Independent State of Papua New Guinea and the Government of Solomon Islands, Ramsar Convention on Wetlands, RFMOs, Secretariat of the Pacific Regional Environment Programme, UNCLOS, and UN Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing. Although numerous conservation efforts apply to the turtles of this DPS, they do not adequately reduce its risk of this DPS, they do not adequately reduce its risk of extinction.

#### *Extinction Risk Analysis*

After reviewing the best available information, the Team concluded that the West Pacific DPS is at high risk of extinction. The DPS exhibits a total index of nesting female abundance of 1,277 females at two currently monitored beaches over the most recent remigration interval. These beaches may represent 75 percent of total DPS nesting activity. This abundance makes the DPS vulnerable to stochastic or catastrophic events that increase its extinction risk. This DPS exhibits low hatching success and decreasing nest and population trends due to past and current threats, which are likely to further lower abundance and increase the risk of extinction. The DPS exhibits genetic diversity and metapopulation structure, with nesting aggregations distributed throughout four nations. Nesting occurs during two seasons (winter and summer), with year-round nesting at some locations and uses multiple foraging areas, throughout the Pacific Ocean. Thus, the DPS has some resilience to stochastic events and environmental perturbations at nesting beaches and foraging areas. However, its abundance and declining trends place the DPS at risk of extinction as a result of past threats.

Current threats also contribute to the risk of extinction of this DPS. The overutilization of turtles and eggs, as a result of legal and illegal harvest, is the primary

threat to this DPS, reducing abundance and productivity. Abundance and productivity are further reduced by fisheries bycatch. Juvenile and adult turtles are taken by numerous, international, coastal, and pelagic fisheries throughout the extensive, pan-Pacific foraging range of the DPS. Predation (especially by dogs and pigs) reduces productivity at high rates. Erosion and inundation result in habitat loss and modification that reduces productivity and contributes to low hatching success. Additional threats include: pollution, vessel strikes, and natural disasters. Climate change is an increasing threat that results in reduced productivity. Though many regulatory mechanisms exist, they do not adequately reduce threats.

We conclude, consistent with the team's findings, that the West Pacific DPS is at risk of extinction. Its nesting female abundance makes the DPS highly vulnerable to threats. The declining nesting trend further contributes to its risk of extinction. While the DPS has spatial structure and diversity, the resilience provided by those factors is likely to be eroded by the reduced and declining abundance. Past egg and turtle harvest reduced the abundance and productivity of this DPS and remains a primary threat. Fisheries bycatch is also a primary threat that reduces abundance by removing mature and immature individuals from the population. Predation is also a major threat to productivity. Though numerous conservation efforts apply to this DPS, they do not adequately reduce the risk of extinction. We conclude that the West Pacific DPS is in danger of extinction throughout its range and therefore meets the definition of an endangered species. The threatened species definition does not apply because the DPS is currently in danger of extinction (*i.e.*, at present), rather than on a trajectory to become so within the foreseeable future.

## **East Pacific DPS**

The Team defined the East Pacific DPS as leatherback turtles originating from the East Pacific Ocean, north of 47° S, south of 32.531° N, east of 117.124° W, and west of the Americas. In the south, the cold waters of the Antarctic Circumpolar Current likely restrict the nesting range of this DPS. We placed the northern and western boundaries at the border between the United States and Mexico because this DPS forages primarily in the East Pacific Ocean, off the coasts of Central and South America.

The range of the DPS (*i.e.*, all documented areas of occurrence) is centered in the eastern Pacific Ocean but may include distant waters for foraging, as demonstrated by a turtle satellite-tracked to waters off the Tonga Trench and a turtle captured by the Hawaii longline fishery, genetically assigned to the population we refer to in this finding as the East Pacific DPS (P. Dutton, NMFS, pers. comm., 2018). Records indicate that the DPS occurs in the waters of the following nations: Chile; Colombia; Costa Rica; Ecuador; El Salvador; France (Clipperton Island); Guatemala, Honduras; Mexico; Nicaragua; Panama; Peru; and the United States (Hawaiian Islands) (Wallace *et al.* 2013).

Leatherback turtles of the East Pacific DPS nest primarily on beaches in Mexico, Costa Rica, and Nicaragua. In Mexico, where the largest nesting aggregations occur, nesting beaches are found in 11 states, over 7,828 kilometers as far north as Baja California Sur (Sarti 2002). The following beaches in Mexico host approximately 40 to 50 percent of total nesting for the nation: Mexiquillo (Michoacán), Tierra Colorada (Guerrero), and Cahuitán, Chacahua, and Barra de la Cruz (Oaxaca; Gaona Pineda and Barragán Rocha 2016). In Costa Rica, approximately 75 percent of nesting occurs within the Parque Nacional Marino Las Baulas (Guanacaste Province) at three nesting beaches:

Playa Ventanas; Playa Grande; and Playa Langosta (based on recent abundance estimates from 2011-2015; Santidrián Tomillo *et al.* 2017). In Nicaragua, small numbers of leatherback turtles nest on Playa Salamina-Costa Grande and Veracruz de Acayo (Chacocente Wildlife Refuge) (FFI 2018). Rare nesting events have been documented in Guatemala (n = 6), El Salvador (n = 4), and Panama (n = 4), with none in Honduras (Sarti *et al.* 1999).

Generally, the nesting season starts in October and ends in March (Santidrián Tomillo *et al.* 2007; Eckert *et al.* 2012). Nesting is generally bound between 10° N and 20° N, falling within the northeast corner of the Intertropical Convergence Zone. The nesting beaches share similarly warm temperatures, moderate annual rainfall, and seasonal dynamics (Saba *et al.* 2012). In general, nesting beach habitat for leatherback turtles is associated with deep water and strong waves and oceanic currents, but shallow water with mud banks are also used by leatherback turtles. Beaches with coarse-grained sand and free of rocks, coral, or other abrasive substrates also appear to be selected by leatherback turtles (reviewed by Eckert *et al.* 2012).

Foraging areas of the East Pacific DPS include coastal and pelagic waters of the southeastern Pacific Ocean. Leatherback turtles are widely dispersed on the high seas throughout the eastern Pacific Ocean (Shillinger *et al.* 2008). They also forage in coastal areas off the coast of Peru and Chile (Alfaro-Shigueto *et al.* 2007; Eckert 1997; Donoso and Dutton 2010). Using satellite telemetry, Morreale *et al.* (1996) tracked the movements of eight post-nesting females and identified a persistent southbound migration corridor from Las Baulas National Park toward the Galapagos Islands. Eckert (1997) found a similar pattern, tracking seven post-nesting females from Mexiquillo in a

similar direction; while three continued to the same foraging habitat as the Costa Rican nesting females, four shifted their movements away from the South American coast, when a strong El Niño caused a warm water anomaly. Additional tracking of 46 post-nesting females from Las Baulas National Park over a 3-year period (2004/2005 to 2006/2007) confirmed the persistent migratory corridor (Shillinger *et al.* 2008). The turtles navigated the equatorial current system, south to around 5° S latitude and negotiated the strong alternating eastward-westward flows of the equatorial current, swimming predominantly in a southward direction and moving rapidly through the productive equatorial region. They then dispersed throughout the South Pacific Gyre ecosystem, which is characterized by low phytoplanktonic biomass. The South Pacific Gyre contains ample mesoplankton forage base, as demonstrated by tuna longline fisheries effort in the eastern tropical Pacific Ocean (Shillinger *et al.* 2008). Of the 46 turtles, only one leatherback moved into coastal foraging areas, which had been documented earlier by Eckert (1997). During the course of the tracking duration, this female occupied nearshore foraging habitats along the coast of Central America, which represents highly productive areas when compared with oceanic areas. Researchers have hypothesized that high bycatch along the coastal areas of Central and South America could have extirpated a coastal migratory phenotype in this population (Saba *et al.* 2007). Recently, Harrison *et al.* (2018) determined that post-nesting females from Las Baulas National Park spent 78.2 percent of their time on the high seas, 17.8 percent of their time in Costa Rica's EEZ, and 3.7 percent of their time around the Galapagos Islands.

In summary, preferred foraging areas for the East Pacific DPS are characterized by low sea surface temperatures and high mesoscale variability. Post-nesting females

migrate relatively quickly through areas that contain the strong equatorial currents as well as high chlorophyll-a concentrations, likely because of the strong currents. While swimming speed was significantly higher in areas of high chlorophyll levels, the association between these two variables was weak (Shillinger *et al.* 2008). Once past this area, they appear to forage in the southern part of their range in the South Pacific Subtropical Convergence, where there is a sharp gradient in primary production. In this area, Ekman upwelling may accelerate the transport of nutrients and consequently increase prey availability. Seasonally, leatherback turtles from the East Pacific DPS foraged at higher southerly latitudes during the austral summer (November to February), which may reflect seasonal patterns in prey abundance during higher latitudes (Bailey *et al.* 2012).

### *Abundance*

The total index of nesting female abundance for the East Pacific DPS is 755 females. We based this total index on 13 nesting aggregations in: Mexico (Mexican Commission for Natural Protected Areas; L. Sarti, CONANP, pers. comm. 2018); Costa Rica (Santidrián Tomillo *et al.* 2017; Leatherback Trust 2018); and Nicaragua (FFI 2018). Our total index does not include several unquantified nesting aggregations in Mexico, Costa Rica, and Nicaragua. To calculate the index of nesting female abundance for nesting beaches in Mexico (*i.e.*, 572 females), we added the total number of nesting females between the 2013/2014 and 2016/2017 nesting seasons (*i.e.*, a 4-year remigration interval; L. Sarti, CONANP, pers. comm., 2018) at each beach. We performed a similar calculation for Costa Rica (n =165 females). To calculate the index of nesting female abundance in Nicaragua (*i.e.*, 20 females), we divided the total number of nests between

the 2014/2015 and 2017/2018 nesting seasons (*i.e.*, a 4-year remigration interval; Santradián Tomillo *et al.* 2007) by the clutch frequency (7.2 clutches/season; Santradián Tomillo *et al.* 2007).

This number represents an index of nesting females for this DPS because it only includes available data from recently and consistently monitored nesting beaches. While rare or sporadic nesting may occur on other beaches, consistent and standardized monitoring only occurs at these beaches, which are for the most part protected.

Our total index of nesting female abundance is similar to published abundance estimates for this DPS. The IUCN Red List assessment estimated the total number of mature individuals (males and females) at 633 turtles, based first on dividing the average annual number of nests ( $n = 926$ ) by the estimated clutch frequency ( $n = 7.2$ , Reina *et al.* 2002) to obtain an average annual number of nesting females. This value was then multiplied by the average remigration interval ( $n = 3.7$  years, Reina *et al.* 2002; Santidrián Tomillo *et al.* 2007) to obtain a total number of adult females that included nesting as well as non-nesting turtles. In order to account for adult males, the authors assumed that the sex ratio of hatchlings produced on nesting beaches in the East Pacific (approximately 75 percent female, or 3:1 female:male ratio) reflected the natural adult sex ratio (Wallace *et al.* 2013). A more recent analysis of primary sex ratios that included multiple years of data and considered hatching success (*i.e.*, lower in hot nests) estimated primary sex ratios at Playa Grande, Costa Rica as approximately 85 percent female (Santidrián Tomillo *et al.* 2014). In Mexico, the female to male ratio is closer to 1.1:1 (A. Barragan, Kutzari, pers. comm., 2019).

In Mexico, the beaches included in our total index represent approximately 70 to 75 percent of total nesting in that nation (Gaona Pineda and Barragan Rocha 2016). However, our total index does not include nesting females from Agua Blanca (40 km in Baja California); Playa Ventura (6 km), Playa San Valentín (21 km), Piedra de Tlacoyunque (44 km in Guerrero), and La Tuza (16 km in Oaxaca) (Sarti *et al.* 2007). These beaches are not regularly monitored for nesting, which is thought to be rare or of low abundance (L. Sarti, CONANP, pers. comm., 2018).

In Costa Rica, 75 percent of nesting occurred at Las Baulas National Park (summarized *in* Santidrián Tomillo *et al.* 2017), although the recent nesting at other beaches may lower this percentage. These beaches include: Naranjo, Cabuyal, Nombre de Jesús, Ostional, and Caletas. The longest data set was provided for Naranjo, which has been intermittently covered from 1971 to 2015. Limited nesting has been documented at Playa Coyote and at Playa Caletas, which is a high energy eight kilometer beach located on the Nicoya Peninsula (Squires 1999). Given the lack of nesting events for Caletas in recent years, it may no longer host leatherback nesting, despite the fact that the Playa Caletas/Ario National Wildlife Refuge was created in 2004 to protect leatherback turtles (Gaos *et al.* 2008).

In Nicaragua, leatherback turtles nest at three beaches. Salamina Costa Grande and Veracruz de Acayo (in the Rio Escalante Chacocente Wildlife Refuge) host the most nesting and have been subject to the most consistent monitoring. Small numbers of females also nest at Juan Venado National Reserve, which is not consistently monitored (V. Gadea, FFI, personal communication, 2018).

Nesting is rare in other nations (Sarti *et al.* 1999). Nesting is very uncommon in Ecuador with one record of a female attempting to nest (according to local reports) in Atacames, a province of Esmeraldas (Salas 1981). Sarti *et al.* (1999) reported six nests at Playa Puntilla, El Salvador, but overall nesting is low and/or unknown throughout the nation. In Guatemala, nesting is rare, with reports by Sarti *et al.* (1999) recording only eight nests during an entire season, and more recently, zero to six nests per year along the Pacific coast of Guatemala (Muccio and Flores 2015). Past nesting sites included Hawaii beach, La Candelaria, Taxico, Santa Rosa, and the zone adjacent to the border with El Salvador, as reported by Chacón-Chaverri (2004). Although nesting has been documented at Barqueta National Refuge, little is known about nesting in Panama (Chacón-Chaverri 2004).

Our total index of nesting female abundance (755 females) places the DPS at risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb losses in individuals. Due to its small size, the DPS has relatively little capacity to buffer such losses. Historical abundance estimates were much greater (*e.g.*, 75,000 leatherback nesting females estimated in Pacific Mexico from a 1980 aerial survey ((Pritchard 1982). However, this estimate was derived from a brief aerial survey and may have been an overestimate (Pritchard 1996)), indicating that this population at one time had the capacity for a much larger nesting population. Therefore, the current nesting female abundance is likely an

indicator of past and current threats, and given the intrinsic problems of small population size, elevates the extinction risk of this DPS.

### *Productivity*

The East Pacific DPS exhibits a decreasing nest trend since monitoring began, with a 97.4 percent decline since the 1980s or 1990s, depending on the nesting beach (Wallace *et al.* 2013). Despite intense conservation efforts, the decline in nesting had not been reversed as of 2011 (Benson *et al.* 2015). We found a declining nest trend at some of the remaining, small nesting aggregations. Abundance at Las Baulas, Costa Rica (previously the single largest nesting aggregation) at its peak was seven times the current abundance at Playa Barra de la Cruz/Playa Grande, Mexico (currently the largest nesting aggregation). From 1988/1989 to 2015/2016, the number of nesting females at Las Baulas declined  $-15.5$  percent annually (sd = 3.8 percent; 95 percent CI =  $-23.1$  to  $-7.8$  percent;  $f = 0.998$ ; mean annual nests = 315).

In recent decades (after a historical decline), nest counts have increased at some beaches in Mexico. The Playa Tierra Colorada nest trend has increased by 0.6 percent annually (sd = 8.9 percent; 95 percent CI =  $-17.1$  to 18.9 percent;  $f = 0.536$ ; mean annual nests = 153) between the 1996/1997 and 2016/2017 nesting seasons. Over the same time period, nesting at Playa Barra de la Cruz/Playa Grande increased by 9.5 percent annually (sd = 8.0 percent; 95 percent CI =  $-6.5$  to 25.8 percent;  $f = 0.918$ ; mean annual nests = 122). In contrast, nest counts at Cahuitán decreased from 1997/1998 through 2016/2017, with a median trend of  $-4.3$  percent annually (sd = 9.7 percent; 95 percent CI =  $-22.1$  to 17.6 percent;  $f = 0.716$ ; mean annual nests = 123).

We lack adequate data on nesting in Nicaragua to estimate trends.

