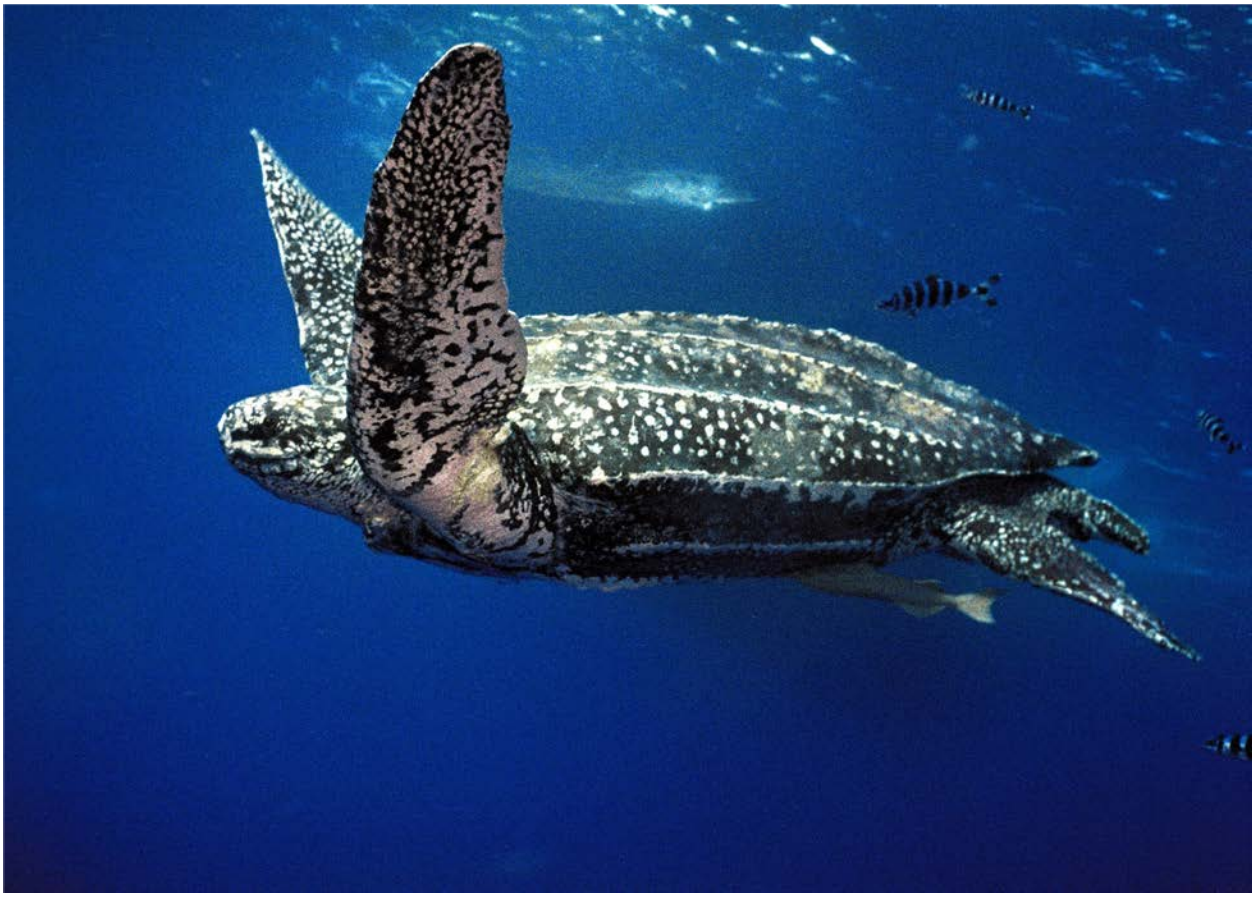
Endangered Species Act
Status Review of the Leatherback Turtle
(*Dermochelys coriacea*)
2020



National Marine Fisheries Service
National Oceanic and Atmospheric Administration
U.S. Department of Commerce

U.S. Fish and Wildlife Service

U.S. Department of the Interior

Exhibit 8 to Declaration of Adam Keats

Disclaimer

The purpose of this document is to synthesize and review the best available scientific and commercial data on the leatherback turtle (*Dermochelys coriacea*), assess the discreteness and significance of populations as outlined in the Policy Regarding the Recognition of Distinct Vertebrate Population Segments under the Endangered Species Act (61 FR 4722, February 7, 1996), and evaluate the extinction risk of any such populations. This document does not represent a decision by the National Marine Fisheries Service and the U.S. Fish and Wildlife Service, together the Services, on the identification of distinct population segment(s) and whether any should be proposed for listing as threatened or endangered under the Endangered Species Act of 1973, as amended (16 U.S.C. 1531 *et seq*). Such decisions will be made by the Services after reviewing this document, efforts being made to protect the species, and all relevant laws, regulations, and policies. The results of those decisions will be published in the *Federal Register*.

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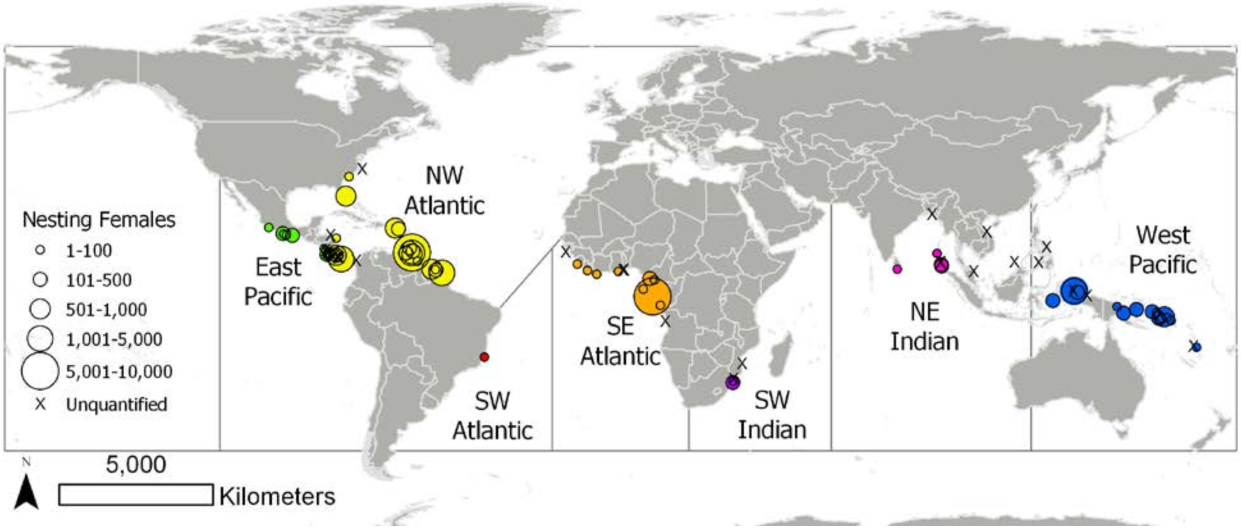
Executive Summary

The leatherback turtle (*Dermochelys coriacea*) is listed as endangered under the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1531 *et seq*). The National Marine Fisheries Service (NMFS) and U.S. Fish and Wildlife Service (USFWS), together the Services, jointly administer the ESA and share jurisdiction of sea turtles, as described in the 2015 Memorandum of Understanding Defining the Roles of the USFWS and NMFS in Joint Administration of the ESA as to Sea Turtles. In the 5-year review of the species (NMFS and USFWS 2013), the Services recommended conducting a status review to apply the Policy Regarding the Recognition of Distinct Vertebrate Population Segments (DPS) under the ESA (i.e., DPS Policy; 61 FR 4722; February 7, 1996). On September 20, 2017, the Blue Water Fishermen's Association petitioned 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) and announced that it would commence, jointly with USFWS, a status review of the species, pursuant to ESA section 4(b)(3)(A) and 50 CFR 424.14. The Services convened a Status Review Team (i.e., the Team or we) to gather and review the best available scientific and commercial data on the species, apply the DPS Policy, and evaluate the extinction risk of any potential DPSs, following the NMFS' Listing Guidance (Conducting a Status Review of NMFS' Guidance on Responding to Petitions and Conducting Status Reviews under the Endangered Species Act; November 9, 2017).

To apply the DPS Policy, the Team assessed the discreteness and significance of leatherback populations. In our analysis of discreteness, we found genetic discontinuity among seven populations. Tagging data demonstrate the marked separation of the seven populations at nesting beaches, with no definitive evidence for nesting females moving among populations. Satellite telemetry data also demonstrate marked separation of these populations at nesting beaches; however, some populations overlap in foraging areas. Because females and males return to the waters off their natal nesting beaches to mate, overlap at foraging areas does not result in gene flow among populations, as demonstrated by genetic analyses. The seven populations are markedly separated by physical factors, including: land masses, oceanographic features, and currents. We did not find evidence for morphological discontinuity among any populations. We concluded that seven leatherback populations are discrete, i.e., markedly separated from each other as a result of physical and behavioral factors. We assessed the significance of each discrete population. We found that the loss of any population would result in a significant gap (i.e., a half or quarter of an ocean basin) in the nesting, and sometimes foraging, range of the species. Further, each population exhibits unique genetic characteristics, and some populations occur in a unique ecological setting. Thus, after reviewing the best available information, we identified seven leatherback populations that meet the discreteness and significance criteria of the DPS Policy (Figure i):

- Northwest (NW) Atlantic DPS
- Southwest (SW) Atlantic DPS
- Southeast (SE) Atlantic DPS
- Southwest (SW) Indian DPS
- Northeast (NE) Indian DPS
- West Pacific DPS
- East Pacific DPS

Figure i. Map of DPS boundaries and nesting beaches. Grey lines represent DPS boundaries. Circles mark the location of nesting beaches, identified by a different color for each DPS. The size of the circle reflects the index of nesting female abundance, calculated over a remigration interval. An "x" marks the location of nesting beaches with unquantified nesting female abundance and a functionally extinct nesting aggregation in Malaysia.



For each potential DPS, we evaluated the extinction risk by assessing demographic factors (abundance, productivity, spatial distribution, and diversity) and threats. We calculated an index of nesting female abundance, defined as the estimated number of nesting females over one remigration interval, at beaches where recent, consistent, and standardized nest monitoring data were available (Figure i). When sufficient data were available, we evaluated nest trends. For each DPS, we reviewed the best available information to evaluate threats (i.e., ESA section 4(a)(1) factors). We found that all DPSs have been and are impacted, to varying degrees, by habitat loss and modification, overutilization, predation, inadequate regulatory mechanisms, fisheries bycatch, pollution, and climate change. For all DPSs, fisheries bycatch is a primary threat. Historically, the harvest of turtles and eggs (i.e., the direct killing of turtles and collection of eggs for consumption or other human use), was the primary threat that led to drastic declines in the species. In many locations, the harvest of eggs and turtles continues, though illegal in many nations. We found that all DPSs met the definition of high risk of extinction, per NMFS' Listing Guidance, as a result of reduced nesting female abundance, declining nest trends (for all but the SW Atlantic DPS, which exhibits extremely low abundance), and numerous, severe threats. We had high confidence in our conclusion of high extinction risk for all DPSs, with the exception of the NW and SE Atlantic DPSs, for which we had moderate confidence. The NW Atlantic DPS faces clear and present threats that, along with a declining nest trend, which has accelerated in recent years, place its continued persistence in question. Though this meets the definition of high extinction risk, our confidence in this conclusion is moderate because of its abundance, spatial distribution, and diversity. The SE Atlantic DPS also faces clear and present threats that, along with a declining nest trend, place its continued persistence in question; however, data outside of Gabon are limited, reducing our ability to quantify threats for more than a small portion of the population.

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Abbreviations and Acronyms

BSSM	Bayesian state-space model
BPUE	Bycatch per unit effort
CCL	Curved carapace length
CFR	Code of Federal Regulations
CI	Credible Interval
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CONANP	La Comisión Nacional de Áreas Naturales Protegidas
CPUE	Catch per unit effort
CTV	Centro Terra Viva Estudos e Advocacia Ambiental, Mozambique
cm	Centimeter
DFO	Fisheries and Oceans Canada
DPS	Distinct population segment
E	East
EEZ	Exclusive economic zone
ENSO	El Niño–Southern Oscillation
ESA	Endangered Species Act
FAO	Food and Agriculture Organization
FFI	Fauna and Flora International
FR	Federal Register
F_{ST}	Fixation index of subpopulation relative to total population
FWC	Florida Fish and Wildlife Conservation Commission
GAMM	Generalized additive mixed models
GOM	Gulf of Mexico
GWD	Ghana Wildlife Division
IAC	Inter-American Convention for the Protection and Conservation of Sea Turtles
IATTC	Inter-American Tropical Tuna Commission
IOSEA	Indian Ocean – South-East Asian Marine Turtles
IOTC	Indian Ocean Tuna Commission
IPCC	Intergovernmental Panel of Climate Change
IUCN	International Union for Conservation of Nature
IUU	Illegal, unreported and unregulated fishing
km	Kilometer
LAST	Latin American Sea Turtles
m	Meter
MOU	Memorandum of Understanding
MRF	Marine Research Foundation
MWWP	Marine Wildlife Watch of the Philippines
MSA	Magnuson–Stevens Fishery Conservation and Management Act
mtDNA	Mitochondrial deoxyribonucleic acid
n	Number of samples
N	North
NE	Northeast
NMFS	National Marine Fisheries Service

NRC	National Resource Council
NW	Northwest
PBDE	Polybrominated diphenyl ethers
PCB	Polychlorinated biphenyls
PIT	Passive integrated transponder
POP	Persistent organic pollutant
RFMO	Regional Fishery Management Organizations
RMU	Regional Management Unit
S	South
SCL	Straight carapace length
sd	Standard deviation
SE	Southeast
SEFSC	Southeast Fisheries Science Center
SICCP	Solomon Islands Community Conservation Partnership
SST	Sea surface temperature
STC	Sea Turtle Conservancy
STRAP	Sea Turtle Recovery Action Plan
STDN	Sea Turtle Disentanglement Network
STSSN	Sea Turtle Stranding and Salvage Network
SW	Southwest
SWFSC	Southwest Fisheries Science Center
SWOT	State of the World’s Sea Turtles
TDA	Tetepare Descendants' Association
TED	Turtle excluder device
TEWG	Turtle Expert Working Group
U.S.C.	United States Code of Laws
USD	United States dollars
USFWS	United States Fish and Wildlife Service
USVI	US Virgin Islands
UNIPA	State University of Papua
W	West
WCPFC	Western and Central Pacific Fisheries Commission
WCS	Wildlife Conservation Society
WIDECAST	Wider Caribbean Sea Turtle Conservation Network
WWF	World Wildlife Fund

1. Introduction

On June 2, 1970, the leatherback turtle (*Dermochelys coriacea*) was listed as endangered under the Endangered Species Conservation Act of 1969 (35 FR 8491), the precursor to the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. 1531 *et seq*). When the ESA was enacted in 1973, the species was listed as endangered, wherever found (see Endangered and Threatened Wildlife; 50 CFR 17.11). The purpose of the ESA and its implementing regulations (50 CFR 424) is to provide a means to conserve the ecosystems upon which endangered and threatened species depend, to provide a program for the conservation of such species, and to take such steps as may be appropriate to achieve the purposes of treaties and conventions to conserve fish, wildlife, and plants facing extinction. Under the ESA, the term "species" includes any subspecies of fish or wildlife or plants, and any distinct population segment (DPS) of any species of vertebrate fish or wildlife which interbreeds when mature (16 U.S.C. 1532(16)). The ESA defines an endangered species as any species which is in danger of extinction throughout all or a significant portion of its range. It defines a threatened species as any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.

The National Marine Fisheries Service (NMFS) and U.S. Fish and Wildlife Service (USFWS), together the Services, jointly administer the ESA and share jurisdiction of sea turtles. Their responsibilities are defined in the 2015 Memorandum of Understanding Defining the Roles of the USFWS and NMFS in Joint Administration of the ESA as to Sea Turtles. This Memorandum supersedes the 1977 Memorandum of Understanding Defining the Roles of the USFWS and NMFS in Joint Administration of the ESA as to Marine Turtles. The 2015 Memorandum essentially establishes that NMFS has jurisdiction for sea turtles in the marine environment, and USFWS has jurisdiction for sea turtles in the terrestrial environment and for all imports and exports of sea turtles and their parts and products.

On November 11, 2013, the Services published the Leatherback Sea Turtle (*Dermochelys coriacea*) 5-Year Review: Summary and Evaluation (i.e., the 5-Year Review; NMFS and USFWS 2013), as required under Section 4(c)(2) of the ESA. The 5-Year Review revealed that a substantial amount of information on the genetics and movement of the species had become available since its listing under the ESA. The 5-Year Review recommended conducting a status review to apply the Policy Regarding the Recognition of Distinct Vertebrate Population Segments under the ESA (i.e., DPS Policy; 61 FR 4722; February 7, 1996).

On September 20, 2017, the Blue Water Fishermen's Association petitioned 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) with 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 species, pursuant to ESA section 4(b)(3)(A) and 50 CFR 424.14.

To address the recommendation of the 5-Year Review and the petitioned action, the Services commenced a status review of the species to provide the best scientific and commercial data available, as required by the ESA (16 U.S.C. 1533(b)(1)(A)).

1.1 Approach to the Status Review

The Services convened a Status Review Team (i.e., the Team or we) to gather and review the best available data on the leatherback turtle, assess the discreteness and significance of populations by applying the DPS Policy, evaluate the extinction risk of any populations that meet the DPS criteria, and document all findings in a status review report (i.e., the Report).

The Services invited Federal biologists from NMFS and USFWS to participate on the Team based on their knowledge of leatherback turtle ecology, population dynamics, modeling, and genetics, or experience with the status review process under the ESA. To provide leatherback turtle expertise not otherwise available within the Team, we invited a non-Federal NMFS biologist to participate informally as an observer in meetings and calls and to help with gathering information for the status review.

To conduct the status review, we followed the directives provided in the ESA, implementing regulations, policies, and internal guidance. Internal guidance included "Conducting a Status Review of NMFS' Guidance on Responding to Petitions and Conducting Status Reviews under the Endangered Species Act, Section II" (i.e., NMFS' Guidance; November 9, 2017) and the "Leatherback Turtle Status Review Team Terms of Reference" (March 9, 2018). First, we applied the criteria outlined in the DPS Policy. Then, for any population which met the DPS criteria, we performed a qualitative analysis of its extinction risk. The ESA requires the Services to determine whether any species is endangered or threatened because of any of the ESA section 4(a)(1) factors (16 U.S.C. 1533(a)(1), 50 CFR 424.11(c)): the present or threatened destruction, modification, or curtailment of habitat or range; overutilization for commercial, recreational, scientific, or educational purposes; disease or predation; inadequacy of existing regulatory mechanisms; or any other natural or manmade factors affecting the species' existence.

The NMFS' Guidance states that the Team " will not use the statutory terms 'endangered' or 'threatened' or their definitions when making conclusions regarding the species' risk of extinction." Instead, the NMFS' Guidance requires a qualitative assessment of extinction risk (i.e., high, moderate, or low, as defined in a later section) based on the ESA Section 4(a)(1) factors (or threats) and how those threats impact demographic factors (i.e., abundance, productivity, spatial distribution, and diversity).

In the following paragraphs, we summarize our status review process. For additional details, please refer to the ESA, implementing regulations, policies, and NMFS' Guidance.

1.1.1 Best Available Information

To gather and review the best available scientific and commercial data, we compiled information on leatherback turtle life history, biology, ecology, demographic factors, and threats. This included the information that we received in the petition and in response to our request in the *Federal Register* (82 FR 57565; December 6, 2017). We identified relevant information in scientific literature, government reports, and bycatch reports. We reviewed all compiled information for content, relevance, and robustness. Generally, we gave the most weight to peerreviewed information, specifically to primary research with large sample sizes and long-term sampling duration. In some locations, reports from governments or non-governmental organizations and expert opinion constituted the best available information. To evaluate recent abundance and trends, unpublished nest monitoring datasets often provided the best available data (i.e., most recent and relevant). We 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). We also addressed the source and magnitude of any uncertainty and its impact on our conclusions.