Our trend analysis yields similar results to other published findings. The IUCN Red List assessment concluded that this subpopulation is decreasing and has declined by –97.4 percent over the past three generations (Wallace *et al.* 2013). The number of nests at Mexico nesting beaches has declined precipitously in recent decades (Benson *et al.* 2013). Historically, Mexico hosted the largest leatherback turtle nesting aggregation in the world, with 75,000 nesting females estimated during an aerial survey in 1980 ((Pritchard 1982). However, this estimate was derived from a brief aerial survey and may have been an overestimate (Pritchard 1996)). Prior to that aerial survey, Marquez *et al.* (1981) reported that the nesting beach of San Juan Chacahua (Oaxaca) was the most important nesting site in Mexico, with approximately 2,000 females nesting each season. Researchers also identified Tierra Colorada and Mexiquillo as important nesting sites, with approximately 3,000 to 5,000 nests per season. Monitoring of the nesting assemblage at Mexiquillo has been continuous since 1982. During the mid-1980s, more than 5,000 nests per season were documented along 4 km of this nesting beach. By 1993, less than 100 nests were counted along the entire 18 km beach (Sarti 2002). According to Sarti *et al.* (1996), nesting declined at this location at an annual rate of over 22 percent from 1984 to 1995. Researchers from the National University of Mexico recorded 3,000 to 5,000 nests annually from 1982 to 1989 at primary nesting beaches, with sharp declines observed in 1993 to 1994 at the nesting sites at Mexiquillo, Tierra Colorada, Chacahua and Barra de la Cruz. These early reports were generally snapshots (*e.g.*, local unpublished data) of leatherback nesting activity in Mexico, until 1995, when a more coordinated conservation effort took shape in the form of complete nesting surveys for the entire Pacific coast of Mexico (Eckert 1997). In 1995, “Proyecto Laud” (Leatherback

Project) was formed to estimate the population size using comprehensive surveys. In 1995 and 1996, Proyecto Laud estimated approximately 1,100 females nesting throughout Mexico; the next two seasons, they estimated between 236 and 250 nesting females, and declines continued. Currently, based on data from 2014 through 2018 (preliminary) between 100 and 250 females nest at all the protected beaches in Mexico.

In Costa Rica, the number of nesting females per season declined from 1,367 females in 1988 to 117 females in 1998 (Spotila 2000). While there were increases in the number of nesting females during the 1999/2000 season (224 females) and 2000/2001 season (397 females), the population has shown a steady decline, with less than 30 nesting females in recent years (*i.e.*, through 2016; The Leatherback Trust 2018).

In Nicaragua, 108 leatherback turtles nested on Playa Chacocente from October to December, 1980; in January 1981, 100 turtles nested in a single night on Playa El Mogote (Arauz 2002). An aerial survey of Playa El Mogote during the 1998/1999 nesting season revealed a nesting density of 0.72 turtles per kilometer (Sarti *et al.* 1999 *in* Arauz 2002). During the 2000/2001 nesting season, community members near Playa El Mogote reported that 210 leatherback nests had been deposited. That number decreased to 29 nests during the 2001/2002 nesting season (Arauz 2002). At Playa Veracruz 48 nesting females were identified between 2002 and 2010 (Urteaga *et al.* 2012). Between 2002 and 2014, Salazar *et al.* (2019) recorded 340 nests, indicating a downward trend. Considering the best available data, nesting has declined in Nicaragua.

Nesting females of the East Pacific DPS are generally smaller and produce fewer eggs per clutch than turtles from other leatherback populations (Sarti *et al.* 2007; Piedra *et al.* 2007; Santidrián Tomillo *et al.* 2007). For example in Mexico, nesting females have

a mean size of 144 cm CCL and 62 eggs per clutch; the average total fecundity per females was estimated to be 341 eggs per season, with a maximum of 744 eggs deposited in a season (Sarti *et al.* 2007). The low productivity parameters, drastic reductions in overall nesting female abundance, and current declines in nesting place the DPS at risk of extinction, especially given the limited nesting female abundance.

### *Spatial Distribution*

The DPS is characterized by somewhat continuous and low density nesting across long stretches of beaches along the coast of Mexico and Central America. Santidrián Tomillo *et al.* (2017) found a contraction of the Costa Rica's overall nesting distribution since the 1990s.

The best available genetic data indicate a high degree of connectivity among nesting aggregations. Dutton *et al.* (1999) did not find any genetic differentiation between nesting populations in Mexico (Playa Mexiquillo) and Costa Rica (Playa Grande) based on analysis of mtDNA control region sequences. Additional analyses of mtDNA sequences and nuclear DNA (microsatellites) from three index nesting beaches in Mexico also failed to find genetic differentiation (Barragan and Dutton 2000; Dutton *et al.* unpublished).

Based on monitoring of tagged nesting females, researchers documented female interchange between nesting beaches within Mexico and within Costa Rica. However, only one interchange has been documented between Mexico and Costa Rica (Sarti *et al.* 2007). Interchange between nesting beaches may occur during or between nesting seasons and may depend on the distance between nesting sites, which can be fairly large, especially in Mexico. For example, the distance between Tierra Colorada and Cahuitán is

25 kilometers, and up to 18.7 percent of nesting females visit both beaches within a season (average of nine percent). Mexiquillo is located approximately 475 kilometers from the closest other nesting beach (Tierra Colorada), and researchers found no interchange of females within seasons. However, a few females were found to nest in either Mexiquillo and/or Tierra Colorado between seasons (Sarti *et al.* 2007).

In Costa Rica, nesting females move among the three nesting beaches of Las Baulas National Park, within and between seasons, particularly between Playa Grande and Playa Langosta, although researchers study both Playa Grande and Playa Ventanas in combination. According to data gathered over 10 years of research (mid 1990s through the mid-2000s), an average of 71 percent of females nested only on Playa Grande, 10 percent nested only on Playa Langosta, and 18 percent nested on both beaches in a given season. In other seasons, females have been shown to shift and nest primarily on a different beach. Within two seasons, 82 percent of nesting females at Playa Langosta also nested at Playa Grande and 100 percent of nesting females at Playa Langosta within three seasons occasionally also nested at Playa Grande (Santidrián Tomillo *et al.* 2007). At the less abundant nesting beaches in Costa Rica, the exchange rate between females ranged between 7 and 28 percent. For example, at Ostional, 12 out of the 43 identified females were observed at least once at other sites (28 percent), while at Naranjo, 4 out of 21 identified females were also observed at other beaches (19 percent). At Cabuyal, 2 out of 15 turtles were observed at other beaches (13 percent), while 1 out of 15 females at Caletas were observed elsewhere (7 percent) (Santidrián Tomillo *et al.* 2017).

The foraging range of the DPS extends into coastal and pelagic waters of the southeastern Pacific Ocean. Individuals forage in the Pacific Gyre ecosystem and along

the coasts of Peru and Chile, with variation resulting from the location of upwelling and ENSO effects. Researchers have hypothesized that high bycatch along the coastal foraging phenotype in this population (Saba *et al.* 2007). Recently, Harrison *et al.* (2018) determined that post-nesting females from Las Baulas National Park spent 78.2 percent of their time on the high seas, 17.8 percent of their time in Costa Rica's EEZ, and 3.7 percent of their time around the Galapagos Islands.

Multiple nesting and foraging distributions likely help to buffer the DPS against local catastrophes or environmental changes that would otherwise modify nesting habitat or limit prey availability. Nesting aggregations are largely connected. However, there is less exchange among distant nesting beaches. Foraging turtles are vulnerable to perturbations in ocean conditions due to climate change, ENSO, and the Pacific Decadal Oscillation.

### *Diversity*

The East Pacific DPS exhibits genetic diversity, as demonstrated by moderate to high mtDNA haplotypic diversity ( $h = 0.66 - 0.71$ ; Dutton *et al.* 1999). Such diversity likely provides the DPS with some capacity for adapting to long-term environmental changes, such as cyclic or directional changes in ocean environments due to natural and human causes (McElhany *et al.* 2000; NMFS 2017). Nesting habitat is mainly restricted to mainland beaches along the same coast. The DPS does not exhibit temporal or seasonal nesting diversity, with most nesting occurring between October and March. This limits resilience. For example, short-term spatial and temporal changes in the environment are likely to affect all nesting females in a particular year. The foraging strategies are somewhat diverse, with turtles foraging in coastal and oceanic waters.

However, most turtles forage in the East Pacific Ocean, where they are similarly exposed to the effects of climate change, ENSO, or the Pacific Decadal Oscillation. Thus, the DPS has limited resilience.

*Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

The destruction or modification of habitat is a threat at many nesting beaches used by turtles of the East Pacific DPS. Foraging habitat has also been characterized as marginal, particularly in the eastern tropical Pacific Ocean (pelagic environment) due to relatively low productivity. Coastal habitat, which is normally associated with high productivity, may have been marginalized due to high levels of interactions with coastal artisanal fisheries.

Development threatens the DPS by modifying the preferred beach habitat for nesting. Sustained and substantial development along the northern and southern ends of the nesting beach at Playa Grande in Las Baulas National Park, and in adjacent areas, has resulted in the loss of nesting beach habitat in addition to the removal of much of the natural beach vegetation. As a result, erosion has increased and led to other environmental damages to sand that are associated with human development, including significant changes to elevation, water content, particle size, pH, salinity, organic content and calcium carbonate content (Clune and Paladino 2008). Within the past two decades, beachfront development in the town of Tamarindo (across Tamarindo Bay from Playa Grande) has resulted in the degradation of nesting beach habitat, including: pollution from artificial light, solid and chemical wastes, beach erosion, unsustainable water consumption, and deforestation. Hotels in this area have replaced a significant leatherback nesting area at Playa Tamarindo, which hosted significant nesting in the

1970s and 1980s (Wallace and Piedra 2012). Playa Langosta, which is just across from Tamarindo, is inundated with lights and noise from the town (Wallace and Piedra 2012). Currently, development has been curtailed due mainly to water issues (*i.e.*, drought). Any additional development would damage the current hydrology. The Leatherback Trust, a local nonprofit working at Las Baulas National Park, has acquired some properties to prevent development, but property costs have increased over time. At Las Baulas National Park, 10 percent of nests were being inundated by tidal flows. To mitigate this threat, nests at risk of tidal inundation were relocated to another site on the same beach or into a hatchery. Hatchling production slightly increased due to the establishment of the hatchery, where approximately two percent of hatchlings were produced from 1998 to 2004 (Santidrián Tomillo *et al.* 2007). We conclude that coastal development in Costa Rica is a threat to this DPS.

In Mexico, the extent of development near nesting beaches is generally low, given the remoteness of the beaches in Baja California and on the mainland. Reviewing the location of these nesting beaches, we found very few roads or development nearby. The main nesting beaches remain somewhat isolated, with very few roads or development adjacent to the nesting beaches. Thus, there is limited threat due to artificial lighting and generally little to no beach driving except perhaps that associated with monitoring efforts (L. Sarti, CONANP, pers. comm., 2018). In 2002, the Commission for Natural Protected Areas designated two of the index beaches (Mexiquillo and Tierra Colorada) as natural protected areas (turtle sanctuaries), which helped protect nesting habitat. Subsequently, in 2003, three of the index beaches (Mexiquillo, Tierra Colorada, and Cahuitán) were listed

as Ramsar Sites, which are wetland sites designated to be of international importance under the Ramsar Convention.

At Veracruz de Acayo beach in Nicaragua, Salazar *et al.* (2019) note that while conservation efforts has reduced the threat of poaching, the establishment of tourism-focused coastal development that do not comply with the existence of management plans could threaten the nesting habitat.

While nesting beaches within this DPS are generally remote and/or protected due to monitoring and existence of national parks and wildlife refuges, nesting females, hatchlings, and eggs at Las Baulas National Park (Costa Rica) nesting beaches are exposed to the modification of nesting habitat, as a result of development. This threat impacts the DPS by reducing nesting and hatching success, thus lowering the productivity of the DPS. We conclude that habitat loss and modification is a threat to the East Pacific DPS.

#### *Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

The harvest of nesting females and eggs was the primary cause of the historical decline in abundance of the East Pacific DPS. Since then, laws have been passed to protect eggs and turtles. However, poaching still occurs.

In Mexico, Sarti *et al.* (2007) attributed the decline of nesting females to the killing of adult females and intensive egg harvest. Adult females were historically killed at nesting beaches and in open waters (Sarti *et al.* 1994; Sarti *et al.* 1998). Since 1990, the harvest of turtles and eggs has been prohibited by national legislation. However, poaching pressure remains high wherever beach patrols do not occur (Santidrián Tomillo *et al.* 2017). For example, Mexiquillo produced hatchlings every season in the 1980s.

However, even with efforts to protect the nests in place, 60 to 70 percent of the total number of clutches were poached. Nichols (2003) notes that leatherback turtles were once harvested off Baja California, but their meat is now considered inferior for human consumption. At present, leatherback turtles are not generally captured for their meat or skin, but the poaching of nesting females has been known to occur on beaches such as Piedra de Tlacoyunque, Guerrero (Sarti *et al.* 2000).

Although poaching of turtles and eggs has been consistently reduced over the years, it still occurs at high levels. Effective conservation and protection depends on human presence at the nesting beaches (Santidrián Tomillo *et al.* 2017). Without such protection, poaching is likely to escalate. This may have occurred at one of the primary nesting beaches (Mexiquillo), where monitoring and conservation has not taken place in recent years due to safety concerns (L. Sarti, CONANP, pers. comm., 2018). Since the mid-1990s, Proyecto Laúd has been relocating clutches (usually within 1-2 hours of deposition) to protected fenced areas and releasing hatchlings in different areas of the beach. These efforts are intended to protect the eggs from poachers/predators and the hatchlings from predators (Sarti *et al.* 2007).

In Costa Rica, the population decline was predominantly caused by egg harvest. Ninety percent of eggs were collected on one of the major nesting beaches, Playa Grande, a decade or more prior to the reduction of nesting females (Santidrián Tomillo *et al.* 2007). In the 1950s, there were few nesting females at Playa Grande (Wallace and Piedra 2012). In the late 1960s and early 1970s, the number of nesting turtles increased to more than 100 nesting females nightly (Wallace and Piedra 2012). In the early 1970s, newly constructed roads provided access to people from distant villages and cities, and egg

harvest increased to more than 90 percent by the late 1970s (Wallace and Piedra 2012). Such high levels of egg harvest persisted for nearly two decades (Wallace and Saba 2009). Despite protection of nesting beaches at Las Baulas National Park, illegal poaching of eggs still occurs, though rarely. The black market for eggs remains strong; local bars throughout Guanacaste and elsewhere continue to offer shots of raw sea turtle egg yolks accompanying beer or liquor (Wallace and Piedra 2012).

In 1991, the Parque Nacional Marino Las Baulas was created and subsequently ratified by law in 1995. The Park consists of three leatherback nesting beaches: Playa Grande, Playa Ventanas, and Playa Langosta. The establishment of the park ensured increased protection at all three nesting beaches, greatly reducing egg poaching in the area. Poaching of eggs was reduced from 90 percent prior to 1990/1991, to 50 percent in 1990/1991, 25 percent in 1991 through 1993, and near 0 percent in 1993/1994 (Santridián Tomillo *et al.* 2007). To mitigate poaching, nests are often relocated. However, relocation may reduce hatching success (reviewed in Hernández *et al.* 2007; Eckert *et al.* 2012). In Playa Grande, Costa Rica, fewer females were produced in translocated nests; cooler nests due to a lower number of metabolizing embryos may have reduced hatchling success (Sieg *et al.* 2011).

In Nicaragua, prior to protection in the early 2000s, poachers took nearly 100 percent of the nests at the three nesting beaches. Nesting beach protection has occurred at Veracruz since 2002, Juan Venado since 2004, and Salamina since 2008. An average of ten community team members (mostly ex-poachers) monitor beaches seasonally. From 2002 to 2010, up to 420 nests were recorded and an estimated 94 were protected (Urteaga *et al.* 2012). While Veracruz de Acayo and Salamina are protected at 100 percent, Isla

Juan Venado is not permanently monitored. Therefore, poaching is likely to occur.

Poaching occurs at high levels at other beaches, such as Playa El Mogote. During the 2001/2002 nesting season, 23 of 29 nests were poached (79 percent), and the remaining six nests were protected in a hatchery (Arauz 2002). Due to the high level of poaching in this area, when possible, researchers from Flora & Fauna International relocated 98 nests between 2002 and 2004. However, these nests had a low emergence rate (22 percent; Urteaga and Chacón 2008).

Extensive and prolonged effects of comprehensive egg harvest have depleted the leatherback population in Costa Rica and Mexico, with egg harvest levels of nearly 90 percent for about two decades (Sarti *et al.* 2007; Santidrián Tomillo *et al.* 2008; Wallace and Saba 2009). Currently, nesting females and eggs of the East Pacific DPS are exposed to poaching. Though efforts have reduced the levels of poaching of both eggs and nesting turtles, egg poaching remains high and affects a large proportion of the DPS. Poaching of nesting females reduces both abundance (through loss of nesting females) and productivity (through loss of reproductive potential). Such impacts are high because they directly remove the most productive individuals from DPS, reducing current and/or future reproductive potential. Egg harvest reduces productivity only, but over a long period of time, this also reduces recruitment and thus abundance. Given the high exposure and impacts, we conclude that overutilization, as a result of poaching, poses a major threat to the DPS.

#### *Disease or Predation*

Little is known about diseases and parasites in leatherback turtles, although fibropapillomatosis has been described as a major epizootic disease in hard shelled

turtles. A fibropapilloma tumor (in regression) was found on one nesting female at Mexiquillo, Mexico in 1997 (Huerta *et al.* 2002). Various bacteria have also been documented in leatherback eggs. Soslau *et al.* (2011) sampled eggs laid on a Costa Rican beach to determine if bacteria were contributing to the low hatching rate (50 percent). The bacteria identified (*i.e.*, species of the *Bacillus*, *Pseudomonas*, and *Aeromonas* genera) are known pathogens to humans and may account for developmental arrest of the turtle embryo (Soslau *et al.* 2011).