1.1.2 DPS Analysis

In the DPS Policy, the Services clarify their interpretation of the phrase "distinct population segment" as one that is discrete in relation to the remainder of the species to which it belongs and significant to the species to which it belongs (61 FR 4722; February 7, 1996). The DPS Policy states that a population segment may be considered discrete if 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
- Delimited by international governmental boundaries within which significant differences exist in regards to control of exploitation, management of habitat, conservation status, or regulatory mechanisms.

The DPS Policy states that a population segment may be considered biologically or ecologically significant if it is important to the species; this consideration may include, but is not limited to:

1. Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon;
2. Evidence that loss of the discrete population segment would result in a significant gap in the range of a taxon;
3. Evidence that the discrete population segment 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 discrete population segment differs markedly from other populations of the species in its genetic characteristics.

The DPS Policy reiterates Congressional guidance, which states that the authority to list DPSs be used "sparingly," while encouraging the conservation of genetic diversity (see Senate Report 151, 96th Congress, 1st Session; 61 FR 4722; February 7, 1996).

After gathering the best available information, the Team met in person to perform the DPS analysis. Following the DPS Policy, we evaluated whether leatherback populations met the discreteness and significance criteria, in a process similar to the one we used for the loggerhead and green turtle status reviews. To evaluate discreteness, we considered genetic discontinuity, behavioral separation (based on satellite tracking studies and mark/recapture data), physical separation (such as large land masses, inhospitable temperatures, or oceanographic barriers), or morphological discontinuity (such as growth and maturity rates) that may reflect physiological or ecological differences. After finding a population to be discrete from other populations, we considered whether it was significant to the species. We considered a population to be significant if it was biologically or ecologically important to the species. At the end of our in-person meeting, we delineated boundaries for the populations that met the discreteness and significance criteria of the DPS Policy. We based these boundaries on the best available scientific information. When limited data were available, we used nearby political boundaries or significant meridians for ease of application. At this point in the process, we started referring to the populations that met the DPS criteria as "potential DPSs," or "DPSs" for ease of reference. We stress that the populations identified within this Report are not DPSs under the ESA unless the Services formally recognize them as such by listing them as threatened or endangered species via the publication of proposed and final rules in the *Federal Register* and inclusion in the *Code of Federal Regulations*.

1.1.3 Extinction Risk Analysis

After gathering the best available scientific information on each DPS, the Team met again to perform the extinction risk analysis. For each potential DPS, we evaluated its extinction risk as follows. We first described and evaluated demographic factors. Then, we identified and characterized all ESA Section 4(a)(1) factors (i.e., threats) that affect the DPS. We evaluated how the threats impact the demographic factors and whether they contribute to the extinction risk of the DPS. We identified and evaluated any uncertainty. Finally, we performed the extinction risk assessment, determining whether each DPS was at high, moderate, or low extinction risk. We describe these steps in detail in the following paragraphs.

1.1.3.1 Demographic Factors

The persistence of a species is influenced by its demographic factors, that when reduced, increase its risk of extinction (Wainwright and Kope 1999; McElhany *et al.* 2000). Following fundamental concepts in the fields of population and conservation biology, as described in the NMFS' Guidance, we evaluated the following demographic factors:

- Abundance
- Productivity
- Spatial distribution
- Diversity

Abundance is of fundamental importance because small populations are more susceptible to extinction than larger ones (McElhany *et al.* 2000). 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. Instead, the majority of sea turtle population monitoring occurs on nesting beaches, where females or nests are counted (NRC 2010). Generally, the best available data are the number of leatherback nests counted on a beach during a nesting season. However, females nest more than once in a season (i.e., clutch frequency, which is the average number of nests per female per season) and do not nest every season (i.e., remigration interval, which is the average number of years between successive nesting seasons). To provide an index of nesting female abundance, we summed the total number of nests over the most recent remigration interval (i.e., the sum of nests over 3 or 4 years) and divided this number by the clutch frequency. For example, for a nesting beach with a clutch frequency of five and a remigration interval of 3 years, the index of nesting female abundance would be calculated as follows:

$$Index\ of\ nesting\ female\ abundance = \frac{Nests\ in\ 2015 + Nests\ in\ 2016 + Nests\ in\ 2017}{5}$$

We identify our measure of nesting female abundance as an index (rather than a census) because it is based on the above calculation rather than a direct count of all nesting females, which is not available for most nesting beaches. When available, we used the "local" estimates for remigration interval and clutch frequency (based on data from that DPS); otherwise, we used the species' averages (i.e., a remigration interval of 3 years and a clutch frequency of five clutches per season; Eckert *et al.* 2012). Errors or biases in breeding periodicity, probability of detection, and annual survival probability may overestimate the abundance of sea turtle populations (Casale and Ceriani 2020); therefore, our index is a minimum estimate of nesting female abundance.

To calculate the index of nesting female abundance, we only included beach monitoring data that met our criteria, as follows:

- Recent: the last year of the remigration interval occurred in 2014 or more recently
- Standardized: data collection methods remained the same over the remigration interval
- Consistent: data were collected over the entire remigration interval with no missing nesting seasons

We required recent data because, at many beaches, nesting female abundance has changed in recent years, and older data do not necessarily reflect current abundance. We required standardized and consistent data because the use of non-standardized or inconsistent data would introduce unnecessary error into our index (*https://www.seaturtlestatus.org/minimum-data-standards*). For data that did not meet our criteria, we categorized the index of nesting female abundance as unquantified.

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.

$$Total\ index\ of\ nesting\ female\ abundance = \sum_{i=1}^{n} (Index\ of\ nesting\ female\ abundance)_i$$

The degree to which our total index represents the actual abundance of nesting females for a DPS is unknown. We recognize that, as with all field studies, observational error is unavoidable (e.g., nests are missed; Ernest and Martin 1993; Schroeder 1994). We applied the above criteria and used the best available data to provide confidence in our estimate. We also identified the level of confidence (high, moderate, or low) we had in the total index for each DPS, given the availability of recent data and the extent to which those data are representative of the DPS. Applying the above criteria, we were able to achieve a high or moderate degree of confidence in our total index of nesting female abundance. Our total indices do not include inconsistently or opportunistically monitored nesting aggregations, which likely host few or ephemeral nesting females. Our total indices generally include the largest nesting aggregations, which are more likely to be monitored. Thus, the total index of nesting female abundance is a suitable representation of total nesting female abundance for each DPS. We also considered published data on current and/or historical abundance to provide context for our index.

While the use of nesting data to estimate abundance and trends is a standard, accepted method for sea turtle status reviews (e.g., Conant *et al.* 2009; Seminoff *et al.* 2015), it is not without caveats. The number of females nesting each year is influenced by many factors. For example, remigration intervals and clutch frequency vary by individual and are influenced by prey availability and the energetic costs of migration (Saba *et al.* 2015). These productivity metrics are based on an average of individuals over time that is expected to capture such variance; however, such averages may not capture recent, large-scale shifts in remigration interval or clutch frequency due to changes in environmental productivity (Saba *et al.* 2015). We acknowledged such uncertainties when identifying our level of confidence in each total index of nesting female abundance. It is also important to note that abundance estimates based on nesting are a snapshot in time. The proportion of nesting females relative to the total population size may vary over time. Thus, conservation efforts that positively affect their progeny may go undetected for years or decades (until the female progeny nest). Alternatively, threats leading to poor recruitment may not become apparent for years or decades and cannot be predicted based on current nesting data. Ideally, a status review would assess abundance estimates and demographic parameters of all life stages so that we could detect potential changes in recruitment (NRC 2010); however, such in-water data are not available. Despite such caveats and data gaps, nest counts or nesting females are the best available data to evaluate abundance. Therefore, we used our total index of nesting female abundance to evaluate whether the demographic factor of abundance placed the DPS at risk of extinction (NMFS 2017a).

Per the NMFS' Guidance (2017a), we evaluated abundance and productivity. Though these factors are related, we evaluated them independently to consider their contribution to extinction risk. We evaluated the productivity for each DPS by evaluating nest trends (through trend analyses or bar graphs) and productivity metrics. We estimated the long-term trend for individual beaches using a Bayesian state-space model of stochastic exponential population growth (i.e., BSSM trend analysis; 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 method allows for specification of both a biological process model (true number of annual nests) and an observation model (imperfect observation of the true number of annual nests), which allows the sources of variability in the annual nest count data to be parsed out and estimated separately. Consequently, the estimate for the long-term trend parameter is better isolated from noise in the data, and the variance surrounding its mean/median more accurately captures the variation in the trend itself without being conflated with data collection errors and natural variability (environmental and demographic). As the BSSM trend analysis produces posterior probability distributions for estimated parameters based on data, we report results as the median and 95 percent credible intervals (CI) of the posterior distributions. These CI are often wide, reflecting uncertainty in the modeled trend due to inter-annual variability in the observed counts and/or the small number of data points (see below for our criteria for trend analyses). Given that our CI often include both positive and negative trends for a given beach, we also report an "f statistic," which is the proportion of the posterior distribution with the same sign as the mean (the confidence that the trend is positive or negative). In our analyses, the mean was always the same sign as the median. For each trend, we provide a figure showing the original data, the model fit (predicted annual values based on the estimated long-term trend parameter and process variability parameter), and the estimated long-term trend parameter without the estimated inter-annual variability (see insets).

Our criteria for trend analyses were as follows: nesting data consistently collected over 9 or more years in a consistent and standardized manner (for that site), with the most recent data collection in 2014 or more recently, and with a minimum average number of nests of 50 annually. We applied these criteria to provide confidence in our results and to ensure that they were based on the best available data. With the exception of nesting data from Brazil (which hosts less than 30 nesting females total), we required a minimum average of 50 nests annually to allow us to differentiate trends at small nesting aggregations from variability (i.e., noise) within a dataset stemming from data collection errors and natural processes (environmental and demographic). We required recent data so that the analysis would reflect the current (not previous) trend of the nesting beach. Trend analyses require multiple data points, and given the species' average remigration interval of 3 years, we set our criteria at 9 years to capture at least three remigration intervals. We allowed up to three consecutive missing data points because our BSSM trend analysis method was robust to three missing years if a minimum of 9 years remained. Standardization of the collection method, however, was essential so that we did not misinterpret a lack of effort as reduced nesting. We stated when data did not meet our criteria and in those cases provided a bar graph instead of a trend analysis. If there were major concerns about consistency and standardization of data collection (other than missing years of data), we acknowledged the dataset but did not include a bar graph. We described the nest trend for each DPS as decreasing, stable, or increasing. We also provided 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; and sex ratio. Each of these metrics contributes to the growth rate, or reproductive potential, of the population.

For each DPS, we evaluated spatial distribution, and we considered the number and location of nesting beaches and foraging areas. We also considered spatial structure (i.e., whether the DPS exists as a single population or several subpopulations connected by metapopulation dynamics). To evaluate spatial structure and connectivity, we considered fine scale genetic data and studies of individual movement (i.e., telemetry and tagging data). Multiple, widely distributed nesting beaches and foraging areas that are connected by some gene flow and individual movement are indicative of robust, functional population structure. Alternatively, stochastic events are more likely to be catastrophic for a DPS with limited spatial distribution and structure.

For each DPS, we evaluated diversity, which, like spatial distribution, is a measure of resilience. In general, diverse populations are more resilient to threats and environmental changes than less diverse populations. When available, we reviewed data on the genetic diversity of each DPS. We reviewed the diversity of nesting beaches and foraging areas. Where relevant, we also reviewed temporal diversity in nesting (i.e. seasonal distribution and duration of nesting). We considered how diversity would protect a DPS from small-scale, catastrophic threats and whether a DPS was resilient to continued threats and new perturbations.

1.1.3.2 ESA Section 4(a)(1) Factors or "Threats"

For each DPS, we evaluated the following ESA Section 4(a)(1) factors (i.e., threats):

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, such as: fisheries bycatch; vessel interaction; pollution (including contaminants, oil and gas activities, marine debris and plastics, and ghost fishing gear); climate change; and oceanographic regime shifts.

For each threat, we reviewed the best available information to identify whether it affects the DPS. If so, we used the best available information to describe the nature of the threat. We evaluated each threat for exposure and impact. For exposure, we identified the life stage affected by that threat. For impact, we described the link between the threat and the demographic factor that it impacted (e.g., poaching of nesting females is likely to reduce abundance and productivity). We identified the primary threat(s) to the continued existence of each DPS. We also identified any sources of uncertainty and the impact of uncertainty on our conclusions. For the threat analyses, we focused on current and emerging threats, as past threats have already been manifested in the demographic factors described above. However, our knowledge of past threats often informed our consideration of the future response to current threats (e.g., bycatch mortality leads to significant decreases in abundance over time). Following NMFS' Guidance, we analyzed all threats assuming the DPS had no ESA protections (e.g., Section 7 consultations, Section 9 take prohibitions, etc.) because a DPS would not receive such protections if it was not listed under the ESA.

1.1.3.3 Extinction Risk Assessment

To determine the extinction risk, we reviewed and evaluated all demographic factors and threats, as described above. Based on the best available information, we then determined whether the extinction risk of the DPS was high, moderate, or low, as defined by NMFS' Guidance:

- High risk: A species, subspecies, or DPS with a high risk of extinction is at or near a level of abundance, productivity, spatial structure, and/or diversity that places its continued persistence in question. The demographics of a species or DPS at such a high level of risk may be highly uncertain and strongly influenced by stochastic or depensatory processes. Similarly, a species or DPS may be at high risk of extinction if it faces clear and present threats (e.g., confinement to a small geographic area; imminent destruction, modification, or curtailment of its habitat; or disease epidemic) that are likely to create imminent and substantial demographic risks.
- Moderate risk: A species, subspecies, or DPS is at moderate risk of extinction if it is on a trajectory that puts it at a high level of extinction risk in the foreseeable future (see description of "High risk" above). A species or DPS may be at moderate risk of extinction due to current and/or projected threats or declining trends in abundance, productivity, spatial structure, or diversity. The appropriate time horizon for evaluating whether a species or DPS is more likely than not to be at high risk in the foreseeable future depends on various case- and species-specific factors. For example, the time horizon may reflect certain life history characteristics (e.g., long generation time or late age-at-maturity) and may also reflect the time frame or rate over which identified threats are likely to impact the biological status of the species or DPS (e.g., the rate of disease

spread). The appropriate time horizon is not limited to the period that status can be quantitatively modeled or predicted within predetermined limits of statistical confidence. The biologist (or Team) should, to the extent possible, clearly specify the time horizon over which it has confidence in evaluating moderate risk.)

- Low risk: A species, subspecies, or DPS is at low risk of extinction if it is not at moderate or high level of extinction risk (see "Moderate risk" and "High risk" above). A species or DPS may be at low risk of extinction if it is not facing threats that result in declining trends in abundance, productivity, spatial structure, or diversity. A species or DPS at low risk of extinction is likely to show stable or increasing trends in abundance and productivity with connected, diverse populations.

After reaching a conclusion on extinction risk, we discussed our confidence level (high, moderate, or low) for our determination. We based our confidence on the availability and robustness of the data on the demographic factors and threats. We identified any uncertainty in data or trends. The Team reached consensus on most conclusions and confidence levels; when we did not reach consensus, we voted and reported our results in this Report. As described in NMFS' Guidance, these conclusions (i.e., high, moderate, or low extinction risk) do not directly translate to a listing decision. Such decisions will be made by the Services after reviewing this document, efforts being made to protect the DPS, and all relevant laws, regulations, and policies. The results of those decisions will be published in the *Federal Register*.

1.1.4 Status Review Report

This Report documents the status review process and conclusions of the Team's calls, meetings, and deliberations. Throughout the Report, we cite the information that we used to apply the DPS Policy and to evaluate the extinction risk of potential DPSs. This information represents the best available data regarding the species. The remainder of this document is organized as follows. After reviewing general information about the species, we summarize our DPS analysis. Next, we provide information on the demographic factors and threats that pertains to all DPSs. Finally, for each DPS, we present the best available information on demographic factors and threats and perform an extinction risk analysis. In an appendix, we provide information on conservation efforts for use by the Services when making listing their determinations.

2. Background Information

Here we provide a brief review of the species' biology, focusing on information relevant to the status review. For more thorough reviews of the species' biology, please see Eckert *et al.* (2012), Wyneken *et al.* (2013), and Spotila and Santidrián Tomillo, editors (2015).

2.1 Evolution and Taxonomy

The leatherback turtle is the only extant species of a complex evolutionary lineage (the Dermochelyids) that diverged from other sea turtles 100 to 150 million years ago, based on molecular and morphological characters (Zangerl 1980; Duchene *et al.* 2012; Pritchard 2015; Evers and Benson 2018). Despite its ancient evolutionary history, the species exhibits a shallow phylogeny based on mitochondrial deoxyribonucleic acid (mtDNA) analyses (Dutton *et al.* 1999). This discrepancy is likely due to extensive population-level extinctions (or extirpations) during the early Pleistocene glaciation, less than one million years ago, reducing the species to a single lineage that subsequently formed the basis for current populations (Dutton *et al.* 1999;

Dutton 2004; Dutton *et al.* 2013b). The Genus *Dermochelys* is monotypic. The species has had many synonyms since its identification by Vandelli in 1761 (Fritz and Havas 2007); however, only one extant species is recognized as follows:

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Dermochelyidae
Genus: *Dermochelys*
Species: *Dermochelys coriacea*
Common name: leatherback turtle

2.2 Description

The leatherback turtle is the largest turtle in the world, with adults weighing an average of 1,000 pounds (453 kg) and over 5 feet (1.52 m) in straight carapace length (SCL; Davenport *et al.* 2011). The carapace of leatherback turtles is easily distinguishable from other sea turtles by its black coloration, prominent dorsal ridges, thick layers of fatty connective tissue, and lack of heavily keratinized scutes (Figure 1). Adults have a pink spot on the top of their heads (Eckert *et al.* 2012), which is associated with the pineal gland (Wyneken 2001). Leatherback turtles have seven longitudinal ridges (which are white in hatchlings) that taper to a blunt point, making it hydrodynamic (Pritchard 2015). They have long, clawless front flippers (which are almost as long as the carapace in hatchlings) and large, paddle-shaped back flippers, which facilitate nest excavation. They have loosely interlocking dermal bones and subdermal flexibility that may absorb rapid pressure changes during fast, deep dives (Pritchard 2015). They also have pointed tooth-like cusps, to slice soft-bodied prey (e.g., jellyfish) and backward pointing keratinized spines (papillae) in the mouth and throat, to retain prey as water is expelled (Pritchard 2015). Adult males have a longer tail and more distal cloaca relative to females (James 2004; James *et al.* 2007; Eckert *et al.* 2012).

In most cases, leatherback tracks and nests are easy to identify, which facilitates the use of nests or tracks to estimate the number of nesting females (see Demographic Factors). Leatherback tracks are wide (150 to 230 cm), deep, broad, and symmetrical, with a deep groove formed by the tail (Pritchard and Mortimer 1999). Nesting females leave a conspicuous body pit and displace a large amount of sand to excavate and cover their nests.

Figure 1. Leatherback adult. (Jack Javech, NOAA)

2.3 Distribution and Habitat

The leatherback turtle 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 rate, 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; Casey *et al.* 2014; Eckert *et al.* 2012; Williard *et al.* 2012; reviewed in Wallace and Jones 2015).