Numerous predators prey on East Pacific leatherback turtles throughout their life stages. Eggs and hatchlings are eaten by crabs, ants, birds, reptiles, mammals, and fish (Eckert *et al.* 2012). In Costa Rica, during the 1993/1994 nesting season, several nests were lost to predation and infestation by maggots (Schwandt *et al.* 1996). In the Nicoya Peninsula, on the Pacific coast of Costa Rica, Squires (1999) documented evidence of potential nest predation by dogs, coyote, and raccoon. Predation of hatchlings by dogs and raccoons has increased in Playa Grande due to an increase in development in the area (P. Santridián Tomillo, The Leatherback Trust, pers. comm., 2019).

For adult turtles, principal predators at sea include killer whales, crocodiles (Pritchard 1981), and sharks, while nesting females are taken by crocodiles (Bedding and Lockhart 1989), tigers, and jaguars (Pritchard 1971). Sarti *et al.* (1994) observed a lone male killer whale feeding on a single gravid female near Michoacán, Mexico, apparently consuming only certain parts of the turtle and discarding others (*e.g.*, female reproductive organs). In summary, eggs, hatchlings, and some adults are exposed to predation. For this DPS, the primary impact is to productivity (*i.e.*, reduced egg and hatching success). Predation on nesting females, while rare, reduces abundance and productivity. Nest

predation is mitigated through screening of nests, relocation of nests to hatcheries and releasing hatchlings in safer areas of the beach, and protecting nesting females from large predators such as dogs and jaguars (Sarti *et al.* 2007); some of these efforts are funded through the MTCA. We conclude that predation is a threat to the East Pacific DPS.

### *Inadequacy of Existing Regulatory Mechanisms*

Several international regulatory mechanisms apply to turtles in this DPS. The IAC, in particular, prohibits the harvest of turtles and eggs. CITES limits all international trade of the species. There are also international efforts to reduce fisheries bycatch.

In 2015, at the 7<sup>th</sup> Conference of the Parties, the IAC resolved to prioritize conservation actions in their work programs that would help “reverse the critical situation of the leatherback sea turtle in the Eastern Pacific.” Specifically, parties were urged to: 1) submit leatherback bycatch information annually to the IAC Secretariat; 2) improve leatherback turtle fishery monitoring efforts through the use of on-board observers; 3) report annually on the measures they have taken to reduce leatherback bycatch in their fisheries; 4) enhance leatherback nest monitoring and protection to increase hatchling survival and protect nesting beach habitat; 5) foster safe handling and release of incidentally bycaught leatherback turtles in fisheries; and 6) agree to a five-year strategic plan containing key activities related to the resolution (CIT-COP7-2015-R2). The strategic plan was patterned after the Regional Action Plan for Reversing the Decline of the Eastern Pacific Leatherback (<http://savepacificleatherback turtles.org>) and included measures to reduce fisheries bycatch of adult and subadult leatherback turtles, the identification of high risk areas with fisheries and leatherback turtles, the identification and protection of important areas for leatherback turtle survival in different life stages,

the elimination of any consumption and illegal use of leatherback turtles, and nesting site protection.

As mandated by the 1994 *North American Agreement for Environmental Cooperation*, the Commission for Environmental Cooperation (CEC) encourages Canada, the United States, and Mexico to adopt a continental approach to the conservation of flora and fauna. In 2003, this mandate was strengthened as the three North American nations launched the *Strategic Plan for North American Cooperation in the Conservation of Biodiversity*. The North American Conservation Action Plan (NACAP) initiative began as an effort promoted by the three nations, through the CEC, to facilitate the conservation of marine and terrestrial species of common concern. In 2005, the CEC supported the development of a NACAP for Pacific leatherback turtles by Canada, the United States, and Mexico. Identified actions in the plan addressed three main objectives: 1) protection and management of nesting beaches and females; 2) reducing mortalities from bycatch throughout the Pacific Basin; and 3) waste management, control of pollution, and disposal of debris at sea.

In 2015, the Eastern Pacific Leatherback Network (also known as La Red de la Tortuga Laúd del Océano Pacífico (Red Laúd OPO) ([www.savepacificleatherbacks.org](http://www.savepacificleatherbacks.org))) was formed to address the critical need for regional coordination of East Pacific leatherback conservation actions to track conservation priorities and progress at the population level. This network has brought together conservationists, researchers, practitioners and government representatives from 22 institutions across nine East Pacific nations with varying priorities, capacities and historical experiences in leatherback research and conservation to contribute to shared activities, projects, and goals. Through

these efforts, Red Laúd OPO now has mutually-agreed upon mechanisms for sharing information and data, as well as standardized protocols for nesting beach monitoring and bycatch assessments/fishing practices.

The Convention for the Protection of Natural Resources and Environment of the South Pacific, also known as the Noumea Convention, has been in force since 1990 and includes 26 Parties (as of 2013). The purpose of the Convention is to protect the marine environment and coastal zones of the South-East Pacific, and beyond that area, the high seas up to a distance within which pollution of the high seas may affect that area.

In 2015, the IATTC passed a resolution that requires large longline vessels fishing in the eastern tropical Pacific Ocean to carry observers. Cooperating parties that have documented interactions with sea turtles in their longline fleet are required to maintain at least five percent observer coverage and provide an annual report to the IATTC. Unfortunately, the forms used by observers to report incidents are not standardized, so in some cases, the reports did not include species identification, condition of the released turtles, and location of the interactions, and the five percent minimum coverage is often not met. Nations without reported bycatch of sea turtles simply provided a statement to that effect. In the few reports we reviewed, leatherback turtles comprised some of the bycatch in the eastern tropical Pacific Ocean, but there were few details on the events (C. Fahy, NMFS, pers. comm., 2018). In 2007, the IATTC passed a resolution requiring nations to conduct research on sea turtle bycatch reduction measures in their longline fleets (*e.g.*, use of circle hooks and fish bait). Despite results in both the Atlantic and Pacific longline fleets showing that use of circle hooks/fish bait significantly reduced leatherback bycatch rates (Swimmer *et al.* 2017), nations are not required to use this

hook/bait combination. In 2017, at an IATTC sea turtle bycatch reduction workshop, the United States presented findings on longline bycatch reduction and proposed a stronger resolution that would require use of this methodology. However, some nations resisted, and the resolution did not move forward for consideration at the annual IATTC meeting.

Throughout the world, illegal, unreported, and unregulated (IUU) fishing leads to underestimates of bycatch. In Mexico, there is a lack of effective fisheries governance, resulting in highly uncertain fishery statistics. For example, from 1950 to 2010, total fisheries catch, including estimated IUU catch and discarded bycatch, was nearly twice as high as the official statistics (Cisneros-Montemayor *et al.* 2013). Thus, the bycatch threat of commercial fisheries in Mexico may be higher than currently estimated.

In addition, several international treaties and/or regulatory mechanisms protect East Pacific leatherback turtles. While no single law or treaty can be 100 percent effective at minimizing anthropogenic impacts to sea turtles in these areas, there are several international conservation agreements and laws in the region that, when taken together, provide a framework within which sea turtle conservation advances can be made (Frazier 2012). In addition to protection provided by local marine reserves throughout the region, sea turtles may benefit from the following broader regional effort: (1) the Eastern Tropical Pacific (ETP) Marine Corridor (CMAR) Initiative supported by the governments of Costa Rica, Panama, Colombia, and Ecuador, which is a voluntary agreement to work towards sustainable use and conservation of marine resources in these nations' waters; (2) the ETP Seascape Program managed by Conservation International that supports cooperative marine management in the ETP, including implementation of the CMAR; (3) the IATTC and its bycatch reduction efforts through resolutions on sea

turtles, observer coverage, *etc.*; (4) the IAC, which is designed to lessen impacts on sea turtles from fisheries and other human impacts; and (5) the Permanent Commission of the South Pacific (Lima Convention), which has developed an *Action Plan for Sea Turtles in the Southeast Pacific*.

Most nations within the range of the East Pacific DPS have laws prohibiting the harvest of turtles and eggs. This applies to nesting turtles and those captured at sea. National laws in Mexico (1990 Presidential Decree), Costa Rica (2002 Presidential Decree N<sup>o</sup>8325: the Law of Protection, Conservation, and Recuperation of Marine Turtles), and Nicaragua (Law No. 651 and Ministerial Resolution No. 043-2005) protect nesting females and eggs and nesting beaches. However, poaching remains a major threat. Although laws prohibit the harvest of turtles in Peru, fishermen consume leatherback turtles bycaught in small-scale fisheries (Alfaro-Shigueto *et al.* 2011), indicating inadequate enforcement of existing laws. In other nations where leatherback turtles of this DPS are bycaught, the turtles are released and not retained (*e.g.*, Chile; Donoso and Dutton 2010).

Several protected areas have been established throughout the range of the DPS. Most of the nesting beaches in Mexico and Costa Rica are protected from egg and turtle poaching, with effective monitoring to ensure low levels of poaching. Poaching likely continues at unprotected and remote beaches, and at those that contain an extensive coastline that is difficult to monitor and protect. Protected nesting beaches in Mexico include: Mexiquillo (until 2013); Playa de Tierra Colorada, Playa Cahuitán, Playa San Juan, Bahía de Chacahua, and Playa Barra de la Cruz. Protected nesting beaches in Costa Rica include: Las Baulas National Park (Playa Grande, Playa Langosta, and Playa

Ventanas), Naranjo (National Park), Cabuyal (under no official management category), Nombre De Jesús (under no official management category), Ostional (wildlife refuge), and Caletas (wildlife refuge). Protected nesting beaches in Nicaragua include: Salamina-Costa Grande, Veracruz de Acayo (Chacocente Wildlife Refuge).

Marine protected areas also exist. The waters of the Las Baulas National Park, which represents a hotspot for inter-nesting females and breeding males, are protected out to 22.2 km as a no-take zone for all fishing activity. However, satellite telemetry data for nesting females at these beaches over three seasons revealed that the turtles move well outside these boundaries during their inter-nesting period, which makes them vulnerable to fisheries outside the park (Shillinger *et al.* 2010). Data from 44 females that were tagged off Las Baulas National Park revealed a high use habitat within 6 nm from the nesting beaches, but overall revealed a generally large range, covering over 33,000 km<sup>2</sup>, from the Nicoya Peninsula, east into the Gulf of Nicoya in Costa Rica, and north to coastal habitats within 30 kilometers offshore from southern Nicaragua. The marine areas adjacent to this protected boundary are not managed under any type of status (Shillinger *et al.* 2010). Fisheries within Costa Rica and Nicaragua's EEZ include trawl, gillnet and longline that continue to operate.

In summary, numerous regulatory mechanisms exist to protect leatherback turtles, eggs, and nesting habitat throughout the range of this DPS. Although the regulatory mechanisms provide some protection to the species, many do not adequately reduce the threat that they were designed to address, generally as a result of limited implementation or enforcement. As a result, bycatch, incomplete nesting habitat protection, and poaching

remain threats to the DPS. We conclude that the inadequacy of existing regulatory mechanisms is a threat to the East Pacific DPS

### *Fisheries Bycatch*

Bycatch in commercial and recreational fisheries, both on the high seas and off the coasts, is the primary threat to the East Pacific DPS. This threat affects the DPS by reducing the abundance of all life stages of the DPS (with the likely exception of hatchlings).

Integrating catch data from over 40 nations and bycatch data from 13 international observer programs, Lewison *et al.* (2004) estimated the numbers of leatherback turtles taken globally by pelagic longliners to be more than 50,000 leatherback turtles in just one year (2000). With over half of the total fishing effort (targeting tuna and swordfish) occurring in the Pacific Ocean, an estimated 20,000 to 40,000 leatherback turtles interacted with longline fishing during the year studied. Fishing effort was highest in the central South Pacific Ocean (south of Hawaii), which overlaps with the foraging range of this DPS. Because observers are in place on only a fraction of longline vessels in the eastern tropical Pacific Ocean, and a requirement came into effect only recently through an IATTC resolution, these estimates are likely a minimum. More recently, Molony (2005) and Beverly and Chapman (2007) estimated sea turtle longline bycatch to be approximately 20 percent of that estimated by Lewison *et al.* (2004), or approximately 200 to 640 leatherback turtles annually. Where tuna species are targeted, bycatch of turtles in the deep-set longline gear often results in mortality due to drowning. Additional studies indicate the high impact of industrial longline fleets on leatherback turtles (*e.g.*, Spotila *et al.* 1996, 2000).

In their global study of sea turtle bycatch, where available, Wallace *et al.* (2013) found that longline bycatch had a low impact, but that net bycatch had a high impact on the East Pacific RMU. The impact of local artisanal fleets (using gillnets and longlines) that fish closer to shore is less documented.

In Mexico, leatherback turtles wash to shore entangled in longlines and driftnet, indicating interaction and mortality (Sarti *et al.* 2007). Ortiz-Alvarez *et al.* (2019) conducted a bycatch survey across 48 different ports (933 fishers) in Mexico, Nicaragua and Costa Rica between October 2016 and July 2017 in an effort to improve the understanding of leatherback bycatch in artisanal fisheries, particularly where data are lacking. The surveys represented on average over 30 percent of the fishing fleet per port for both Nicaragua and Costa Rica and 6 percent per port for Mexico. In Mexico, where gillnets were the most frequently reported gear, fishers (n = 709) reported an estimated bycatch of 300 leatherback turtles in the previous year, with 65 percent in “good condition;” 76 percent of fishers released turtles alive (three percent consumed or sold the turtles). Estimated average bycatch rates per vessel were 1.0 for Costa Rica and Nicaragua and 2.3 for Mexico. In Costa Rica, leatherback turtles were primarily caught in longlines and released alive; 75 percent of the Costa Rican fishermen reported that bycaught leatherback turtles were in "good condition." In Nicaragua, where gillnets were the most frequently reported gear, 18 percent of fishers reported that leatherback turtles were in “good condition;” 76 percent of fishers released turtles alive (six percent consumed or sold the turtles; Ortiz-Alvarez *et al.* (2019).

Recent surveys of 765 Ecuadorian, Peruvian, and Chilean fishermen (at 43 ports, representing 28 to 63 percent of ports) reported the following leatherback interaction

rates (as a percentage of total interactions with sea turtles): 2.81 percent of 40,480 interactions (32.5 percent mortality) in Ecuador, 14.87 of 5,828 interactions (50.8 percent mortality) in Peru, and 27.83 percent of 170 interactions (3.2 percent mortality) in Chile (Alfaro-Shigueto *et al.* 2018). Mortality rates reported for all sea turtles were 3.2 percent in Chile, 32.5 percent in Ecuador, and 50.8 percent in Peru (Alfaro-Shigueto *et al.* 2018).

The swordfish gillnet fisheries in Peru and Chile may have contributed to the decline of the DPS. The decline in the nesting population at Mexiquillo occurred at the same time that effort doubled in the Chilean driftnet fishery (Eckert 1997). Using data collected from Frazier and Montero (1990) regarding leatherback takes in a swordfish gillnet fishery from one port in Chile (San Antonio), and extrapolating to other ports in Chile and Peru, with an increased level of effort observed through the mid-1990s, Eckert (2007) estimated that a minimum of 2,000 leatherback turtles were killed annually by the combined swordfish fishing operations (only gillnet) off Peru and Chile. After some fleets switched from large mesh gillnet to longline to target swordfish, this estimate has declined by at least an order in magnitude. Research conducted in the Chilean large-mesh gillnet fishery to reduce bycatch of marine mammals and sea turtles indicates that less than five leatherback turtles have interacted with the fishery (on observed vessels) since 2014, and all were released alive (C. Fahy, NMFS, pers. comm., 2018).