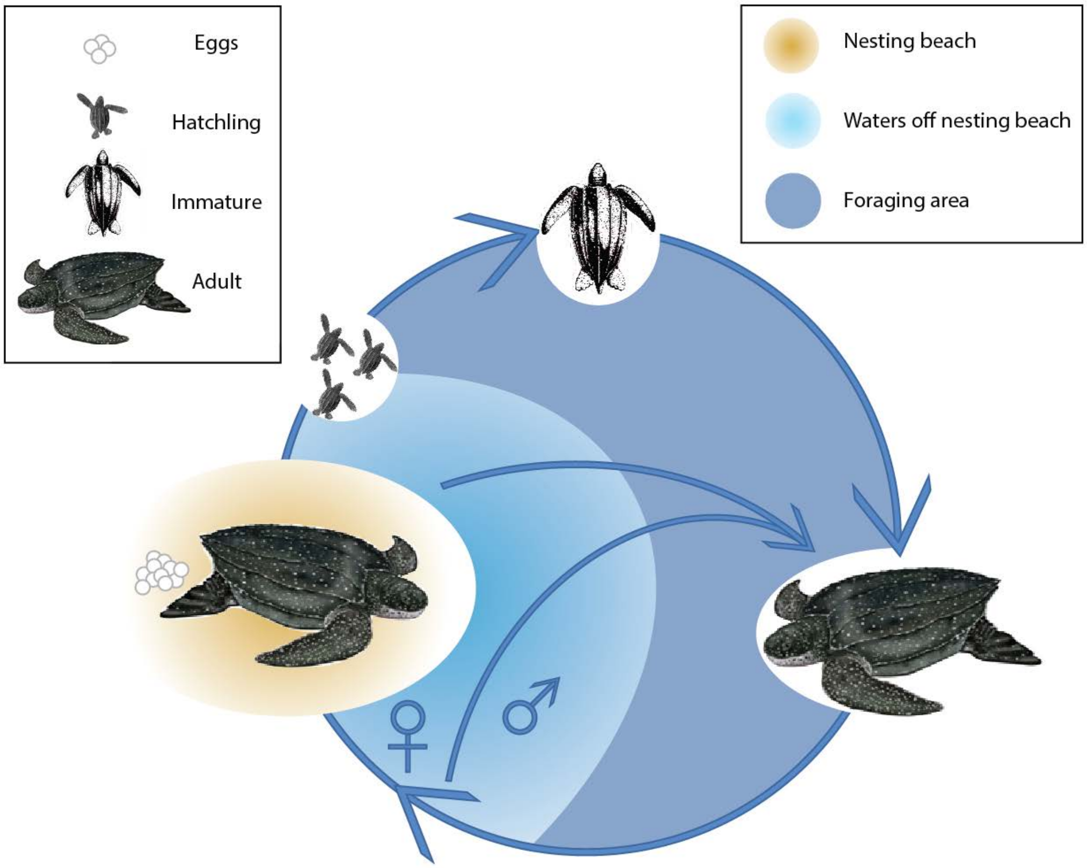
Nesting is restricted to mainly tropical or subtropical beaches; however, nesting also occurs on temperate beaches of the southwest Indian Ocean (Pritchard and Mortimer 1999). Nesting usually occurs on high-energy beaches (Pritchard 1976). The primary factors influencing shoreline suitability for nesting appear to be a lack of abrasive 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 (Pritchard 1979).

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 identified and described 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.* 2006a; James *et al* 2006b).

2.4 Life History

Leatherback turtles spend the majority of their lives at sea, where they develop, forage, migrate, and mate (Figure 2). Less is known about their life at sea because of their wide ranging habits, migratory behavior, multiple life stages, and the logistical difficulties of finding and capturing turtles at sea. For simplification, we identify the leatherback life stages as hatchling, immature (including juveniles and subadults), and adult.

Figure 2. Lifecycle of the leatherback turtle.



2.4.1 Egg

Females lay 20 to 100 eggs per nest (clutch size; Eckert *et al.* 2012). Clutch size is correlated with female size (Rostal 2015). The eggs are laid in a nest chamber of cool, wet, subsurface sand (Eckert *et al.* 2015) at an average depth of 70 cm (Billes and Fretey 2001). Leatherback turtles lay the heaviest eggs among reptiles, ranging from 71.8 to 84.3 g (Eckert *et al.* 2012). They also lay shelled albumen gobs that may reduce predation on eggs (Caut *et al.* 2006) or simply represent "productive over-runs" (Wallace *et al.* 2006b; 2007; Sotherland *et al.* 2015). Though the presence of gobs may facilitate the circulation of oxygen around the eggs, Wallace *et al.* (2004) and Sotherland *et al.* (2015) did not find differences in oxygen levels near shelled albumen gobs versus other regions of the nest. As with all sea turtles, sex is determined by egg temperature during the second trimester of development (thermosensitive period; reviewed by Wibbels 2003). Warmer temperatures produce more female embryos and temperatures exceeding 32 °C may result in death (Mrosovsky *et al.* 1984; Hawkes *et al.* 2007). After about 2 months of incubation, hatchlings emerge from the nest (Eckert *et al.* 2015).

2.4.2 Hatchling

Hatchlings emerge from the nest at 55 to 65 mm SCL. They crawl toward the sea, guided by light, specifically the light differential between darker land and brighter open ocean horizon. Once in the surf, they swim continuously, for hours to days, away from land (Hall 1987; Wyneken and Salmon 1992; Eckert *et al.* 2012). Thereafter, they engage in a diel swimming pattern and are capable of diving (Davenport 1987; Wyneken and Salmon 1992; Price *et al.* 2007; Eckert *et al.* 2012). Few empirical data exist about hatchling dispersal (Hamann *et al.* 2011; Gaspar *et al.* 2012; Shillinger *et al.* 2012). Gaspar *et al.* (2012) hypothesized that after an initial period of mostly passive drift, juveniles begin to actively swim towards warmer latitudes before winter and towards higher latitudes during spring. A recent modeling exercise suggests that hatchlings produced on nesting beaches in French Guiana and Suriname move to foraging areas of Europe and North Africa in less than 4 years (Lalire and Gaspar 2019). By 2 to 8 weeks of age, leatherback post-hatchlings forage exclusively on gelatinous prey, and their diet is similar to later life stages (Salmon *et al.* 2004). Unlike other sea turtle hatchlings, there is no evidence to indicate that leatherback post-hatchlings associate with *Sargassum* or epipelagic debris (Carr 1987).

2.4.3 Immature

Because small leatherback turtles are rarely encountered, little is known about their biology (Pritchard 2015). Synthesizing data from numerous sources, Eckert (2002) found that small leatherback turtles (less than 100 cm curved carapace length, CCL) were generally documented only in warm, tropical waters (greater than 26 °C). At sizes larger than 100 cm CCL, they likely develop the attributes (larger body mass, insulation, and countercurrent circulation) that enable forays into colder, temperate waters (Eckert *et al.* 2012). Leatherback turtles grow at a rapid rate, relative to other sea turtles (Jones *et al.* 2011). This is possibly facilitated by the presence of blood vessels running through the cartilaginous ends of the bones (Rhodin *et al.* 1996). As explained under Section 2.5 Foraging Ecology, the distribution of leatherback turtles at this life stage is likely determined by the distribution and abundance of their gelatinous prey (Eckert *et al.* 2012).

2.4.4 Adult

Based on mean estimates, leatherback turtles mature at approximately 20 years of age and approximately 130 cm CCL in size (see Section 3.1.3 Ecological and Physiological Factors (Morphological Discontinuity)) (reviewed by Jones *et al.* 2011). Adult males and females migrate from foraging areas to the waters off nesting beaches, where mating likely occurs (Godfrey and Barreto 1998; James *et al.* 2005a; James *et al.* 2005b; Rostal 2015). Such migrations often require navigation over long distances using bathymetric and possibly geomagnetic cues (Morreale *et al.* 1996; Gaspar *et al.* 2006; Shillinger *et al.* 2008). Analyses of maternally and biparentally inherited genetic markers indicate that both female and male leatherback turtles exhibit natal homing or philopatry, returning to the waters off their natal beaches to mate (Dutton *et al.* 1999; Dutton *et al.* 2013b; Jensen *et al.* 2013). Nesting females exhibit low site-fidelity to their natal beaches, returning to the same region, but not necessarily the same beach, to nest (Dutton *et al.* 1999; Dutton *et al.* 2007; Dutton *et al.* 2013b). Multiple paternity has been found within nests (Curtis 1998; Dutton and Davis 1998; Rieder *et al.* 1998; Dutton *et al.* 2000; Crim *et al.* 2002; Stewart and Dutton 2011, 2014). Because females mate with multiple males (Figgener *et al.* 2012; Stewart and Dutton 2011), sperm competition may occur (Dutton *et al.* 2000) and sperm storage is likely (Stewart and Dutton 2011).

The nesting season lasts for 3 to 6 months generally during the austral or boreal summer, depending on the location of the nesting beaches. Nesting occurs at night and generally lasts 80 to 140 minutes (Eckert *et al.* 2012). Non-nesting emergences (those that do not involve oviposition) occur and vary in frequency based on location and month. As summarized by Eckert *et al.* (2012), the nesting process involves:

- Emergence from sea: generally involving a steep approach and/or strong wave action to minimize crawl length
- Crawl and selection of nest site: above high tide line, to optimize respiratory requirements of embryos and avoid inundation and erosion, but below vegetation
- Excavation of body pit: removal of dry, loose sand using front flippers
- Excavation of nest chamber: dug and shaped using hind flippers
- Oviposition: egg-laying of up to 100 eggs plus shelled albumen gobs
- Filling the nest chamber: sand is scooped and compacted with rear flippers
- Covering and concealing the nest site: displacing sand from a wide area to camouflage the nest
- Returning to sea using the light differential between darker land and brighter open ocean horizon

For leatherback turtles, the average clutch frequency is five to seven (with a range of one to 14) clutches per season, with an internesting interval of 7 to 15 days (Eckert *et al.* 2012; Eckert *et al.* 2015). At the end of the nesting season, adults return to foraging areas, the locations of which are likely determined by the distribution and abundance of their gelatinous prey (Eckert *et al.* 2012). Nesting occurs on average every 2 to 4 years (the remigration interval) with a range of 1 to 11 years (Eckert *et al.* 2015). The remigration interval represents the amount of time needed to acquire enough resources for migration and egg production. It is likely influenced by prey availability, which is determined by primary productivity and oceanographic or climate conditions (Hays 2000; Rivalan *et al.* 2005; Wallace *et al.* 2006a; Saba *et al.* 2008; Reina *et al.* 2009; Saba *et al.* 2015). Therefore, the remigration interval may fluctuate over time and reveal information about prey availability.

2.5 Foraging Ecology

It is assumed that all life stages forage on gelatinous prey such as jellyfish (Cnidaria), tunicates (Tunicata/Urochordata), and ctenophores (Ctenophora). Eckert *et al.* (2012) and Jones and Seminoff (2013) summarized previous studies on the leatherback diet, identifying species consumed and location. They conclude that the species feeds primarily on pelagic medusa and that other organisms (and plastics) may be opportunistically or accidentally ingested (Eckert *et al.* 2012; Jones and Seminoff 2013). Leatherback turtles forage by sucking ambient water into the mouth and seizing prey with sharp posteriorly-pointed cusps. Gelatinous prey have relatively low energy content, requiring the turtles to consume large quantities of prey to meet metabolic demands (Heaslip *et al.* 2012; Jones *et al.* 2012; Wallace *et al.* 2018), possibly focusing on the higher-nutrient gonads (Doyle *et al.* 2007). Leatherback turtles likely maximize their caloric intake by aligning their foraging behavior to prey distribution and abundance (Sherrill-Mix *et al.* 2007). This means diversifying their behavior to include foraging in: coastal and pelagic waters; subtropical, temperate, and boreal waters; ephemeral eddies; and at the ocean surface or to depths of over one kilometer for almost 90 minutes (Robinson and Paladino 2015). Such flexibility allows the leatherback turtle to consume large amounts of prey at various locations throughout the year. Within these locations, oceanic features such as water temperature, downwelling, Ekman upwelling, sea surface height, chlorophyll-a concentration, and mesoscale eddies may attract and retain foraging turtles (Benson *et al.* 2011; Bailey *et al.* 2012a).

3. DPS Analysis

In the following sections, we evaluate the discreteness and significance of populations within the species following the DPS Policy.

3.1 Discreteness

Below we evaluate discreteness of leatherback turtle populations using the criteria for discreteness under the DPS Policy:

- 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
- Delimited by international governmental boundaries within which significant differences exist in regards to control of exploitation, management of habitat, conservation status, or regulatory mechanisms.

3.1.1 Behavioral Factors

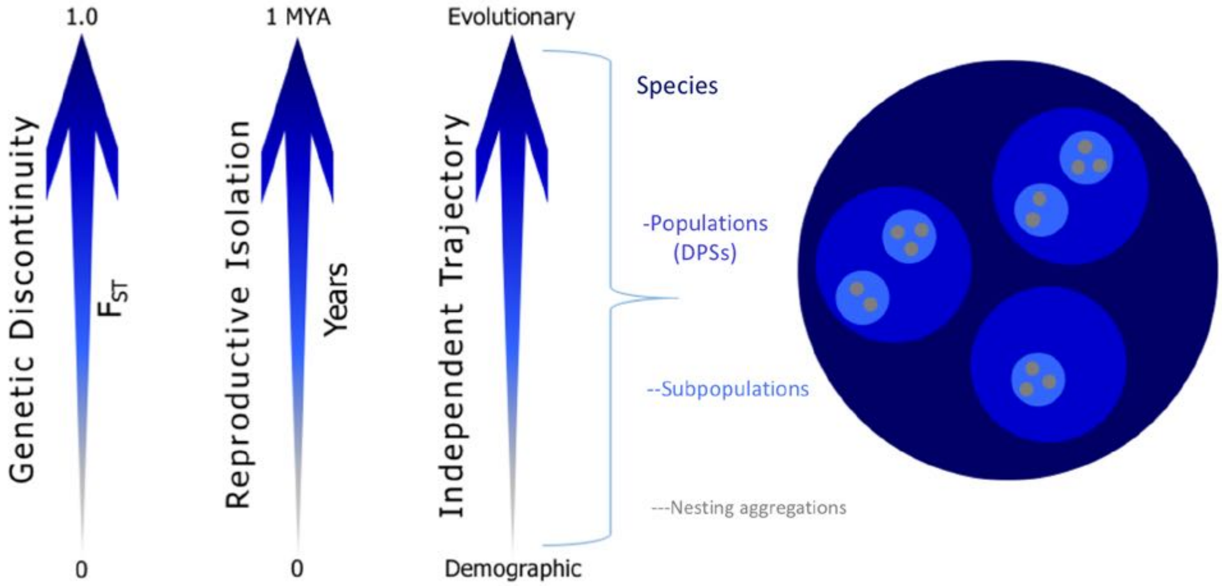
Behavioral factors result in marked separation of populations when they prevent interbreeding among individuals from different populations. Often, such reproductive isolation originated thousands of years prior and are manifested as genetic discontinuity among populations. Tagging and tracking studies provide a contemporary snapshot of leatherback movement, which may also indicate that interaction is limited. Here we evaluate whether these behavioral factors result in marked separation among populations.

3.1.1.1 Genetic Discontinuity

Although leatherback turtles are part of an evolutionary trajectory that originated at least 100 million years ago, the intraspecific phylogeny recorded in mtDNA haplotypes is less than a million years old (Dutton *et al.* 1999). The species is unlike other sea turtle species, which each have several mtDNA lineages that indicate intraspecific separation 2 to 4 million years ago, corresponding to the emergence of the Isthmus of Panama (Dutton *et al.* 1996; Bowen and Karl 1997). Instead, leatherback genetic structure reflects global radiation from a single mtDNA lineage (Bowen and Karl 1997; Dutton *et al.* 1996, 1999), with the most divergent haplotypes indicating separation between the Atlantic and Indo-Pacific Ocean populations approximately 0.17 million years before present (Duchene *et al.* 2012). This post-Pleistocene expansion appears to have originated from a refugium in the Indian Ocean (Dutton *et al.* 1999) with subsequent recolonization of the Atlantic Ocean most likely occurring via the eastern Atlantic, as nesting populations in Ghana and Gabon share haplotypes with populations in the Indo-Pacific (Dutton *et al.* 2013b). The most divergent mtDNA haplotypes occur between the western Atlantic Ocean (Florida, Costa Rica, Trinidad, French Guiana/Suriname, and St. Croix) and the eastern Pacific Ocean (Costa Rica, Mexico; Dutton *et al.* 1999; Duchene *et al.* 2012).

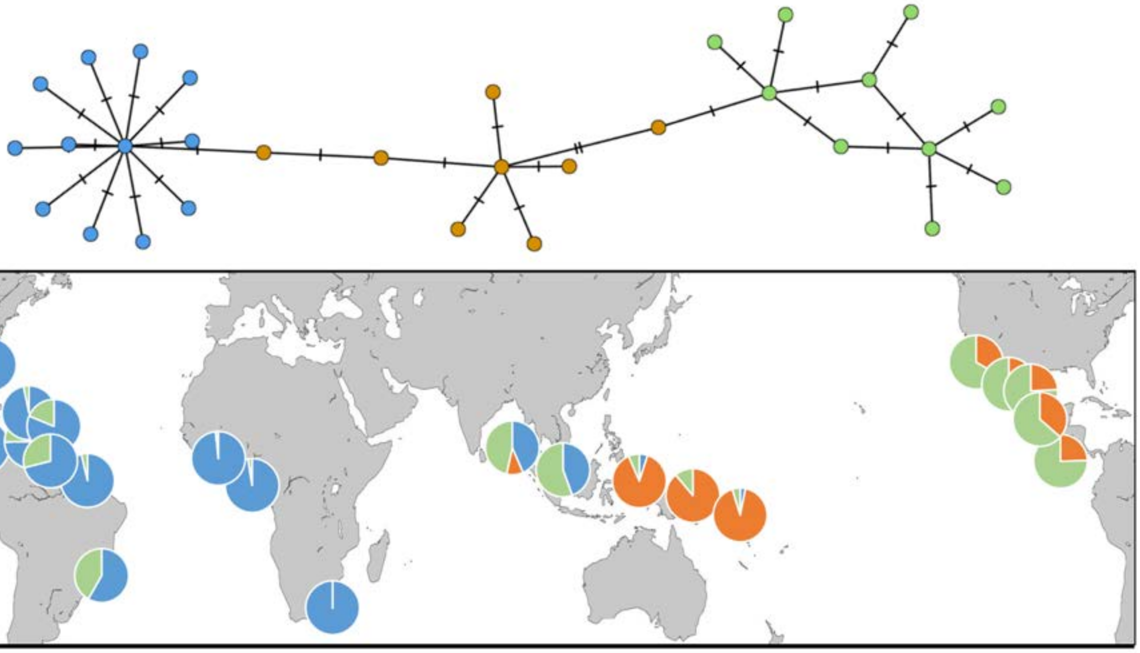
Thus, despite the ancient age of the species, leatherback populations are more recently derived than other sea turtle species (Dutton *et al.* 1996; Dutton *et al.* 1999; Bowen and Karl 2007). As a result, shared haplotypes between leatherback populations may reflect common ancestry linked to the global expansion thousands of years prior, rather than contemporary gene flow from interbreeding within the past tens or hundreds of years. Furthermore, genetic discontinuity (measures of genetic differentiation, such as F_{ST} values) occurs on a spectrum, which reflects hierarchical population structure (Figure 3). Populations that are made up of two or more subpopulations often demonstrate metapopulation dynamics, which are characterized by colonization and extinction events of local subpopulations as well as the movement of individuals and genes among subpopulations. While genetic discontinuity indicates marked separation among populations (discreteness), fine-scale genetic differentiation among subpopulations reflects metapopulation dynamics and does not meet the DPS criteria. Within a species, populations may exist on independent evolutionary trajectories, which are generally shaped by processes on evolutionary timescales and global spatial scales, creating patterns of genetic division and variation that are easily detected with standard genetic markers and reflect reproductive isolation (Wallace *et al.* 2010a; Dutton *et al.* 2013b). Examples of these populations include DPSs, Regional Management Units (RMUs; Wallace *et al.* 2010a), and evolutionary significant units, all of which are similar in concept (Dutton *et al.* 2013b; and see the DPS Policy, 61 FR 4722, February 7, 1996). Within these populations (e.g., a DPS or RMU), there may be fine-scale genetic differentiation as a result of environmental or behavioral processes that occur on ecological timescales and smaller spatial scales (Dutton *et al.* 2013b). Examples of these subpopulations, which demonstrate demographic independence but at present share a common evolutionary trajectory, include "management units" *sensu* Moritz (1994) and "demographically independent populations" (Dutton *et al.* 2013b; Komoroske *et al.* 2017). We considered fine-scale genetic differentiation and metapopulation dynamics during our analysis of spatial distribution (see 1.1.3.1 Demographic Factors). For our analysis of discreteness, we focused on genetic discontinuity that demonstrates *marked* separation (i.e., broad-scale rather than fine-scale differentiation), indicative of populations on independent evolutionary trajectories.