In Peru, the capture of leatherback turtles has been prohibited since 1976, although retention of bycaught leatherback turtles continues (FAO 2004). From 1985 to 1999, based on field books, diaries, specimen data sheets, fishery statistics files and unpublished reports, 30 leatherback turtles were captured in fisheries (*in* Alfaro-Shigueto *et al.* 2007). From July 2000 to November 2003, observers at 8 ports, from Mancora in

northern Peru to Morro Sama in the south, reported 133 leatherback turtles caught by artisanal fishing gear, with 76 percent caught in gillnets and 24 percent caught in longlines targeting fish, sharks, and rays (Alfaro-Shigueto *et al.* 2007). Of the total caught, 41.4 percent (n = 55) were released alive and 58.6 percent (n = 78) were retained for human consumption. Of the leatherback turtles retained and measured (n = 6), the size ranged from 98 to 123 cm curved carapace length (CCL), indicating that both subadults and adults are encountered by artisanal fisheries off Peru. Researchers recently assessed and quantified sea turtle mortality levels in one fishing village in central-southern Peru (San Andrés) through sampling dump sites (97.3 percent) and strandings (2.7 percent) over a 5-year period (2009 to 2014). Of 953 carapaces recorded, leatherbacks comprised only 1.4 percent of sea turtles (n = 13). However, this study still confirmed that they were consumed or sold for human consumption. With a mean CCL of 113.0 cm (range: 80 to 135, n = 10), 70 percent of the leatherbacks were juveniles and 30 percent were subadults. There were no adults. Researchers noted that the meat was used to support separate demands: fishermen families' consumption, local trade, and "special" orders from Lima (Quispe *et al.* 2019). Using data from shore-based and on-board observers, Alfaro-Shigueto *et al.* (2011) estimated the mean annual leatherback bycatch as follows: 40 turtles (with a range of 37 to 44) in the driftnet fishery, with 80 percent released alive; six turtles (with a range of 3 to 9) in the dolphinfish longline fishery, all released alive; and 26 turtles (with a range of 24 to 27) in the shark longline fishery, all released alive. Alfaro-Shigueto *et al.* (2015) assessed the bycatch of leatherback turtles in driftnet vessels in northern Peru (through at-sea monitoring) and central Peru (shore-based monitoring). From December 2013 to November 2014, 31 leatherback turtles were

captured, of which 13 died. Interactions occurred primarily with juveniles and subadults (mean CCL was  $125.1 \pm 14.8$ ). Nearshore driftnets from San Jose (northern Peru) captured 20 leatherback turtles (five dead). At least one animal was butchered, indicating that even animals caught alive may be killed, despite Peruvian laws restricting such practices. Approximately 3,000 net vessels fish along the coast of Peru, but only a fraction were included in this study (Alfaro-Shigueto *et al.* 2015). Efforts are being made to patrol nets to reduce bycatch, conduct extensive education and outreach, and increase regulation and enforcement (Alfaro-Shigueto *et al.* 2015). A review of information collected from official statistics, literature, and surveys of beaches and dumpsites revealed that the size of captured leatherback turtles declined over the years. In 1987, the mean CCL of captured leatherback turtles was  $117 \pm 10.65$  cm, while in 2005, the mean CCL was  $109.27 \pm 14.4$ , possibly indicating overexploitation due to systematic and sustained harvests, particularly during El Niño years (Campos *et al.* 2009). Greater captures of all sea turtles, including leatherback turtles, occurred during periods of El Niño, when turtles are more likely to be found in more coastal waters (where there is increased artisanal fishery activity) due to environmental variability and availability of jellyfish in those areas (Campos *et al.* 2009).

In Chile, a commercial fishery was established in 2001 that permitted longlining for swordfish (shallow-set) with the condition that all vessels were required to take an observer on board to collect information on bycatch. Between 2001 and 2005, over 10 million hooks were observed, and leatherback turtles were the most common species caught ( $n = 284$ ), with the majority ( $n = 282$ ) released alive. Leatherback turtles were caught primarily between  $24^\circ$  S and  $38^\circ$  S (furthest south was  $38^\circ 39'$  S and  $84^\circ 15'$  W) in

less than 4 percent of the sets with an overall mean of 0.0268 turtles per one thousand hooks. Size estimates revealed both juveniles and adults. Fishermen were trained to use the best practices for de-hooking, disentangling, and releasing sea turtles, which likely increased the survival rate of leatherback turtles (Donoso and Dutton 2010). Researchers recently presented information on the incidental capture of sea turtles in industrial and artisanal longlines, gillnets and artisanal espinel (*i.e.*, small-scale handline or longline) fisheries all targeting swordfish off Chile (Zárate *et al.* 2019). Over an 8-year period (2006–2014), 182 leatherbacks were documented as bycatch (mortality of bycaught turtles was not reported). Over this study period, 44 percent of turtles were caught in industrial longline, 28 percent in artisanal espinel, 17 percent in gillnets and 11 percent in artisanal longline (with sea turtle species undefined). Researchers noted that while observer coverage in the industrial longline fleet has been generally high (> 70 percent of total fishing trips), the monitoring coverage of artisanal espinel and gillnets is very low (< 3 percent). Thus, these estimates of bycatch can be considered minimal. While the number of industrial and artisanal vessels has declined (from 12 vessels in 2001 to 3 vessels in 2014, the number of artisanal espinel and gillnet vessels has not declined, remaining around 90 vessels (Zárate *et al.* 2019).

We conclude that juvenile and adult life stages of the East Pacific DPS are exposed to high fishing effort throughout their foraging range and in coastal waters near nesting beaches. Mortality is also high in some fisheries, with reported mortality rates of up to 58 percent due in part to the use of gillnets and as well as consumption of bycaught turtles in Peru. As noted above, there have been efforts by individual nations and regional fishery management organizations to mitigate and reduce the threat of bycatch, but those

efforts have not been successful at ameliorating the risks. We conclude that fisheries bycatch remains a major threat to the East Pacific DPS.

### *Pollution*

Pollution is a threat to the East Pacific DPS. Pollution includes contaminants, marine debris, and ghost fishing gear. The South Pacific Garbage Patch, discovered in 2011 and confirmed in 2017, contains an area of elevated levels of marine debris and plastic particle pollution, most of which is concentrated within the ocean's pelagic zone and in area where leatherback turtles forage for many years of their life. The area is located within the South Pacific Gyre, which spans from waters east of Australia to the South American continent and as far north as the Equator.

Given the amount of floating debris in the Pacific Ocean (Lebreton *et al.* 2018), marine debris has the potential to be a significant threat to the East Pacific leatherback population. The precise impact cannot be quantified using the best available data. Leatherback turtles subsist primarily on jellyfish and other gelatinous zooplankton and may be prone to ingesting plastics resembling their food source (Mrosovsky 1981; Schuler *et al.* 2013, 2015). Dead leatherback turtles have been found choked on plastic bags, and phthalates derived from plastics have been found in leatherback egg yolk (Lebreton *et al.* 2018).

Prior to the early 1990s, high seas driftnet fisheries freely operated in the Pacific Ocean and interacted with thousands of sea turtles. Researchers estimated that over 1,000 leatherback turtles were taken by the combined fleets of Japan, Korea, and Taiwan during a one-year period (Wetherall 1997). However, because genetic analyses of Pacific leatherback turtles were relatively new at that time, the data does not indicate the nesting

beach origin of those bycaught leatherback turtles. In 1992, a UN moratorium banned high seas driftnet fisheries, so that active large scale driftnets no longer pose a threat to leatherback turtles. However, numerous discarded driftnets continue to entangle and drown leatherback turtles in a phenomenon known as “ghost fishing” (Gilman *et al.* 2016),

In 2007, the IATTC passed a resolution pertaining to sea turtle bycatch in purse seine and longline fisheries which primarily target tuna. In order to address the marine debris and potential interactions with sea turtles in the eastern tropical Pacific Ocean, fishermen are required to disentangle sea turtles entangled in fish aggregating devices, even if the device does not belong to the vessel.

Only a few studies of levels or effects of toxins on leatherback turtles have examined effects to their health and fitness, as well as any effects to eggs and hatchlings. Sill *et al.* (2008) sampled non-viable leatherback eggs and hatchlings that died in the egg chamber at Las Baulas National Park. Researchers analyzed the samples for metals and other toxicants to explore the relationship between pollution and hatching success for 30 females. Metal levels were highly variable, but there were no significant differences within and between groups of females, and none of the pesticides tested were present in the samples (Sill *et al.* 2008). Overall, the study found no relationship between metal concentrations and hatching success. The researchers postulated that eggs may take up some metals from the nest environment and deposit other metals in the egg shell, as unhatched eggs contained more nickel, copper, and cadmium and contained significantly less iron, manganese and zinc than dead hatchlings (Sill and Paladino 2008).

As with all leatherback turtles, entanglement in and ingestion of marine debris and plastics is a threat that likely kills several individuals a year. However, data are not available because most affected turtles are not observed. Given the amount of pollution turtles are exposed to throughout their lifetime, this has the potential to be a significant threat to the East Pacific leatherback population, although the impact cannot be quantified using the best available data. We conclude that pollution is a threat to this DPS.

#### *Oceanographic Regime Shifts*

The East Pacific DPS is affected by oceanographic regime shifts. In the eastern equatorial Pacific Ocean, reductions in productivity parameters are primarily associated with ENSO, during which sex ratios become biased up to 100 percent female (Santidrián Tomillo *et al.* 2014). There is also an effect on hatching and emergence success in North Pacific Costa Rica (Santidrián Tomillo *et al.* 2012): during El Niño years, hatching success is very low due to dry and hot conditions on the nesting beaches and is high during la Niña events due to increased precipitation in this area. La Niña events are characterized by high phytoplankton productivity, cooler sea surface temperatures, enhanced precipitation in northwestern Costa Rica, and cooler air temperatures. These factors lead to increases in the biomass and distribution of gelatinous zooplankton, the primary food of leatherback turtles. Foraging success and the frequency of reproduction are enhanced following such periods of high primary productivity (Saba *et al.* 2007). Nesting seasons that follow the La Niña events, result in peaks in the number of nesting females, higher than average hatching success and emergence rates, and a larger proportion of male hatchlings (Saba *et al.* 2012). Saba *et al.* (2008) found that a shift

from 1°C to -1°C in the El Niño sea surface temperature anomaly resulted in a five-fold increase in leatherback remigration probabilities at Playa Grande. Such large-scale regime shifts are likely to affect the entire DPS. Productivity is positively (La Niña) or negatively (El Niño) impacted. Wallace *et al* (2006) hypothesize that prey availability related to ENSO exacerbates the effects of fisheries bycatch mortality, resulting in declining trends. Because of the small abundance of the DPS, extended El Niño events are likely to pose a threat to the East Pacific DPS.

### *Climate Change*

Climate change is a threat to the East Pacific DPS. The impacts of climate change include: increases in temperatures (air, sand, and sea surface); sea level rise; increased coastal erosion; more frequent and intense storm events; and changes in oceanographic regimes and currents.

Climate projections assessed by the IPCC indicate that Central America is very likely (defined as 90 to 99 percent probability; IPCC 2007) to become warmer and likely (defined as 66 to 90 percent probability; IPCC 2007) to become drier by 2100 (Saba *et al.* 2012). In addition, climate variability is likely to change the strength and frequency of El Niño events, although there is less scientific consensus on the frequency and magnitude of changes to these events. A climate-forced population dynamics model developed by Saba *et al.* (2012) showed sea surface temperatures to be highly correlated with large phytoplankton productivity throughout a 100-year projection to the year 2100. Relative to a stable nesting population given mean surface air temperatures and precipitation from 1975 to 1999, Saba *et al.* (2012) estimated that the nesting population at Playa Grande would decline at a rate of 7 ( $\pm 1$ ) percent per decade over the next century of climate

change under a scenario which considered increasing emissions from 2000 to 2100 (A2 scenario). Similar declines occurred for other scenarios (Special Report on Emissions Scenarios 2007). The nesting population was projected to remain stable up until around 2030 but reduced 75 percent by the year 2100. Hatching success and emergence rates, which would decrease associated with 2.5 °C warming of the nesting beaches, served as a primary driver of the decline. Santidrián Tomillo *et al.* (2012) developed a similar climate forcing model, which considered projected changes associated with El Niño events and demonstrated that hatching success would decline from approximately 42 to 18 percent by 2100, while emergence rates would decline between approximately 76 to 29 percent. The authors concluded that even with protection at the primary nesting beaches in Costa Rica, with the general warming of Central America in the near future, the chances of a new nesting area emerging with more ideal conditions (*i.e.*, cooler and wetter) is unlikely (Santidrián Tomillo *et al.* 2012).

Increasing sand temperature is an existing threat to the DPS. The long-term data set on leatherback turtles nesting at Playa Grande, Costa Rica indicates reduced emergence success, skewed sex ratios, and increased hatchling mortality as a result of increased sand temperature (Santidrián Tomillo *et al.* 2015). From 2004 to 2013, primary sex ratios fluctuated between a minimum sex ratio of 41 percent females (and the only year with a male-biased hatchling production) to 100 percent females produced during two seasons (Santidrián Tomillo *et al.* 2014). Low emergence success and low hatchling output (*i.e.*, higher mortality as a result of high sand temperatures) were associated with a strongly biased female ratio, because these resulted from female-producing high temperatures. Variability in these results occur during and between nesting seasons,

largely due to highly variable climatic conditions in northwestern Costa Rica, resulting in “boom-bust” cycles in leatherback hatchling production and primary sex ratios (*in Santidrián Tomillo et al. 2014*). Sand temperatures are projected to continue to increase, which will likely result in a further decline in the number of hatchlings produced (*Santidrián Tomillo et al. 2014*). An increase in the percentage of females could potentially benefit the productivity of the DPS in the short-term. However, any such benefits would be tempered by the associated lower emergence and hatchling success rates. Relocation of sea turtle clutches that may be “doomed” due to high sand temperatures and inundation is a common conservation practice, particularly at areas with warming beaches. However, relocation is not always possible and is also associated with lower emergence and hatchling success rates.

In addition to climate change influencing the nesting beach habitat of eastern Pacific leatherback turtles, the impacts of a warming ocean may also affect the environmental variables of their pelagic migratory and foraging habitat, which may further increase population declines. As mentioned previously, the preferred foraging habitat of eastern Pacific is characterized by relatively low sea surface temperatures and low levels of chlorophyll-a. Using information derived from satellite tracked leatherback turtles, which established migratory pathways and core foraging habitat (as summarized in *Shillinger et al. 2008*), in combination with generalized additive mixed models, researchers were able to project that between 2001 and 2100, there would be a net loss of the core foraging habitat of the DPS. The loss was predicted to be a 15 percent decline over the next century (*Willis-Norton et al. 2014*). Depending on whether this population is able to shift their preferred migratory routes and foraging habitat over time (which is

unclear), remigration intervals may shorten or lengthen, which could influence reproductive productivity.

Climate change is a threat to the East Pacific DPS that affects nesting females (*e.g.*, remigration interval and fitness), their progeny (*e.g.*, hatching success, embryonic development, and feminization of hatchlings), and foraging subadult and adult leatherback turtles. Detrimental impacts of increased sand temperatures have already occurred and are likely to continue or worsen. Foraging areas will also be impacted via changes in ocean productivity, sea surface temperatures, and availability of prey.

### *Conservation Efforts*

There are numerous efforts to conserve the leatherback turtle. The following conservation efforts apply to turtles of the East Pacific DPS (for a description of each effort, please see the section on conservation efforts for the overall species): Convention on the Conservation of Migratory Species of Wild Animals, Convention on Biological Diversity, Convention on International Trade in Endangered Species of Wild Fauna and Flora, Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention), Convention for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPF Convention), Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention), Eastern Pacific Leatherback Network, Eastern Tropical Pacific Marine Corridor Initiative, FAO Technical Consultation on Sea Turtle-Fishery Interactions, IAC, MARPOL, IUCN, Ramsar Convention on Wetlands, RFMOs, Secretariat of the Pacific Regional Environment Programme, UNCLOS, and UN Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing. Although numerous

conservation efforts apply to the turtles of this DPS, they do not adequately reduce its risk of extinction.

### *Extinction Risk Analysis*

After reviewing the best available information, the Team concluded that the East Pacific DPS is at high risk of extinction. The DPS exhibits a total index of nesting female abundance of 755 females at monitored beaches. Such a limited nesting population size makes this DPS vulnerable to stochastic or catastrophic events that increase its extinction risk. This DPS exhibits a decreasing nest trend, which along with lower than-average productivity metrics, has the potential to further reduce abundance and increase the risk of extinction. The nesting range is somewhat limited to the Pacific Central American coast, with little diversity among sites. Thus, stochastic events could have catastrophic effects on nesting for the entire DPS, with no distant subpopulations to buffer losses or provide additional diversity. Most foraging occurs in the eastern Pacific Ocean, which is subject to oceanographic regimes shifts that expose the DPS to low-productivity events. Based on these demographic factors, we find the DPS to be at risk of extinction as a result of past threats.

Current threats also contribute to the risk of extinction of this DPS. Fisheries bycatch is the major threat, capturing, and often killing, turtles throughout their foraging areas, thus reducing abundance. There are few mechanisms in place, including internationally through the IATTC or other bilateral or international instruments and through monitoring and enforcement of coastal fisheries laws, to mitigate or reduce bycatch. Overutilization is also a major threat. Historically, harvest of turtles and eggs reduced the once high abundance of turtles to current low levels. The poaching of eggs

continues, reducing productivity, especially at unprotected beaches, where egg collection may reach 100 percent and nesting females may also be at risk of poaching. The effects of climate change, including the observed and predicted increase in frequency and strength of ENSO events (*i.e.*, oceanographic regime shifts), are threats to this DPS, given its restricted foraging range and the vulnerability of nesting beaches to high sand temperatures and low levels of rainfall, which affect sex ratios and emergence and hatching success (*i.e.*, productivity). Additional threats include: habitat loss and modification; predation; and pollution. Development modifies nesting habitat. However, most beaches are protected throughout the nesting range. Though many regulatory mechanisms are in place, they do not adequately reduce the impact of these threats. Further, it is important to note that efforts (*e.g.*, relocation) to protect and mitigate threats from the harvest of turtles and eggs, predation, and environmental impacts related to erosion and lethal temperatures are dependent upon the presence of monitoring or management programs. Some of these are dependent on funding from the MTCA. Even when undertaken, these efforts may not be successful.

We determine, consistent with the Team's findings, that the East Pacific DPS is currently in danger of extinction. Its nesting female abundance and declining trend make the DPS highly vulnerable to threats. Though numerous conservation efforts apply to this DPS, they do not adequately reduce the risk of extinction. We conclude that the East Pacific DPS is currently in danger of extinction throughout its range and therefore meets the definition of an endangered species. The threatened species definition does not apply because the DPS is currently at risk of extinction (*i.e.*, at present), rather than on a trajectory to become so within the foreseeable future.

## **Leatherback Turtle, Overall Species**

The petition under review sought specifically to identify the NW Atlantic population of leatherback sea turtles as a separate DPS and assign it a different status from the global listing. As explained throughout this finding, we have determined that seven leatherback populations would satisfy the tests for recognition under our DPS Policy (*i.e.*, that they are discrete from one another and significant to the overall species), and we have referred to these hypothetically, for purposes of our analysis only, as DPSs. This includes the NW Atlantic DPS. However, we have also determined that, even if these populations were formally recognized as DPSs through a listing process under the Act, each of the DPSs would have the same status as the overall species, which is currently listed throughout its range (globally) as endangered. Nothing in the petition or in the best available information we have reviewed has led us to conclude that there is any basis to disturb the long-standing global listing, which remains in effect and is unaffected by this finding. For completeness, here we present an overview of current information pertaining to the status of the overall species, including a summary of some of the key information from the DPS-specific sections as well as an evaluation of the demographic factors affecting the overall species.

As explained in the *Background* section, the leatherback turtle was originally listed as endangered in 1970 under the precursor to the ESA and was carried forward as an “endangered species” when the ESA became effective. The Services designated the nesting beaches at Sandy Point, St. Croix (43 FR 43688; September 26, 1978) and surrounding marine waters (44 FR 17710; March 23, 1979) as critical habitat. NMFS designated additional marine habitat along 41,914 square miles (108,558 square km) of

the U.S. West Coast as critical habitat (77 FR 4170; January 26, 2012). The Services issued the recovery plans for leatherback turtles in the U.S. Caribbean, Atlantic, and Gulf of Mexico (1991) and U.S. Pacific (1998; <https://www.fisheries.noaa.gov/action/recovery-plans-leatherback-sea-turtle>).

The species has the widest distribution of any reptile, with a global range extending from 71° N, based on an at-sea capture off Norway (Carriol and Vader 2002) to 47° S, based on an at-sea sighting off New Zealand (Eggleston 1971; Eckert *et al.* 2012). The species has several thermoregulatory adaptations to allow such a large latitudinal range, maintain its core temperature while foraging, and avoid overheating during nesting. These include its large size, low metabolic rates, countercurrent heat exchange at the base of its limbs, and peripheral insulation (Frair *et al.* 1972; Greer *et al.* 1973; Paladino *et al.* 1990; Fossette *et al.* 2009; Bostrom *et al.* 2010; Eckert *et al.* 2012; Casey *et al.* 2014; reviewed in Wallace and Jones 2015).

Nesting is restricted to mainly tropical or subtropical beaches. However, nesting also occurs on temperate beaches of the SW Indian Ocean (Pritchard and Mortimer 1999). Nesting usually occurs on high-energy beaches (Pritchard 1976), resulting in high rates of natural erosion. The primary factors influencing shoreline suitability for nesting appear to be a lack of abrasive substrate material, a deep-water approach to minimize energy expenditure needed to reach nesting sites, and proximity to oceanic currents that can facilitate hatchling dispersal (Eckert *et al.* 2012). Leatherback turtles appear to prefer wide, long beaches with a steep slope, deep rock-free sand, and an unobstructed deep water or soft-bottom approach (Pritchard and Mortimer 1999; Eckert *et al.* 2015). As a

result, it has been proposed that the choice of nesting location is based on site characteristics within a geographic location (MacKay *et al.* 2014).

Foraging areas are generally characterized by zones of upwelling, including off the edges of continents, where major currents converge, and in deep-water eddies (Saba 2013). Important foraging areas include but are not limited to: upwelling off the west coasts of North and South America (Benson *et al.* 2011; Roe *et al.* 2014); Benguela Current Marine Ecosystem (Honig *et al.* 2007); and Canadian waters on the Scotian Shelf (James *et al.* 2005a, 2006b, 2007b).

### *Abundance*

Adding together the total indices of nesting female abundance for all DPSs, the total index of nesting female abundance for the species is 32,174 females. This number, however, should be considered as a compilation of seven populations ranging in size from 27 to 20,659 nesting females because nesting female exchange does not occur between DPSs.

Comparisons with historical accounts of nesting female abundance are complicated by the discovery of new nesting beaches over time, changes in remigration intervals and/or clutch frequency, and modified observational effort. Abundance estimates for even large nesting beaches were not available prior to 1950 (Rivalan *et al.* 2006), several large nesting beaches were not discovered until the 1960s or later (NMFS and USFWS 2013), and monitoring efforts were variable over time. Pritchard's 1971 global estimate of 29,000 to 40,000 nesting females included a maximum estimate (*i.e.*, 40,000 nesting females) based on the assumption that large nesting aggregations had yet to be discovered (Pritchard 1971); this estimate did not include large nesting female

abundances from the East Pacific and SE Atlantic Oceans. At that time, the nesting aggregation at Terengganu, Malaysia nesting population was thought to be one of the largest; however it has since been extirpated (Chan and Liew 1996). In 1982, Pritchard revised his initial global estimate to 115,000 nesting females, based largely on the nesting beaches in Pacific Mexico ( $n = 75,000$ ; Pritchard 1982). However, the 1982 estimate was extrapolated from a brief aerial survey and may have been an overestimate (Pritchard 1996). When the Mexico nesting population collapsed, Spotila (1996) estimated the total global estimate to be 34,500 nesting females, with a range of 26,200 to 42,900 nesting females. However, this estimate did not include the nesting aggregation in Gabon, which in 2002 was identified as the largest in the world at that time, with tens of thousands of nesting females (Witt *et al.* 2009). Recent data indicate less than 9,000 nesting females in Gabon (Formia in progress). Thus, we find that leatherback nesting female abundance has declined rapidly in several populations. Our total index of nesting female abundance for the species, which does include the largest nesting aggregations from all DPSs, is lower than previous estimates by at least 10,000 females.

Species go extinct through the loss of populations. Therefore, the loss of any of these populations (which we refer to in this finding hypothetically as DPSs) would increase the extinction risk of the species. Most of the DPSs exhibit total indices of nesting female abundances that place them at risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017). The current total index of nesting female abundance for the species reflects the impact of threats that have affected the species to

this point. This reduced abundance renders it particularly vulnerable to threats and contributes to its extinction risk.

### *Productivity*

Nest trends are decreasing across the species, except at the least abundant nesting aggregation in Brazil (*i.e.*, the SE Atlantic DPS), with a total index of 27 nesting females, which is increasing by 4.8 percent annually. Current nest trends are declining at rates ranging from -0.3 percent (within the SW Indian DPS) to -9.3 percent (the overall decline for the NW Atlantic DPS). Historical declines are even larger. Aerial surveys of nesting beaches in Mexico detected declines from over 70,000 nesting females in 1982 to fewer than 250 in 1998, with an annual mortality rate of 22.7 percent (Spotila 2000) and an overall decline of 97.4 percent in three generations (Wallace *et al.* 2013). The Terengganu, Malaysia nesting aggregation has declined by 17.9 percent annually from 1967 to 2010. It was been reduced to less than one percent of its original size between the 1950s and 1995 (Chan and Liew 1996) and is now considered functionally extirpated. Significant declines in nesting have been documented for other populations (Benson *et al.* 2015). Declining nesting trends reflect the impact of threats that have been operating on the species, and these trends increase the extinction risk of the species.

### *Spatial Distribution*

The species occurs over a broad spatial range, in tropical and temperate waters worldwide, from 71° N to 47° S (Goff and Lien 1988; Carriol and Vader 2002; McMahon and Hayes 2006; Shillinger *et al.* 2008; Wallace *et al.* 2010; Benson *et al.* 2011; Eckert *et al.* 2012). It nests and forages across a wide spatial range, which provides some degree of resilience against local impacts to nesting and foraging areas. The DPSs

are reproductively isolated with little to no gene flow connecting them. However, within some DPSs there is fine-scale population structure (Dutton *et al.* 1999; Dutton *et al.* 2003; Dutton *et al.* 2013; Molfetti *et al.* 2013). These subpopulations exhibit metapopulation dynamics, which make a DPS more resilient to stochastic and environmental changes. It is likely that all DPSs once exhibited such dynamics, given the ephemeral, high-energy beaches where they nest and their regional, but not necessarily beach-specific, philopatry (Dutton *et al.* 1999; Dutton *et al.* 2013). However, the reduction of nesting aggregations within a DPS has likely reduced or removed this structure, and the associated resilience, in some DPSs and in the overall species.

### *Diversity*

Relative to other sea turtle species, the leatherback turtle has low genetic diversity and shallow mtDNA coalescence (Dutton *et al.* 1999), reflecting its recent global radiation, *i.e.*, post-Pleistocene expansion from a refugium in the Indian Ocean (Dutton *et al.* 1999). As a species, it uses diverse and widely distributed nesting and forage areas. Differences in size at maturity, remigration rate, clutch frequency, and clutch size likely reflect environmental variability among DPSs (Saba *et al.* 2008; Saba *et al.* 2015). The age of the species and its flexible use of multiple foraging and nesting areas indicate that the species has some resilience to stochastic and environmental changes.

### *Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range*

The destruction or modification of nesting habitat is a threat to most leatherback turtles, and in some areas, this threat is major, as a result of development, erosion, or obstruction from logs. By the year 2025, the UN Educational, Scientific and Cultural Organization (2001) forecasts that human population growth and migration will result in

75 percent of people living within 60 km of the sea. This will place significant additional pressure on coastal habitats.

Coastal development and associated activities cause accelerated erosion rates and interruption of natural shoreline migration (National Research Council 1990). Numerous beaches are eroding due to both natural (*e.g.*, storms, sea level changes, waves, shoreline geology) and anthropogenic (*e.g.*, development and expansion, construction of armoring structures, groins, jetties, marinas, coastal development, inlet dredging) factors. Such shoreline erosion has led and will continue to lead to a loss of nesting habitat for leatherback turtles and potential loss of nests from inundation. Erosion or inundation and accretion of sand above incubating nests appear to be the principal abiotic factors that negatively affect incubating egg clutches in some areas (Dow *et al.* 2007; USFWS 1999; NMFS and USFWS 2013). Shoreline structuring can also physically prevent females from reaching suitable nesting habitat or prevent them from returning to sea (Witherington *et al.* 2011).

Low hatching success, relative to other sea turtle species, is characteristic of many leatherback populations despite high fertility rates (reviewed by Bell *et al.* 2003; Eckert *et al.* 2012). Nest relocation is undertaken as a conservation measure in some locations when erosion (or poaching and predation) threaten the viability of a nest. However, studies have found that hatching success of nests in hatcheries or nests relocated to another area of a beach is lower than *in situ* nests (reviewed in Hernández *et al.* 2007; Eckert *et al.* 2012). In addition, nest relocation results in altered sand temperatures, which influences the sex ratio of hatchlings produced (Sieg *et al.* 2011).

Coastal development and expansion also contributes to habitat degradation via artificial lighting (*i.e.*, light pollution). The presence of artificial lighting on or adjacent to nesting beaches alters the behavior of nesting females (often deterring nesting) and is often fatal to post-nesting females and emerging hatchlings, when they are attracted to terrestrial light sources and drawn away from the water (Witherington 1992; Sella *et al.* 2006; Witherington *et al.* 2014). As hatchlings head toward lights or meander along the beach, their exposure to predators and likelihood of desiccation are greatly increased. Artificial lighting may also affect hatchlings that successfully find the water, causing them to be misoriented after entering the surf zone or while in nearshore waters.

The modification of nesting habitat generally results in loss of productivity for the species, as a result of reductions in nest and hatching success. In addition, several DPSs experience nesting beach habitat modifications (*e.g.*, artificial lighting, logs, and other obstructions) that result in the death of nesting females and hatchlings. Therefore, abundance is also reduced, posing an even greater threat to the continued existence of the turtles of the DPS. The loss and modification of nesting habitat poses a major threat to the species.

#### *Overutilization for Commercial, Recreational, Scientific, or Educational Purposes*

Historically, the harvest of turtles and eggs was the primary threat to the species, leading to the loss of severe depletion of many nesting aggregations worldwide (Spotila *et al.* 1996). At one point in time, egg harvest was ubiquitous with all nests taken at many beaches (Chan and Liew 1996; Sarti *et al.* 2007; reviewed by Eckert *et al.* 2012). For the NW Atlantic, NE Indian, and West Pacific DPSs, legal harvest of turtle and/or eggs continues. Despite laws in many countries, the poaching of eggs continues at most

nesting beaches, ranging in severity from minor at monitored or protected beaches to near 100 percent harvest at unmonitored beaches. Nesting females, and turtles caught at sea, continue to be poached for their meat, eggs, and fat in many locations (Eckert *et al.* 2012). As described in detail in the prior sections evaluating the status of each individual DPS, the harvest of eggs and turtles is a threat to each and to the species overall, and for the NE Indian and West Pacific DPSs, it is a primary threat. The legal and illegal harvest of turtles and eggs poses a threat to the species.

#### *Disease or Predation*

We do not have adequate information on disease to assess its impact on the species. However, we have enough information to conclude that predation is clearly a threat. Numerous species prey on leatherback eggs and hatchlings. Eckert *et al.* (2012) provide an exhaustive list of the documented predators for each life stage and area. For eggs, common predators include ants, ghost crabs, monitor lizards, crows, mongoose, domestic and feral dogs, and feral pigs (Eckert *et al.* 2012). For hatchlings, common predators include the terrestrial predators listed above as well as numerous species of carnivorous fish, including sharks. Sharks and killer whales, and in some areas jaguars and crocodiles, prey on subadult and adult turtles. Predation on eggs and hatchlings is common and reduces productivity of the species; predation on subadults and adults is less prevalent but reduces abundance when it occurs. Predation is a threat to the species, and for some DPSs, it is a major threat.

#### *Inadequacy of Existing Regulatory Mechanisms*

Numerous regulatory mechanisms provide certain protections to sea turtles at the international, regional, national, and local levels. For example, the harvest of sea turtles

and their eggs is prohibited by regional conventions and national laws. Fisheries bycatch is also addressed, although not comprehensively, by several international and national instruments and/or governing bodies. However, as we detail below and has been discussed in prior sections reviewing each individual DPS, these measures are often poorly implemented or enforced, resulting in inadequate protections against the threats they are designed to ameliorate.

In some nations (*e.g.*, South Africa) sea turtles were among the first species to receive legal protections and have been the focus of concentrated conservation efforts. However, current regulatory mechanisms often fall short of preventing further population declines and ensuring persistence (Eckert *et al.* 2012). For many nations the regulations in place are inadequate (usually due to lack of enforcement and implementation) to address the impacts of a wide range of anthropogenic activities that directly injure and kill turtles, disturb eggs, disrupt necessary behaviors, and alter terrestrial and marine habitats used by the species. In many areas, regulations for the harvest of turtles and eggs are inadequate due to a lack of enforcement. In some areas, the regulation of fisheries bycatch do not adequately reduce associated mortality. Fishery observer coverage is often inadequate to accurately estimate leatherback bycatch.

Due in part to their worldwide distribution and highly migratory nature, combined with nesting site fidelity, leatherback turtles require international, national, regional, and local protection. Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments and concluded that they vary in their effectiveness. Often, international treaties do not realize their full potential because: they do not include all key nations; do not specifically address sea turtle conservation; are handicapped by the lack

of a sovereign authority to promote enforcement; and/or lack of legally-binding requirements. Lack of implementation or enforcement by some nations may make them less effective than if they were implemented in a more consistent manner across the target region. A thorough discussion of this topic is available in the 2002 special issue of the *Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation* (Hykle 2002). Additional information on national, regional, and local protection is provided in the prior sections of this finding relating to each individual DPS.

In summary, numerous regulatory mechanisms protect leatherback turtles, eggs, and nesting habitat throughout the range of the species. Although the regulatory mechanisms provide some protection, many do not adequately reduce the threat that they were designed to address, generally as a result of limited implementation or enforcement. As a result, bycatch, incomplete nesting habitat protection, and poaching remain threats to the species. We conclude that the inadequacy of the regulatory mechanisms is a threat to the leatherback turtle.

#### *Fisheries Bycatch*

Fisheries bycatch is the primary threat to leatherback turtles (Crowder 2000; Spotila *et al.* 2000; Lewison *et al.* 2004; Wallace *et al.* 2011; Wallace *et al.* 2013; Angel *et al.* 2014). It is a primary threat to all DPSs. Leatherback turtles are susceptible to bycatch in a wide range of fisheries, from large scale commercial to artisanal. Gear types that affect leatherbacks include: longlines, purse seines, driftnets, gillnets, trawls, pots/traps, and pound nets (Gray and Diaz 2017). Turtles often drown after becoming entangled in nets and other gear or become injured and possibly die as a result of hooking

or interactions with the gear. While bycatch in pelagic shallow-set swordfish longline fisheries has received the most attention to date, small-scale coastal fisheries occur worldwide, employing over 99 percent of the world's 51 million fishers (FAO 2011).