Figure 3. Genetic discontinuity and hierarchical population structure. Hierarchical (or nested, as shown by circles) population structure results in genetic discontinuity (F_{ST}), reproductive isolation (years), and independent trajectories (demographic to evolutionary), which range from high among species to low among nesting aggregations.



To identify genetic discontinuity indicative of marked separation, we reviewed several genetic studies, starting with global analyses, followed by regional and local analyses. The most recent and comprehensive global analysis of published and unpublished mtDNA haplotype sequence data (28 haplotypes) evaluated leatherback samples, collected from 21 nesting sites representing key regions from all ocean basins (Dutton *et al.* 2007; Shanker *et al.* 2011; Dutton *et al.* 2013b; Dutton and Shanker 2015; Dutton unpublished data). Phylogenetic analysis of these data revealed three distinct haplogroups (similar haplotypes that cluster together, relative to other haplotypes) that are geographically segregated across ocean basins (Dutton unpublished data; Figure 4). The global structure evident in this pattern of phylogeographic distribution of genetic diversity provides a starting point for assessing the discreteness of populations because it represents the deepest level of genetic partitioning, over tens of thousands years of reproductive isolation.

Figure 4. Global distribution of leatherback turtle mtDNA diversity. The haplotype network (top) identifies haplotypes (circles) belonging to haplogroups (represented by three colors) and separated from other haplogroups by mutations (hash marks). The map contains pie charts that show relative frequencies of haplotypes from the three distinct haplogroups (corresponding to the three colors) identified at 21 nesting sites. (Data from Dutton *et al.* 2007, 2013, unpublished; Shanker *et al.* 2011; Dutton and Shanker 2014).



All three haplogroups include a number of different haplotypes, such that populations with haplotypes from the same haplogroups are not necessarily genetically similar. Further investigating the patterns of haplotypic variation among populations, we found evidence for regional levels of genetic partitioning. Early mtDNA analyses indicated strong subdivision 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). Subsequent genetic studies (involving nuclear and mtDNA) further explored and refined basin-specific partitions, which we describe in the following paragraphs.

Wallace *et al.* (2010) identified global RMUs, which are defined similarly to DPSs, by reviewing the genetic data available at the time (see Appendix S2 from Wallace *et al.* 2010a). They performed a spatial analysis of these genetic data, combined with nesting, tagging, and tracking data, to identify seven leatherback RMUs (Wallace *et al.* 2010a):

- Northwest (NW) Atlantic Ocean
- Southwest (SW) Atlantic Ocean
- Southeast (SE) Atlantic Ocean
- Southwest (SW) Indian Ocean
- Northeast (NE) Indian Ocean
- West Pacific Ocean
- East Pacific Ocean

These RMUs were identified as “subpopulations” in the IUCN Red List assessment (note that we would refer to them as “populations” due to their marked separation, which demonstrates evolutionary independence in addition to demographic independence). These RMUs have been reviewed and upheld as valid by subsequent genetic studies, which provide additional evidence of significant genetic discontinuity (Dutton *et al.* 2013b; Molfetti *et al.* 2013; Komoroske *et al.* 2017; Vargas *et al.* 2017). Based on our review of published and unpublished genetic data on leatherback turtles, we conclude that the RMUs identified by Wallace *et al.* (2010) are discrete populations because they exhibit genetic discontinuity representative of marked separation. In the paragraphs below, we ask whether any other populations also exhibit this level of genetic discontinuity.

3.1.1.1.1 Atlantic Ocean

In the Atlantic Ocean, three populations exhibit genetic discontinuity and marked separation: NW Atlantic, SW Atlantic, and SE Atlantic. To evaluate the RMUs and fine-scale structure within the Atlantic, Dutton *et al.* (2013) conducted a comprehensive genetic re-analysis of rookery stock structure using longer (more informative) 763 base pair mtDNA sequences combined with nuclear marker data from 17 microsatellite loci. They evaluated larger sample sizes and previously unsampled rookeries in the Atlantic and SW Indian Oceans (Dutton *et al.* 2013b). Nesting sites included Brazil, Costa Rica, French Guiana/Suriname, Gabon, Ghana, South Africa, Trinidad, Florida, and St. Croix (U.S. Virgin Islands, USVI). Dutton *et al.* (2013) concluded that their nuclear and mtDNA results confirm the three discrete populations in the Atlantic. This genetic discontinuity is indicative of marked separation (discreteness) as demonstrated by elevated microsatellite F_{ST} values ($F_{ST} = 0.211 – 0.86$). The mtDNA also reflect this discontinuity, but this genetic marker is less variable, and thus less informative. Dutton *et al.* (2013) also identified fine-scale genetic partitioning in the Atlantic Ocean, as demonstrated by small but significant mtDNA and microsatellite F_{ST} values ($F_{ST} = 0.011 – 0.149$). However, based on the magnitude of the F_{ST} values, these fine-scale differences do not rise to the level of marked separation or discreteness. The genetic differentiation detected using nuclear DNA also suggests that males, in addition to 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.* 2013b; see Jensen *et al.* 2013).

Other genetic analyses corroborate the conclusions of Dutton *et al.* (2013). Molfetti *et al.* (2013) used nuclear and mtDNA to evaluate fine-scale structure at three nesting sites in French Guiana and the French West Indies. They concluded that leatherback turtles of the NW Atlantic Ocean derive from a single ancestral origin and should be considered part of a single evolutionary unit (Molfetti *et al.* 2013). Vargas *et al.* (2017) built on the Dutton *et al.* (2013) study by analyzing the mtDNA of additional samples from the nesting beaches of and foraging areas off Brazil. They too concluded that the SW and SE Atlantic populations have divergent origins from those in the NW Atlantic Ocean and should be considered discrete populations (Vargas *et al.* 2017). They also identified fine-scale genetic partitioning among Atlantic subpopulations (Vargas *et al.* 2017). Evaluating mtDNA haplotypes, Carreras *et al.* (2013) identified fine-scale genetic differentiation among females nesting on the beaches of the Dominican Republic and French Guiana, Trinidad, and possibly St. Croix. Thus, multiple studies provide support for the identification of three discrete populations in the Atlantic Ocean (NW, SW, and SE Atlantic). Though all studies detected additional fine-scale genetic differentiation within these three discrete populations, none indicated that these fine-scale genetic differences are sufficient to be considered marked separation.

Despite clear boundaries between nesting beaches, Atlantic Ocean populations may overlap at foraging areas. Stewart *et al.* (2016) genetically assigned one leatherback in the Gulf of Mexico (GOM) to the SW Atlantic population; however, Stewart *et al.* (2013) found no turtles originating from the SE or SW Atlantic populations in Atlantic Canadian waters (Stewart *et al.* 2013). Telemetry and tagging data, described below, provide evidence that the SW and SE Atlantic populations overlap at foraging areas.

We also considered the occurrence of leatherback turtles in the Mediterranean Sea. However, nesting has not been documented there (Camiñas 1998; reviewed by Casale and Margaritoulis 2010), and unpublished data indicate that leatherback turtles in the Mediterranean Sea likely originate from the NW Atlantic Ocean (NMFS and USFWS 2013). We conclude that three populations within the Atlantic Ocean exhibit genetic discontinuity and marked separation: NW Atlantic, SW Atlantic, and SE Atlantic.

3.1.1.1.2 Indian Ocean

In the Indian Ocean, two populations exhibit genetic discontinuity and marked separation: SW Indian and NE Indian. However, we do not have genetic data from most nesting beaches within this region. Published mtDNA haplotypes only exist for the South African population and the functionally extinct Malaysian population (Chan and Liew 1996; Dutton *et al.* 1999, 2007, 2013; Dutton 2005, 2006). Samples from the South African nesting population were analyzed in the global study (Dutton *et al.* 1999) and in the studies of Atlantic nesting sites described above (Dutton *et al.* 2013b; Vargas *et al.* 2017). The latter studies verified genetic discontinuity between the South African population and Atlantic populations, as demonstrated by nuclear ($F_{ST} > 0.126$, $P < 0.001$; Dutton *et al.* 2013b) and mtDNA ($F_{ST} > 0.061$, $P = 0.05 - 0.001$; Dutton *et al.* 2013b; $F_{ST} > 0.061$, $P < 0.01$; Vargas *et al.* 2017). These authors concluded that the South African population should be considered an RMU (SW Indian Ocean), as proposed by Wallace *et al.* (2010). The original global study did not have adequate resolution (due to shorter mtDNA sequences) to detect differentiation from the Atlantic populations, but it demonstrated genetic discontinuity between the South African and Malaysian populations ($F_{ST} = 0.415$, $P < 0.003$; Dutton *et al.* 1999). The Malaysian population is also significantly differentiated from the West Pacific population ($X_2 = 49.346$, $P = 0.002$; Dutton *et al.* 2007). Wallace *et al.* (2010) combined Malaysia and nesting beaches in Sri Lanka and the Nicobar Islands into a single RMU (NE Indian Ocean), which was originally hypothesized by Dutton (2005, 2006). Preliminary mtDNA results for leatherback turtles nesting at Little Andaman Island, India (Shanker *et al.* 2011; Dutton and Shanker 2014), indicate that this population is closely related to the functionally 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). Further genetic sampling has been recommended for all the Andaman and Nicobar Islands, as well as northern and eastern Australia, Mozambique, Sri Lanka, Sumatra, Java, Thailand, and Vietnam (Dutton *et al.* 1999, 2007). However, we considered the available data to be sufficient to conclude that two populations within the Indian Ocean exhibit genetic discontinuity and marked separation: SW Indian and NE Indian.

3.1.1.1.3 Pacific Ocean

In the Pacific Ocean, two populations exhibit genetic discontinuity and marked separation: West Pacific and East Pacific. In the global study, Dutton *et al.* (1999) compared mtDNA sequences sampled from leatherback turtles of Solomon Islands, Pacific Mexico, and Pacific Costa Rica. There was significant genetic differentiation between the samples from Solomon Islands and Mexico or Costa Rica ($F_{ST} = 0.270$ and 0.331, $P < 0.001$). However, there was no genetic differentiation between Mexico and Costa Rica ($F_{ST} < 0$, $P = 0.576$; Dutton *et al.* 1999). Additional studies verified that Mexico and Costa Rica nesting sites are genetically homogenous but distinct from the West Pacific population (Barragan *et al.* 1998; Barragan and Dutton 2000), indicating support for the discreteness of the East Pacific population. Dutton *et al.* (2007) compared the mtDNA of samples from Papua Barat in Indonesia (Jamursba-Medi and Wermon beaches), Papua New Guinea, and Solomon Islands. They found the same common haplotype and no significant differences in haplotype frequencies among the four nesting sites ($X^2 = 7.363$, $P = 0.972$; Dutton *et al.* 2007). Additional studies of this region further confirmed these results (Dutton *et al.* 2000b, 2005, 2006), providing additional support for the discreteness of the West Pacific population.