Bycatch data are most commonly collected by trained observers on fishing vessels or via surveys or interviews (Lewison *et al.* 2015). Though often the best available data on bycatch, observer data generally cover less than five percent of fisheries' total effort (Finkbeiner *et al.* 2011) and are rarely available for small-scale fisheries (Wallace *et al.* 2013; Lewison *et al.* 2015). The use of different metrics also makes the data difficult to compare among fisheries, gear types, and regions (Lewison *et al.* 2015). Therefore, estimates of bycatch and resulting mortality often underestimate the magnitude of this threat.

Furthermore, IUU fishing is a significant yet unquantified threat to sea turtles worldwide. In addition to killing and injuring turtles, it undermines national and regional efforts to estimate fisheries bycatch. IUU fishing represents up to 26 million tonnes of fish caught annually (<http://www.fao.org/iuu-fishing/en/>). We have no estimates of the impacts to leatherback turtles from IUU fishing, though interaction and mortality rates are likely high because of the magnitude of this additional fishing pressure and because it is unregulated.

Generally, leatherback turtles do not attempt to consume the bait associated with fishing gear, as other sea turtles do, but become entangled in fishing gear (Lewison *et al.* 2015). Longline fisheries involve the deployment of a horizontal main line and vertical branchlines with baited hooks, which may entangle leatherback turtles. Bycatch reduction measures include using circle hooks, finfish bait, minimizing soak times, and limiting

mainline length (Angel *et al.* 2014; <https://www.fisheries.noaa.gov/national/bycatch/fishing-gear-pelagic-longlines#risks-to-sea-turtles>). Purse seines capture schools of fish in a vertical wall of netting that can be closed at the bottom (<https://www.fisheries.noaa.gov/national/bycatch/fishing-gear-purse-seines>); bycatch rates are generally much lower than longline bycatch rates (Angel *et al.* 2014). Leatherback turtles also become entangled and drowned in drift or set gillnets (<https://www.fisheries.noaa.gov/national/bycatch/fishing-gear-gillnets>). Gillnets can be devastating to leatherback populations when set near nesting beaches and represent the primary threat to leatherback turtles in some areas (*e.g.*, Trinidad; Eckert and Eckert 2005). Trawl fisheries drag nets along the substrate or through the water column and can capture and drown sea turtles. Although TEDs may mitigate this threat, they are not always required or used in all areas. Vertical lines extending and/or connecting pot and trap gear with surface buoys commonly entangle and can kill leatherback turtles.

Longline and net fisheries are often the greatest threats to leatherback turtles. In a global study of sea turtle bycatch, Wallace *et al.* (2013) compiled data (n = 239 records) published between 1990 and 2011 to compare gear types (longline, net, and trawl) and their impacts to leatherback RMUs, which are similar to the DPSs discussed in this rule, though their exact boundaries differ. Wallace *et al.* (2013) defined high bycatch impact as follows: a weighted median bycatch per unit effort (BPUE) greater than or equal to one; median mortality rate greater than or equal to 0.5; and affecting adult or subadult turtles. They found that longline bycatch had a high impact on SW Atlantic, SE Atlantic, and SW

Indian RMUs and that net bycatch had a high impact on the NW Atlantic and East Pacific RMUs (Wallace *et al.* 2013).

Integrating catch data from over 40 nations and bycatch data from 13 international observer programs, Lewison *et al.* (2004) estimated the numbers of leatherback turtles taken by pelagic longliners to be more than 50,000 leatherback turtles in just one year (2000). With over half of the total fishing effort (targeting tuna and swordfish) occurring in the Pacific Ocean, an estimated 20,000 leatherback turtles interacted with longline fishing gear, with 1,000 to 3,200 mortalities in 2000 (Lewison *et al.* 2004). However, Beverly and Chapman (2007) estimated sea turtle longline bycatch mortality to be approximately 20 percent of that estimated by Lewison *et al.* (2004), or approximately 200 to 640 leatherback turtle mortalities annually. We consider the estimate of Beverly and Chapman (2007) to be more realistic, considering the low nesting females abundance of Pacific leatherback turtles, and because Beverly and Chapman (2007) combined the effort data from Lewison *et al.* (2004) with bycatch data from Molony (2005) that differentiated between deep-set and shallow-set fisheries (which have different interaction rates).

In the Pacific Ocean, Roe *et al.* (2014) predicted leatherback turtle bycatch hotspots by comparing the satellite tracks of 135 adult turtles with longline fishing effort. The greatest bycatch risk occurred adjacent to primary nesting beaches of the West Pacific DPS. Bycatch risk was also high in the South Pacific Gyre, where the East Pacific DPS forages. Expanding on this study, a study of observer data from 34 swordfish-targeting shallow-set longline fleets found there were 331 leatherback turtle interactions

between 1989 and 2015 (Clarke 2017). Clarke (2017) identified two bycatch hotspot areas: central North Pacific Ocean and eastern Australia (Clarke 2017).

In the Atlantic Ocean, Fossette *et al.* (2014) compared leatherback telemetry data to longline fishing effort data from ICCAT to identify nine areas in which leatherback turtles are exposed to bycatch associated with high longline fishery pressure. The high pressure fishing areas include foraging areas in the North and South Atlantic Ocean and in waters off Brazil and western Africa. These high pressure fishing areas are not comparable to those identified by Roe *et al.* (2014), who used a different methodology, but both studies identify high risk areas within each ocean basin.

Additional bycatch information that we have set out in prior sections specific to each DPS applies to our consideration of the risk to the overall species. In summary, fisheries bycatch is a threat that is encountered by numerous juvenile and adult leatherback turtles. Mortality rates are often high, and individuals that are released may experience injuries or sublethal effects associated with entanglement, submergence, or handling. Fisheries bycatch reduces abundance, and when it prevents nesting females from returning to nesting beaches, reduces productivity as well. Fisheries bycatch is the primary threat to the leatherback species.

#### *Vessel Strikes*

Vessel strikes pose a threat to the species throughout its range. As mature individuals move from oceanic foraging areas into coastal waters to reproduce, they are exposed to a greater concentration of vessels. Vessel strikes off nesting beaches may injure or kill these individuals, reducing the abundance and productivity of the DPS. Most vessel strikes likely go unnoticed or unreported, making this threat potentially much

more significant than documented occurrences would suggest. Vessel strikes are a threat to the leatherback species.

### *Pollution*

We define pollution as including contaminants, marine debris, and ghost or derelict fishing gear. Such interactions are likely to go unnoticed and unreported and thus likely present a more significant impact than documented occurrences would suggest. Leatherback turtles of all life stages are vulnerable to oil spills, on land and at sea, where exposure to oil and dispersants occurs via contact (*i.e.*, physical fouling), inhalation, or ingestion (reviewed by Stacy *et al.* in press).

Marine debris is ubiquitous throughout the range of the species. Marine debris includes plastics (including plastic bags), microplastics, derelict fishing gear (*e.g.*, ghost nets and other discarded or lost gear), and other man-made materials. Leatherback turtles may directly consume floating plastics, mistaking it for their gelatinous prey or accidentally ingest plastics while foraging. In particular, plastic bags appear similar to jellyfish in the marine environment, inappropriately triggering the sensory cue to feed (Schuyler *et al.* 2014; Nelms *et al.* 2016). Plastic bags have been found during necropsy of stranded leatherback turtles, and phthalates derived from plastics have been found in leatherback egg yolk (Lebreton *et al.* 2018). Mrosovsky *et al.* (2009) reviewed 408 necropsy records from 1885 to 2007 and found evidence of plastic in the gastrointestinal tract of 34 percent of leatherback turtles, including some cases in which the plastic obstructed the passage of food through the gut. The most commonly identified items were plastic bags, fishing lines, twine, and fragments of mylar balloons. Ghost or derelict fishing gear include discarded or lost nets, line, and other gear. Ghost fishing gear can

drift in the ocean and fish unattended for decades and kill numerous individuals (Wilcox *et al.* 2013). The main sources of ghost fishing gear are gillnet, purse seine, and trawl fisheries (Stelfox *et al.* 2016). Marine debris affects leatherback turtles via ingestion or entanglement and can reduce food intake and digestive capacity, cause distress and/or drowning, expose turtles to contaminants, and in some cases cause direct mortality (Mrosovsky *et al.* 2009; NMFS and USFWS 2013). In terms of microplastics, all samples analyzed from all species (including leatherbacks) had microplastics evident in their gastro-intestinal tracts (Duncan *et al.* 2018). Given the increase of pollution entering the marine environment over the past 30 years or approximately 5.2 to 19.3 million tonnes per year (Lebreton *et al.* 2018), we conclude that pollution is a threat to the species.

#### *Natural Disasters and Oceanographic Regime Shifts*

Leatherback turtles are susceptible to the impacts of natural disasters and oceanographic regime shifts as a result of their nesting and foraging preferences. Nesting usually occurs on high-energy beaches that are inherently unstable (Pritchard 1976) and which are susceptible to natural erosion. The primary factors influencing shoreline suitability for nesting appear to be a lack of abrasive substrate material, a deep-water approach to minimize energy expenditure needed to reach nesting sites, and proximity to oceanic currents that can facilitate hatchling dispersal (Eckert *et al.* 2012). Leatherback turtles nest lower on the beach than other species, exposing their nests to erosion and inundation. Storm events, King Tides, tsunamis, and hurricanes can destroy or modify preferred nesting beaches of some DPSs.

Gelatinous prey have relatively low energy content, requiring leatherback turtles to consume large quantities to meet metabolic demands (Heaslip *et al.* 2012; Jones *et al.*

2012). Leatherback turtles likely maximize their caloric intake by aligning their foraging behavior to prey distribution abundance. Foraging areas are generally characterized by zones of upwelling, including off the edges of continents, where major currents converge, and in deep-water eddies (Saba 2013). Some of these areas experience oceanographic regime shifts that alter water temperature, downwelling, Ekman upwelling, sea surface height, chlorophyll-a concentration, and mesoscale eddies (Bailey *et al.* 2013; Benson *et al.* 2011). These shifts alter prey availability, and thus productivity parameters (*e.g.*, remigration rates, clutch size, and clutch frequency), for leatherback turtles. Some DPSs are not affected by such shifts because they have access to diverse foraging areas, such as: coastal and pelagic waters; subtropical, temperate, and boreal waters; and ephemeral eddies (Neeman *et al.* 2015). Such flexibility allows the leatherback turtle to consume large amounts of prey at various locations throughout the year.

We conclude that natural disasters and oceanographic regime shifts are threats to the species, affecting some but not all populations, depending on the location of nesting and foraging areas. These threats reduce productivity by reducing nesting, nesting habitat, and nest and hatching success.

### *Climate Change*

Climate change is a threat that affects leatherback turtles of all life stages and within all DPSs. A warming climate and rising sea levels can impact leatherback turtles through changes in beach morphology, increased sand temperatures leading to a greater incidence of lethal incubation temperatures, changes in hatchling sex ratios, and the loss of nests or nesting habitat due to beach erosion (Benson *et al.* 2013).

Impacts from climate change, especially due to global warming, are already being observed and are likely to become more apparent in future years (IPCC 2007a). In its Fifth Assessment Report, the IPCC (2014) stated that the globally averaged combined land and ocean surface temperature data has shown a warming of 0.85°C from 1880 to 2012. The mean rate of globally averaged sea level rise was 1.7 millimeters annually between 1901 and 2010, 2.0 millimeters annually between 1971 and 2010, and 3.2 millimeters annually between 1993 and 2010. Climate model projections exhibit a wide range of plausible scenarios for both temperature and precipitation over the next several decades. The global mean surface temperature change for the period 2016 to 2035 relative to 1986 to 2005 will likely be in the range of 0.3° to 0.7°C (medium confidence; IPCC 2014). The global ocean temperature will continue to warm, and increases in seasonal and annual mean surface temperatures are expected to be larger in the tropics and Northern Hemisphere subtropics (*i.e.*, where leatherback turtles nest; IPCC 2014). Under Representative Concentration Pathway 8.5, the change in global mean sea level rise for the mid- and late 21st century relative to the reference period of 1986 to 2005 is projected to be 0.30 meters higher from 2046 to 2065 and 0.63 meters higher from 2081 to 2100, with a rate of sea level rise during 2081 to 2100 of 8 to 16 millimeters annually (medium confidence; IPCC 2014).

For all sea turtles, including leatherback turtles, a warming climate and rising sea levels are likely to result in changes in beach morphology, increased sand temperatures leading to a greater incidence of lethal incubation temperatures, changes in hatchling sex ratios, and the loss of nests and nesting habitat due to beach erosion (Benson *et al.* 2015; Hamann *et al.* 2013). Leatherback turtles are most likely to be affected by climate change

at nesting beaches due to warming temperatures, sea level rise, and storm events and due to oceanic changes that are likely to alter foraging and migration. Warming temperatures and increased precipitation at nesting beaches affect reproductive output including hatching success, hatchling emergence rate, and hatchling sex ratios (*e.g.*, Hawkes *et al.* 2009). Sea level rise results in a reduction or shift in available nesting beach habitat, an increased risk of erosion and nest inundation (*e.g.*, Boyes *et al.* 2010), and reduced nest success (Fish *et al.* 2005; Fuentes *et al.* 2010; Fonseca *et al.* 2013). Increased frequency and severity of storm events impact nests and nesting habitat, thus reducing nesting and hatching success (*e.g.*, Van Houtan and Bass 2007; Fuentes and Abbs 2010). Changes in productivity affect the abundance and distribution of forage species, resulting in changes in the foraging behavior and distribution of leatherback turtles (*e.g.*, Saba *et al.* 2008, 2012) as well as changes in leatherback fitness and growth. Changes in water temperature lead to a shift in range and changes in phenology (timing of nesting seasons, timing of migrations) and different threat exposure (*e.g.*, Saba *et al.* 2008, 2012).

Increasing sand temperatures will alter the thermal regime of incubating nests, resulting in altered sex ratios and reduced hatching output (Hawkes *et al.* 2009). Leatherback turtles exhibit temperature-dependent sex determination (reviewed by Binckley and Spotila 2015), whereby phenotypic sex is determined by temperatures experienced during the thermosensitive period of egg incubation. A 1:1 sex ratio is produced when this pivotal temperature lies between 29.2 and 30.4°C for leatherback turtles in Malaysia, 29.2 and 29.8°C in French Guiana/Suriname, and 29.2 and 29.5 °C in Pacific Costa Rica (Binckley and Spotila 2015). Warmer temperatures produce more female embryos (Mrosovsky *et al.* 1984; Hawkes *et al.* 2007), but temperatures over 32

°C are likely to result in death. As temperatures continue to increase, emergence rates decrease (Santidrián Tomillo *et al.* 2015), removing any advantage of increased female production. Santidrián Tomillo *et al.* (2015) conclude that leatherback turtles may not survive if temperatures rise as projected by current climate change models. Increases in precipitation might temporarily reduce the temperatures at some nesting beaches thereby mitigating some impacts relative to increasing sand temperatures.

Beach erosion and nest inundation already threaten leatherback nesting habitat globally. Sea level rise is likely to increase the number of nests lost to erosion and inundation. Such loss of nests is especially problematic in areas prone to storm events, which are likely to increase in intensity and duration, and in areas where coastal development impedes natural shoreline migration.

Climate change is also likely to alter the productivity in some marine environments, which could affect leatherback prey availability. With reports on the increasing incidence of jellyfish blooms in some locations, there is the perception that jellyfish abundance is increasing globally (Condon *et al.* 2012), which could result in more prey for leatherback turtles (Hawkes *et al.* 2009). However, after analyzing all available long-term datasets on jellyfish abundance, Condon *et al.* (2012) found that there is no robust evidence for a global increase in jellyfish. Rather, jellyfish populations undergo larger, worldwide oscillations with an approximate 20-year periodicity (Condon *et al.* 2012). Additional monitoring is needed to determine whether the weak linear trend in jellyfish abundance since 1970 represents an actual increase or is a phase of an oscillation (Condon *et al.* 2012). Therefore, the effects of climate change on productivity are uncertain.

As described in prior sections with respect to each individual population, some impacts from climate change have already been observed. At several nesting beaches, increased erosion occurs, and sex ratios are severely skewed toward females. Beach erosion reduces productivity. Although the skew toward females could increase productivity in the short-term, it is often correlated with low hatching success. For these reasons, climate change is a threat to the species.

### *Conservation Efforts*

The ESA requires the Services to make their listing determinations solely on the basis of the best scientific and commercial data available, after conducting a status review, and after taking into account those efforts, if any, being made by any State or foreign nation to protect the species, whether by predatory control, protection of habitat and food supply, or other conservation practices, within any area under its jurisdiction, or on the high seas (16 U.S.C. 1533 (b)(1)(A)). In addition, the Services published a policy for the evaluation of domestic conservation efforts which have yet to be implemented or to show effectiveness (68 FR 15100; March 28, 2003). We did not identify any conservation efforts that required such evaluation for leatherbacks (*i.e.*, the conservation efforts reviewed are international in nature or have already been implemented to a sufficient degree that they have a track record of being effective or not being effective). Several conservation efforts have been previously discussed in prior sections evaluating regulatory mechanisms with respect to each DPS. Therefore, the list below describes only those conservation efforts that have not been previously discussed and that apply generally to the leatherback species rather than being clearly associated with a particular population. We considered these efforts prior to making our listing determination. After

reviewing these efforts, we concluded that they have been somewhat effective, in that they have prevented this endangered species from going extinct. However, these efforts have not reduced the threats to a level at which protections under the ESA are no longer necessary.

African Convention on the Conservation of Nature and Natural Resources (Algiers Convention): Adopted in September 1968, the contracted states were “to undertake to adopt the measures necessary to ensure conservation, utilization and development of soil, water, floral and faunal resources in accordance with scientific principles and with due regard to the best interests of the people.” The Algiers Convention recently has undergone revision, and its objectives are to enhance environmental protection, foster conservation and sustainable use of natural resources, and harmonize and coordinate policies in these fields with a view to achieving ecologically rational, economically sound, and socially acceptable development policies and programs. Additional information is available at

*<http://www.unep.ch/regionalseas/legal/afr.htm>*.

Atlantic Sea Turtle Network (ASO): Created in 2003 to foster greater collaboration in southern Brazil, Uruguay, and Argentina for the protection of sea turtles and their habitats. ASO represents dozens of local and regional NGOs and government agencies as well as hundreds of community members. ASO and its partners have significantly advanced policies to protect sea turtles from fisheries interactions, which is one of the most severe threats in the region. Brazil plays a major role in South American (and global) sea turtle conservation and research, and it serves as an example to other countries. Projeto TAMAR, a partnership of the Centro TAMAR/ICMBio, government

agencies, and Fundação Pró TAMAR, has been active since 1980. Today, the group carries out sea turtle research and conservation from 22 stations on the coast and the offshore islands of Brazil. Another NGO based in the southern Brazilian state of Rio Grande do Sul, called NEMA has been collecting systematic sea turtle stranding data since 1990. Those data have been instrumental to conservation efforts in Brazil and have shown that southern Brazil has the highest stranding rates for loggerheads in the western Atlantic Ocean.

Association of Southeast Asian Nations (The ASEAN) Ministers on Agriculture and Forestry (AMAF): A Memorandum of Understanding (MoU) on ASEAN sea turtle conservation was created in 1999. From this, a Sea Turtle Conservation and Protection Program and Work plan has developed; research and monitoring activities have also been produced regionally (Kadir 2000). The objectives of this Memorandum of Understanding, initiated by ASEAN, are to promote the protection, conservation, replenishing, and recovery of sea turtles and their habitats based on the best available scientific evidence, taking into account the environmental, socio-economic and cultural characteristics of the Parties. It currently has nine signatory states in the South East Asian Region (<http://document.seafdec.or.th/projects/2012/seaturtles.php>).

Andaman and Nicobar Island Environmental Team (ANET): A division of the Centre for Herpetology/ Madras Crocodile Bank Trust has been conducting surveys and monitoring since 1991. Over the last few years, conservation and monitoring of sea turtles in these islands has been carried by Dakshin Foundation and Indian Institute of Science in collaboration with ANET, centered around a leatherback monitoring program on Little Andaman Island. A multi- institution stakeholder platform for marine

conservation, including government and non- governmental agencies, was established by these groups to facilitate the conservation of marine turtles and other endangered species (Tripathy *et al.* 2012). The Trust, along with the Wildlife Institute of India and Ministry of Environment and Forests, produced a series of manuals on sea turtle conservation, management and research to help forest officers, conservationists, NGOs and wildlife enthusiasts conduct sea turtle conservation and research programs (ANET, 2003 as cited in Shanker and Andrews 2004). A consolidated manual has been produced to achieve these goals by Dakshin Foundation and the Trust (Tripathy *et al.* 2012).

Central American Regional Network: This collaborative effort created the national sea turtle network in each country of the region, as well as the development of first hand tools, such as a regional diagnosis, a 10-year strategic plan, a manual of best practices, and four regional training and information workshops for people in the region (*e.g.*, Chacón and Arauz, 2001). This initiative is managed by stakeholders in various sectors (private, non-governmental and governmental) across the region.

Convention on the Conservation of Migratory Species of Wild Animals (CMS): This Convention, also known as the Bonn Convention or CMS, is an international treaty that focuses on the conservation of migratory species and their habitats. As of December 2018, the Convention had 127 Parties, including Parties from Africa, Central and South America, Asia, Europe, and Oceania. While the Convention has successfully brought together about half the countries of the world with a direct interest in sea turtles, it has yet to realize its full potential (Hykle 2002). Its membership does not include a number of key countries, including Canada, China, Indonesia, Japan, Mexico, Oman, and the United States. Under the CMS, two Memoranda of Understanding (MOUs) apply to leatherback

turtles: the MOU concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa and the MOU on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia. Additional information is available at <http://www.cms.int>.

Convention on Biological Diversity (CBD): The primary objectives of this international treaty are: (1) the conservation of biological diversity, 2) the sustainable use of its components, and 3) the fair and equitable sharing of the benefits arising out of the utilization of genetic resources. This Convention has been in force since 1993 and had 193 Parties as of March 2013. While the Convention provides a framework within which are broad conservation objectives, it does not specifically address sea turtle conservation (Hykle 2002). Additional information is available at <http://www.cbd.int>.

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES): Known as CITES, this Convention was designed to regulate international trade in a wide range of wild animals and plants. CITES was implemented in 1975 and currently has 183 Parties. Although CITES has been effective at minimizing the international trade of sea turtle products, it does not limit legal harvest within countries, nor does it regulate intra-country commerce of sea turtle products (Hykle, 2002). The leatherback turtle is included (since 1977) in CITES Appendix I, which bans trade, including individuals and products, except as permitted for exceptional circumstances, not to include commercial purposes (Lyster 1985). Additional information is available at <http://www.cites.org>.

Convention on the Conservation of European Wildlife and Natural Habitats: Also known as the Bern Convention, the goals of this instrument are to conserve wild flora and

fauna and their natural habitats, especially those species and habitats whose conservation requires the cooperation of several States, and to promote such cooperation. The Convention was enacted in 1982 and currently includes 51 European and African States and the European Union. Additional information is available at [http://www.coe.int/t/dg4/cultureheritage/nature/bern/default\\_en.asp](http://www.coe.int/t/dg4/cultureheritage/nature/bern/default_en.asp).

Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region (Abidjan Convention): The Abidjan Convention covers the marine environment, coastal zones, and related inland waters from Mauritania to Namibia. The Abidjan Convention countries are Angola, Benin, Cameroon, Cape Verde, Congo, Cote d'Ivoire, Democratic Republic of Congo, Equatorial Guinea, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Liberia, Mauritania, Namibia, Nigeria, Sao Tome and Principe, Senegal, Sierra Leone, and Togo. The Abidjan Convention is an agreement for the protection and management of the marine and coastal areas that highlights sources of pollution, including pollution from ships, dumping, land-based sources, exploration and exploitation of the sea-bed, and pollution from or through the atmosphere. The Convention also identifies where co-operative environmental management efforts are needed. These areas of concern include coastal erosion, specially protected areas, combating pollution in cases of emergency and environmental impact assessment.

Convention for the Protection Management and Development of the Marine and Coastal Environment of the Eastern African Region (Nairobi Convention): The Nairobi Convention was signed in 1985 and came into force in 1996. This instrument “provides a mechanism for regional cooperation, coordination and collaborative actions, and enables

the Contracting Parties to harness resources and expertise from a wide range of stakeholders and interest groups towards solving interlinked problems of the coastal and marine environment.” Parties are responsible for “the conservation and wise management of the sea turtle populations frequenting their waters and shores [and] agree to work closely together to improve the conservation status of the sea turtles and the habitats upon which they depend.” The Western Indian Ocean-Marine Turtle Task Force, which was created under the Nairobi Convention and the IOSEA, plays a role in sea turtle conservation. This is a technical, non-political working group comprised of specialists from eleven countries: Comoros, France (La Réunion), Kenya, Madagascar, Mauritius, Mozambique, Seychelles, Somalia, South Africa, United Kingdom and Tanzania, as well as representatives from inter-governmental organizations, academic, and non-governmental organizations within the region. Additional information is available at <http://www.unep.org/NairobiConvention>.

Convention for the Protection of the Marine Environment of the North-East Atlantic: Also called the OSPAR Convention, this 1992 instrument combines and updates the 1972 Oslo Convention against dumping waste in the marine environment and the 1974 Paris Convention addressing marine pollution stemming from land-based sources. The convention is managed by the OSPAR Commission, which is comprised of representatives from 15 signatory nations (Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, The Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, and United Kingdom), as well as the European Commission, representing the European Community. The mission of the OSPAR Convention “...is to conserve marine ecosystems and safeguard human health in the North-East Atlantic by

preventing and eliminating pollution; by protecting the marine environment from the adverse effects of human activities; and by contributing to the sustainable use of the seas.” Leatherback turtles are included on the OSPAR List of Threatened and/or Declining Species and Habitats, used by the OSPAR Commission for setting priorities for work on the conservation and protection of marine biodiversity. Additional information is available at <http://www.ospar.org>.

Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region: Also called the Cartagena Convention, this instrument that benefits turtles of the Northwest Atlantic leatherback DPS, has been in place since 1986 and currently has 38 member states and territories. Under this Convention, the component that relates to leatherback turtles is the Protocol Concerning Specially Protected Areas and Wildlife (SPA) that has been in place since 2000. The goals are to encourage Parties “to take all appropriate measures to protect and preserve rare or fragile ecosystems, as well as the habitat of depleted, threatened or endangered species, in the Convention area.” The SPA protocol has partnered with WIDECAST to develop a program of work on sea turtle conservation, which has helped many of the Caribbean nations to identify and prioritize their conservation actions through Sea Turtle Recovery Action Plans. Each recovery action plan summarizes the known distribution of sea turtles, discusses major causes of mortality, evaluates the effectiveness of existing conservation laws, and prioritizes implementing measures for stock recovery. The objective of the recovery action plan series is not only to assist Caribbean governments in the discharge of their obligations under the SPA Protocol, but also to promote a regional capability to

implement science-based sea turtle management and conservation programs. Additional information is available at <http://www.cep.unep.org/about-cep/spaw>.

Convention on Nature Protection and Wildlife Preservation in the Western Hemisphere (Washington or Western Hemisphere Convention): Elements of the Convention include the protection of species from human-induced extinction, the establishment of protected areas, the regulation of international trade in wildlife, special measures for migratory birds and stressing the need for co-operation in scientific research and other fields are all elements of wildlife conservation. Additional information is available at <http://www.oas.org/juridico/english/treaties/c-8.html>.

Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention): This Convention's signatories include all countries along the Pacific Rim of South America from Panama to Chile. Among other resource management components, this Convention established protocol for the conservation and management of protected marine resources. Stemming from this Convention is the Commission Permanente del Pacifico Sur (CPPS) that has developed a Marine Turtle Action Plan for the Southeast Pacific that outlines a strategy for protecting and recovering marine turtles in this region. Convention for the Protection of the Natural Resources and Environment of the South Pacific Region (Noumea Convention): This Convention has been in force since 1990 and currently includes 26 Parties. The purpose of the Convention is to protect the marine environment and coastal zones of the South-East Pacific within the 200-mile area of maritime sovereignty and jurisdiction of the Parties and, beyond that area, the high seas up to a distance within which pollution of the

high seas may affect that area. Additional information is available at

*<http://www.unep.org/regionalseas/programmes/nonunep/pacific/instruments/default.asp>*.

Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention): The World Heritage Convention was signed in 1972 and, as of November 2007, 185 states were parties to the Convention. The instrument requires parties to take effective and active measures to protect and conserve habitat of threatened species of animals and plants of scientific or aesthetic value. The World Heritage Convention currently includes 31 marine sites. Additional information is available at *<http://whc.unesco.org/en/conventiontext>*.

Convention for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPFC Convention): The convention entered into force on 19 June 2004. The WCPFC Convention draws on many of the provisions of the UN Fish Stocks Agreement [UNFSA] while, at the same time, reflecting the special political, socio-economic, geographical and environmental characteristics of the western and central Pacific Ocean (WCPO) region. The WCPFC Convention seeks to address problems in the management of high seas fisheries resulting from unregulated fishing, over-capitalization, excessive fleet capacity, vessel re-flagging to escape controls, insufficiently selective gear, unreliable databases and insufficient multilateral cooperation in respect to conservation and management of highly migratory fish stocks.

Convention for the Prohibition of Fishing with Long Driftnets in the South Pacific: This regional convention, also known as the Wellington Convention, was adopted in 1989 in Wellington, New Zealand, and entered into force in 1991. The

objective of the Convention is “to restrict and prohibit the use of drift nets in the South Pacific region in order to conserve marine living resources.” Additional information is available at <http://www.mfat.govt.nz/Treaties-and-International-Law/01-Treaties-for-which-NZ-is-Depositary/0-Prohibition-of-Fishing.php>.

Eastern Pacific Leatherback Network: Also known as La Red de la Tortuga Laúd del Océano Pacífico (Laúd OPO) ([www.savepacificleatherbacks.org](http://www.savepacificleatherbacks.org)) was formed to address the critical need for regional coordination of East Pacific leatherback conservation actions necessary to track conservation priorities and progress at the population level. Led by Fauna & Flora International, this network has brought together conservationists, researchers, practitioners and government representatives from 22 institutions across nine East Pacific countries with varying priorities, capacities and historical experiences in leatherback research and conservation to contribute to shared activities, projects, and goals. Through these efforts, Laúd now has mutually-agreed upon mechanisms for sharing information and data, as well as standardized protocols for nesting beach monitoring and bycatch assessments/fishing practices.

The Eastern Tropical Pacific Marine Corridor (CMAR) is a regional and cross-border initiative for the conservation and sustainable use of the region’s marine and coastal resources. Its objective is to sustainably manage biodiversity through ecosystem based management and the development of regional intergovernmental strategies with support of non-governmental organizations and international cooperation agencies.

United Nations’ Food and Agricultural Organization (FAO) Technical Consultation on Sea Turtle-Fishery Interactions: While not a true international instrument for conservation, the 2004 FAO of the UN’s technical consultation on sea turtle-fishery

interactions was groundbreaking in that it solidified the commitment of the lead UN agency for fisheries to reduce sea turtle bycatch in marine fisheries operations.

Recommendations from the technical consultation were endorsed by the FAO Committee on Fisheries (COFI) and called for the immediate implementation by member nations and Regional Fishery Management Organizations (RFMOs) of guidelines to reduce sea turtle mortality in fishing operations, developed as part of the technical consultation. Currently, all five of the tuna RFMOs call on their members and cooperating non-members to adhere to the 2009 FAO “Guidelines to Reduce Sea Turtle Mortality in Fishing Operations,” which describes all the gear types sea turtles could interact with and the latest mitigation options. The Western and Central Pacific Fisheries Commission (<http://www.wcpfc.int>) has the most protective measures (CMM 2008-03), which follow the FAO guidelines and ensure safe handling of all captured sea turtles. Fisheries deploying purse seines, to the extent practicable, must avoid encircling sea turtles and release entangled turtles from fish aggregating devices. Longline fishermen must carry line cutters and use dehookers to release sea turtles caught on a line. Longliners must either use large circle hooks, whole finfish bait, or mitigation measures approved by the Scientific Committee and the Technical and Compliance Committee.

Inter-American Tropical Tuna Convention (IATTC) has enacted a resolution to mitigate the impact of tuna fishing vessels on sea turtles by reducing bycatch, injury, and mortality of sea turtles. The IATTC has also developed a memorandum of understanding with the IAC. For more information, see

[http://www.iattc.org/PDFFiles/Resolutions/IATTC/\\_English/C-07-03-Active\\_Sea%20turtles.pdf](http://www.iattc.org/PDFFiles/Resolutions/IATTC/_English/C-07-03-Active_Sea%20turtles.pdf).

The International Commission for the Conservation of Atlantic Tunas (ICCAT) has adopted a resolution for the reduction of sea turtle mortality (Resolution 03-11), encouraging States to submit data on sea turtle interactions, release sea turtles alive wherever possible, and conduct research on mitigation measures. It calls for implementing the FAO Guidelines for sea turtles, avoiding encirclement of sea turtles by purse seiners, safely handling and releasing sea turtles, and reporting on interactions. The Commission does not have any specific gear requirements applicable to longline fisheries. ICCAT is currently undertaking an ecological risk assessment to better understand the impact of its fisheries on sea turtle populations. For more information see <http://www.iattc.org/>. Other international fisheries organizations that may influence leatherback turtle recovery include the Southeast Atlantic Fisheries Organization (<http://www.seafo.org>) and the North Atlantic Fisheries Organization (<http://nafo.int>). These organizations regulate trawl fisheries in their respective Convention areas. Given that sea turtles can be incidentally captured in these fisheries, both organizations have sea turtle resolutions calling on their Parties to implement the FAO Guidelines on sea turtles as well as to report data on sea turtle interactions.

The Indian Ocean Tuna Commission (IOTC) is playing an increased role in turtle conservation. Resolution 05/08, superseded by Resolution 09/06 on Sea Turtles, sets out reporting requirements related to interactions with sea turtles and accordingly provides an executive summary per species for adoption at the Working Party on Ecosystem and By-catch and then subsequently at the Scientific Committee. In 2011, IOTC developed a “Sea Turtle Identification Card” to be distributed to all longliners operating in the Indian Ocean ([www.iotc.com](http://www.iotc.com)). In 2012, the Indian Ocean Tuna Commission (IOTC) began

requiring its 31 contracting Parties to report sea turtle bycatch and to use safe handling and release techniques for sea turtles on longline vessels.

Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA): Under the auspices of the Convention of Migratory Species, the IOSEA memorandum of understanding provides a mechanism for States of the Indian Ocean and South-East Asian region, as well as other concerned States, to work together to conserve and replenish depleted marine turtle populations. This collaboration is achieved through the collective implementation of an associated Conservation and Management Plan. Currently, there are 33 Signatory States. The United States became a signatory in 2001. The IOSEA has an active sub-regional group for the Western Indian Ocean, which has improved collaboration amongst sea turtle conservationists in the region. Further, the IOSEA website provides reference materials, satellite tracks, on-line reporting of compliance with the Convention, and information on all international mechanisms currently in place for the conservation of sea turtles. Finally, at the 2012 Sixth Signatory of States meeting in Bangkok, Thailand, the Signatory States agreed to procedures to establish a network of sites of importance for sea turtles in the IOSEA region (<http://www.ioseaturtles.org>).

Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC): This Convention is the only legally binding international treaty dedicated exclusively to sea turtles and sets standards for the conservation of these endangered animals and their habitats with a large emphasis on bycatch reduction. The Convention area is the Pacific and the Atlantic waters of the Americas. Currently, there are 15 Parties. The United States became a Party in 1999. The IAC has worked to adopt fisheries

bycatch resolutions, carried out workshops on Caribbean sea turtle conservation, and established collaboration with other agreements such as the Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region and the International Commission for the Conservation of Atlantic Tunas. Additional information is available at <http://www.iacseaturtle.org>.

International Convention for the Prevention of Pollution from Ships (MARPOL): The MARPOL Convention is a combination of two treaties adopted in 1973 and 1978 to prevent pollution of the marine environment by ships from operational or accidental causes. The 1973 treaty covered pollution by oil, chemicals, and harmful substances in packaged form, sewage and garbage. The 1978 MARPOL Protocol was adopted at a Conference on Tanker Safety and Pollution Prevention which included standards for tanker design and operation. The 1978 Protocol incorporated the 1973 Convention as it had not yet been in force and is known as the International Convention for the Prevention of Marine Pollution from Ships, 1973, as modified by the Protocol of 1978 relating thereto (MARPOL 73/78). The 1978 Convention went into force in 1983 (Annexes I and II). The Convention includes regulations aimed at preventing and minimizing accidental and routine operations pollution from ships. Amendments passed since have updated the convention.

International Union for Conservation of Nature (IUCN): The IUCN Species Programme assesses the conservation status of species on a global scale. This assessment provides objective, scientific information on the current status of threatened species. The IUCN Red List of Threatened Species provides taxonomic, conservation status and distribution information on plants and animals that have been globally evaluated using

the IUCN Red List Categories and Criteria. This system is designed to determine the relative risk of extinction, and the main purpose of the IUCN Red List is to catalogue and highlight those plants and animals that are facing a higher risk of global extinction (*i.e.*, those listed as Critically Endangered, Endangered and Vulnerable). Additional information is available at <http://www.iucnRed List.org/about>.

Marine Turtle Conservation Act (MTCA): The MTCA is a key element of sea turtle protection in the United States and internationally. This Act authorizes a dedicated fund to support marine turtle conservation projects in foreign countries, with emphasis on protecting nesting populations and nesting habitat. Additional information is available at <https://www.fws.gov/international/wildlife-without-borders/marine-turtle-conservation-fund.html>.

Memorandum of Agreement between the Government of the Republic of the Philippines and the Government of Malaysia on the Establishment of the Turtle Island Heritage Protected Area: Through a bilateral agreement, the Governments of the Philippines and Malaysia established The Turtle Island Heritage Protected Area (TIHPA), made up of nine islands (6 in the Philippines and 3 in Malaysia). The following priority activities were identified: management-oriented research, the establishment of a centralized database and information network, appropriate information awareness programs, a marine turtle resource management and protection program, and an appropriate ecotourism program (Bache and Frazier 2006).

Memorandum of Understanding of a Tri-National Partnership between the Government of the Republic of Indonesia, the Independent State of Papua New Guinea and the Government of Solomon Islands: This agreement promotes the conservation and

management of Western Pacific leatherback turtles at nesting sites, feeding areas and migratory routes in Indonesia, Papua New Guinea and Solomon Islands. This is done through the systematic exchange of information and data on research, population and migratory routes monitoring, nesting sites and feeding areas management activities for Western Pacific leatherback turtles and by enhancing public awareness of the importance of conserving these turtles and their critical habitats.

*[http://awsassets.wwf.or.id/downloads/mou\\_trinationalpartnershipagreement\\_clean.pdf](http://awsassets.wwf.or.id/downloads/mou_trinationalpartnershipagreement_clean.pdf)*

Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum): This MOU was concluded under the auspices of the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and became effective in 1999. The MOU area covers 26 Range States along the Atlantic coast of Africa extending approximately 14,000 km from Morocco to South Africa. The goal of this MOU is to improve the conservation status of marine turtles along the Atlantic Coast of Africa. It aims at safeguarding six marine turtle species – including the leatherback turtle – that are estimated to have rapidly declined in numbers during recent years due to excessive exploitation (both direct and incidental) and the degradation of essential habitats. This includes the protection of the life stages from hatchlings through adults with particular attention paid to the impacts of fishery bycatch and the need to include local communities in the development and implementation of conservation activities. However, despite this agreement, killing of adult turtles and harvesting of eggs remains rampant in many areas along the Atlantic African coast.

Additional information is available at

[http://www.cms.int/species/africa\\_turtle/AFRICAturtle\\_bkgd.htm](http://www.cms.int/species/africa_turtle/AFRICAturtle_bkgd.htm).

National Sea Turtle Conservation Project in India: Launched in 1998 with the aim of protecting *Lepidochelys olivacea*, but it also has conservation and protection strategies for all the other turtle species nesting in the country. This project was undertaken by the Indian government to oversee: surveys, monitoring programs, fisheries interactions, community and NGO participation, awareness raising and education, research support, and other support for regional and international co-operation and collaboration for sea turtles conservation (Choudhury *et al.*, 2001).

North American Agreement for Environmental Cooperation: As mandated by the 1994 North American Agreement for Environmental Cooperation, the Commission for Environmental Cooperation (CEC) encourages Canada, the United States, and Mexico to adopt a continental approach to the conservation of flora and fauna. In 2003, this mandate was strengthened as the three North American countries launched the Strategic Plan for North American Cooperation in the Conservation of Biodiversity. The North American Conservation Action Plan (NACAP) initiative began as an effort promoted by the three countries, through the CEC, to facilitate the conservation of marine and terrestrial species of common concern. In 2005, the CEC supported the development of a NACAP for Pacific leatherbacks by Canada, the United States, and Mexico. Identified actions in the plan addressed three main objectives: 1) protection and management of nesting beaches and females; 2) mortality reduction from bycatch throughout the Pacific Basin; and 3) waste management, control of pollution, and disposal of debris at sea.

Ramsar Convention on Wetlands: The Convention on Wetlands, signed in Ramsar, Iran, in 1971, is an intergovernmental treaty, which provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources. Currently, there are 158 parties to the convention, with 1,752 wetland sites, including important marine turtle habitat. Additional information is available at <http://www.ramsar.org>.

Secretariat of the Pacific Regional Environment Programme (SPREP): SPREP's turtle conservation program seeks to improve knowledge about sea turtles in the Pacific through an active tagging program, as well as maintaining a database to collate information about sea turtle tags in the Pacific. SPREP supports capacity building throughout the central and southwest Pacific. SPREP established an action plan for the Pacific Islands (<http://www.sprep.org/>).

South-East Atlantic Fisheries Organization (SEAFO): SEAFO manages fisheries activities in the Southeast Atlantic high seas area, excluding tunas and billfish. SEAFO adopted Resolution 01/06, "to Reduce Sea Turtle Mortality in Fishing Operations," in 2006. The Resolution requires Members to: (1) implement the FAO Guidelines; and (2) establish on-board observer programs to collect information on sea turtle interactions in SEAFO-managed fisheries. This Resolution is not legally binding. Additional information is available at <http://www.seafo.org>.

South Atlantic Association: In the southwest Atlantic, the South Atlantic Association is a multinational group that includes representatives from Brazil, Uruguay, and Argentina and meets bi-annually to share information and develop regional action plans to address threats including bycatch (<http://www.tortugasaso.org/>). At the national

level, Brazil has developed a national plan for sea turtle bycatch reduction that was initiated in 2001 (Marcovaldi *et al.* 2002). This national plan includes various activities to mitigate bycatch, including time-area restrictions of fisheries, use of bycatch reduction devices, and working with fishermen to successfully release live-captured turtles. In Uruguay, all sea turtles are protected from human impacts, including fisheries bycatch, by presidential decree (Decreto Presidencial 144/98).

United Nations Convention on the Law of the Sea (UNCLOS): To date, 155 countries, including most mainland countries lining the western Pacific, and the European Community have joined in the convention. The United States has signed the treaty and abides by some provisions, but the Senate has not ratified it. Aside from its provisions defining ocean boundaries, the convention establishes general obligations for safeguarding the marine environment through mandating sustainable fishing practices and protecting freedom of scientific research on the high seas. Additional information is available at <http://www.un.org/Depts/los/index.htm>.

United Nations' Food and Agricultural Organization (FAO): The FAO published guidelines for sea turtle protection, entitled Technical Consultation on Sea Turtle-Fishery Interactions (FAO 2005). The UN 1995 Code of Conduct for Responsible Fisheries (FAO 2004) provides guidelines for the development and implementation of national fisheries policies, including gear modification (*e.g.*, circle hooks, fish bait, deeper sets, and reduced soak time), new technologies, and management of areas where fishery and sea turtle interactions are more severe. The guidelines stress the need for mitigation measures, data on all fisheries, fishing industry involvement, and education for fishers, observers, managers, and compliance officers (FAO 2004).

United Nations Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing: In 1989, the UN called, in a unanimous resolution, for the elimination of all high seas driftnets by 1992. Additional information is available at <http://www.un.org/documents/ga/res/44/a44r225.htm>.

Although numerous conservation efforts apply to the species, they do not adequately reduce its risk of extinction for the reasons discussed previously.

#### *Extinction Risk Analysis*

The best available information is consistent with the species' current "endangered" listing. The species exhibits a global total index of nesting female abundance of 32,060 females at monitored beaches. This number is lower than historical estimates of nesting female abundance ( $n = 115,000$ , Pritchard 1982; and  $n = 34,500$ , Spotila 1996), which did not include the large, but then unknown, Gabon nesting aggregation. Limited nesting female abundance is a major source of concern for most DPSs, whose small population sizes place them in danger of stochastic or catastrophic events that increase extinction risk. The limited nesting female abundance increases the extinction risk of the species.

The species also exhibits declining nesting trends for all but one of the DPSs. With the exception of the DPS with the smallest index of nesting female abundance (*i.e.*, SW Atlantic DPS, with 27 nesting females), the DPSs are declining at rates of 0.3 to 9.3 percent annually. Even low levels of decline are a threat for DPSs with limited nesting female abundance, and nesting declines of approximately nine percent (*i.e.*, NW and SE Atlantic DPSs) are unsustainable. Total declines of 97 and 99 percent have occurred within the East Pacific and NE Indian DPSs, respectively, since nesting was first

identified and quantified for these populations. The declining trends in nesting increase the extinction risk of the species.

The species exhibits broad nesting and foraging ranges. However, metapopulation dynamics have likely been reduced, with reductions in abundance and the loss of some nesting aggregations. The species also demonstrates little genetic diversity, relative to other sea turtle species. Although the species demonstrates some resilience to threats, overall we find it to be at risk of extinction, due to limited abundance and declining nesting trends, which reflect the cumulative impacts of threats that have acted on the species in the past (and in many cases continue to act on the species).

Current threats continue to place the species in danger of extinction. The primary threat to the species is bycatch in commercial and artisanal, pelagic and coastal, fisheries. Fisheries bycatch reduces abundance by removing individuals from the population. Because several fisheries operate near nesting beaches, productivity is also reduced when nesting females are prevented from returning to nesting beaches. The harvest of eggs and turtles is also a major threat to the species. Illegal poaching occurs throughout the range of the species, and harvest is legal but poorly documented in some nations. The loss and modification of nesting habitat is another major threat, reducing productivity and, in some instances, abundance, when nesting females die as a result of artificial lighting or obstructions preventing them from returning to sea. Predation results in the loss of eggs and hatchlings, reducing productivity of the species. Additional threats that occur throughout the range of the species include vessel strikes, pollution, marine debris, oil and gas exploration, and climate change. Natural disasters and oceanographic regime

shifts are threats in some areas. Though many regulatory mechanisms are in place, they do not adequately reduce the impact of these threats.

Based on our review of the best available scientific and commercial data, we find nothing that is inconsistent with the leatherback species' current listing as an endangered species. In sum, the best available information is consistent with the current listing status of the leatherback sea turtle as an endangered species throughout its range. The threatened species definition does not apply because the species is currently in danger of extinction (*i.e.*, at present), rather than on a trajectory to become so within the foreseeable future.

### **Final Determination**

The Services determined that the best available scientific and commercial information would support recognizing seven populations as DPSs (including the NW Atlantic) because they meet the discreteness and significance criteria for DPSs. However, we found that – even were they to be recognized and listed separately -- all DPSs meet the definition of an endangered species because they are in danger of extinction throughout all of their ranges. The leatherback turtle is currently listed throughout its range as an endangered species. Replacing this listing with seven endangered DPSs would not be consistent with Congressional guidance to use the authority to list DPSs “sparingly” while encouraging the conservation of genetic diversity (see Senate Report 151, 96th Congress, 1st Session). Such guidance clearly indicates that the Services have some discretion to determine whether or not to recognize DPSs that would require disaggregating an existing listing even where those populations can be shown to meet the discreteness and significance tests of the DPS Policy.

After determining that all seven populations would have the same status as the overall species, we next considered whether there was any reason to nevertheless replace the global (range-wide) listing with individual listings for the seven DPSs. We conclude that disaggregating the global listing is not warranted. It would be inconsistent with Congressional guidance and run counter to the conservation purposes of the Act to disaggregate the current listing into DPSs, because those DPSs would have the same listing status as the whole currently. Disaggregating this listing would bring about significant complications and possible public confusion without any meaningful corresponding conservation benefit. Replacing the range-wide listing with seven DPSs having the same status would not provide leatherback turtles with an overriding conservation benefit, as all members are currently protected to the fullest extent under the ESA as an endangered species. Section 7 consultations already consider the effects of an action on individuals and populations to determine whether a Federal agency has insured that its action is not likely to jeopardize the continued existence of the species. Even if the species were disaggregated into DPSs, this change would not be expected to result in different substantive outcomes in consultations. In addition, focused conservation efforts have been, and will continue to be, applied at scales smaller than the species-level. For example, FWS' Marine Turtle Conservation Fund provides funding to partners in foreign nations to protect leatherback turtles and their nesting habitats; projects include efforts to monitor and protect leatherback turtles in Indonesia and Gabon (<https://www.fws.gov/international/wildlife-without-borders/marine-turtle-conservation-fund.html>). Similarly, Pacific leatherback turtles are highlighted under NMFS' Species in the Spotlight: Survive to Thrive initiative, which directs attention and resources to

highly-at-risk species (<https://www.fisheries.noaa.gov/topic/endangered-species-conservation#species-in-the-spotlight>).

For these reasons, the Services have determined that replacing the existing global listing with separate listings for individual DPSs is not warranted. Although the best available data indicates that the populations meet the criteria for significance and discreteness, we find that it would not further the purposes of the Act to recognize and list seven DPSs separately as endangered under the ESA. The current global listing of the species remains in effect.

We conclude that the petitioned actions, to identify the NW Atlantic population as a DPS and list it as a threatened species under the ESA, are not warranted. This is a final action, and, therefore, we are not soliciting public comments.

### **Peer Review**

In December 2004, the Office of Management and Budget (OMB) issued a Final Information Quality Bulletin for Peer Review, establishing minimum peer review standards, a transparent process for public disclosure of peer review planning, and opportunities for public participation. The OMB Bulletin, implemented under the Information Quality Act (Pub. L. 106-554), is intended to enhance the quality and credibility of the Federal government's scientific information and applies to influential or highly influential scientific information disseminated on or after June 16, 2005. To satisfy our requirements under the OMB Bulletin, we obtained independent peer review of the Status Review Report by independent scientists with expertise in leatherback turtle biology, endangered species listing policy, and related fields. All peer reviewer

comments were addressed prior to the publication of the Status Review Report and this finding.

### **References Cited**

A complete list of references is available upon request to the NMFS Office of Protected Resources (see **ADDRESSES**).

### **Authority**

The authority for this action is the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq.*).

---

Samuel D. Rauch III,  
Deputy Assistant Administrator for Regulatory Programs,  
National Marine Fisheries Service.

---

Aurelia Skipwith  
Director, U.S. Fish and Wildlife Service.