Despite clear boundaries between nesting beaches, Pacific Ocean populations may overlap at foraging areas. Genetic assignment tests assigned two turtles captured by the Hawaii longline fishery operating in the Central North Pacific to the East Pacific population (P. Dutton, NMFS, pers. comm., 2018). In a genetic study of 59 leatherback turtles caught in longline and gillnet fisheries off Peru and Chile, Donoso and Dutton (2010) revealed that approximately 15 percent originated from West Pacific nesting beaches. The two populations remain reproductively isolated because mating likely occurs off nesting beaches. We conclude that two populations within the Pacific Ocean exhibit genetic discontinuity and marked separation: West Pacific and East Pacific.

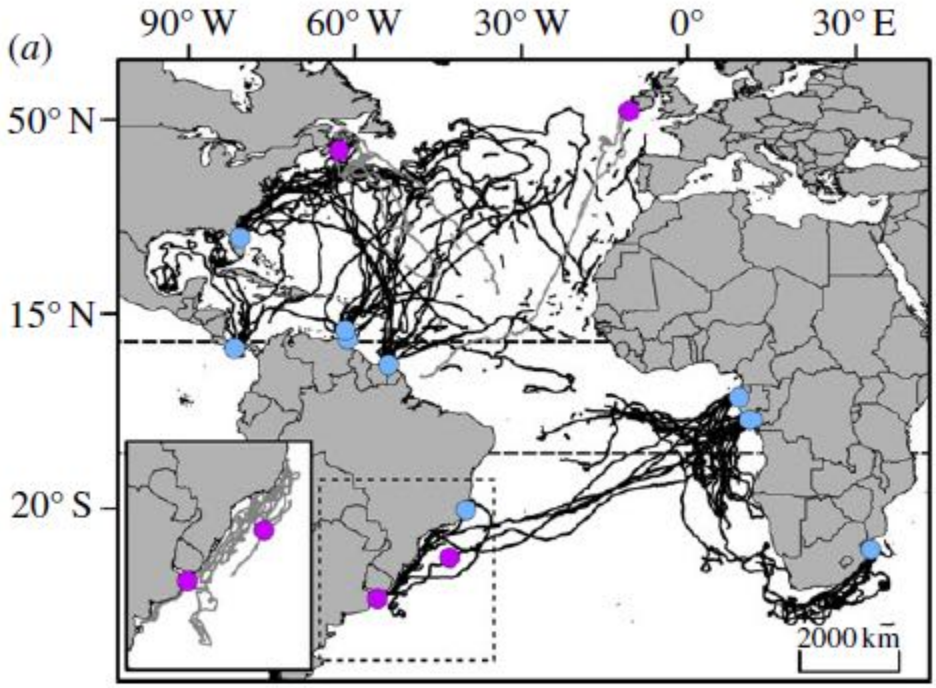
3.1.1.2 Movement (Flipper Tagging and Satellite Telemetry)

The leatherback turtle engages in the most wide-ranging movements of all sea turtle species (Hays and Scott 2013). Movements of adults and subadults span across all major ocean basins and range from equatorial waters to temperate high-latitude regions (Shillinger and Bailey 2015). Leatherback turtles originating from the same nesting beach may forage in diverse and geographically distant regions, with variance among individuals (Eckert 2006; Eckert *et al.* 2006b; Hays *et al.* 2006; Benson *et al.* 2011; Witt *et al.* 2011; Namboothri *et al.* 2012a). Conversely, leatherback turtles from different nesting beaches may move to the same foraging regions as adults (Fossette *et al.* 2010b, 2014). Patterns of leatherback movements between nesting beaches and foraging areas are complex, and appear to be linked to ocean currents that facilitate hatchling dispersal (Fossette *et al.* 2010a; Gaspar *et al.* 2012; Scott *et al.* 2017) or adult movements throughout the oceans (Lambardi *et al.* 2008; Robinson *et al.* 2016). Adults are known to return to the same foraging areas after nesting (Seminoff *et al.* 2012), and hatchlings from different nesting beaches may reach the same foraging areas, creating a mosaic of overlapping population ranges. These overlapping foraging patterns can confound our ability to clearly delineate separations based solely on movement patterns, making it essential to also consider other data, including genetics. Below we summarize leatherback movement patterns from flipper tagging and satellite telemetry studies within each ocean basin, with the caveat that the available information varies markedly by ocean basin. While these data are informative to our understanding of movement, there are temporal biases inherent in telemetry data. While remigration intervals last 3 years or more, telemetry data only provides information for the first 6-12 months of deployment.

3.1.1.2.1 Atlantic Ocean

The greatest number of flipper tagging and satellite telemetry studies has been conducted at multiple nesting beaches and foraging areas in the Atlantic Ocean. They collectively show a clear separation of leatherback movements between the North and South Atlantic Oceans (Figure 5; Fossette *et al.* 2014). Earlier studies involved recovery of flipper tagged animals (e.g., Pritchard 1979), but recent satellite telemetry studies provide a more comprehensive picture of Atlantic leatherback movements, and we focus on the latter in our descriptions below unless flipper tagging studies contribute different or unique information to inform overall leatherback movements.

Figure 5. Satellite tracking of 106 leatherback turtles in the Atlantic Ocean. Turtles (93 nesting females, four males, and 1 immature turtle) were tagged at nesting beaches (blue dots and black lines) or at sea (purple dots and grey lines) between 1995 and 2010. Image: Figure 1a from Fossette *et al.* (2014), Proceedings of the Royal Society B, *http://doi.org/10.1098/rspb.2013.3065*



Within the North Atlantic Ocean, extensive telemetry studies have been conducted on females at diverse nesting beaches in northern South America (Ferraroli *et al.* 2004; Hays *et al.* 2004), Central America (Evans 2008, Fossette *et al.* 2014, Aleksa 2017), the Caribbean (Eckert 2006a, Hays *et al.* 2006), and Florida (Eckert *et al.* 2006b). Additional telemetry deployments were conducted on free-swimming or bycaught turtles of both sexes foraging in high-latitude areas of the NW Atlantic Ocean (James *et al.* 2005a; James *et al.* 2005b; Dodge *et al.* 2014), NE Atlantic Ocean (Doyle *et al.* 2008), and northern GOM (Aleksa *et al.* 2018). Collectively, these studies show that females nesting on NW Atlantic beaches move throughout most of the North Atlantic from the Equator to about 50° N latitude. Key foraging destinations included: waters along the eastern coast of North America and the GOM; open-ocean areas of the North Atlantic Ocean (e.g., the Flemish Cap); and coastal areas off Great Britain, Ireland, Spain, Portugal, and the Canary and Cape Verde Islands (Eckert *et al.* 2006). There appears to be considerable mixing of turtles between NW Atlantic nesting beaches and multiple foraging areas throughout the North Atlantic Ocean (Fossette *et al.* 2014), but their movements very rarely extend into waters south of the Equator.

Flipper tagging studies further support the connectivity among nesting beaches and foraging areas of the North Atlantic Ocean. Individual females may nest up to 532 km apart within a season and 1,849 km between seasons, resulting in gene flow among distant nesting beaches within the North Atlantic Ocean (Troëng *et al.* 2004; Chacón-Chaverri and Eckert 2007; Horrocks *et al.* 2016). Tagging data indicate that nesting females move among Panama, Colombia, Venezuela, and Guyana (Dutton *et al.* 2013b). Post-nesting females tagged on Honduran and Colombian beaches also nested on beaches in Costa Rica (Troëng *et al.* 2004). Post-nesting females tagged in Costa Rica and Panama also nested in Cuba, Florida, St. Croix, and Grenada (Turtle Expert Working Group 2007). A post-nesting female tagged in St. Croix also nested in Dominica (reviewed by Bräutigam and Eckert 2006; Turtle Expert Working Group 2007). These examples illustrate contemporary gene flow within the NW Atlantic population, supporting our conclusion that the fine-scale genetic differentiation identified within this population is not indicative of marked separation. Flipper tagging also provides evidence for connectivity throughout the large foraging range of the population: one leatherback turtle, tagged on a nesting beach in Trinidad, stranded in the NE Mediterranean Sea, where it was likely foraging (Sönmez *et al.* 2008).

In the eastern Atlantic Ocean, leatherback nesting occurs along the coast of western Africa (Turtle Expert Working Group 2007). Genetic studies (Dutton *et al.* 2013b) have linked the Ghana nesting subpopulation with turtles that nest in Gabon and forage throughout the South Atlantic Ocean. This linkage is supported by the southward movement of 31 telemetry tracked leatherback turtles from Gabon (Fossette *et al.* 2014; Figure 5). A telemetry study of leatherback turtles nesting in Gabon indicated use of multiple foraging areas, including waters off SW Africa, in the south equatorial Atlantic, and waters off SE Brazil and Uruguay (Witt *et al.* 2011). Telemetry data available for post-nesting females at Bioko Island indicate movement into the South Atlantic Ocean (*https://my.wildlifecomputers.com/data/map/?id=5cfa59c12c72b03fb029f269*). Flipper tagging data are limited due to less extensive monitoring in the area. However, two studies identified nesting females in Gabon with flipper tagging scars that may be indicative of previous tagging in French Guiana (Fretey and Girardin 1989; Fretey and Fernández-Cordeiro 1996). Because this hypothesis has not been substantiated, we conclude that nesting females do not move between beaches of the NW and SE Atlantic Ocean.

In the SW Atlantic Ocean, leatherback turtles nest in SE Brazil. Telemetry studies have been conducted on post-nesting females tagged at nesting beaches (Almeida *et al.* 2011) and turtles bycaught in fisheries off South America (Billes *et al.* 2006, López-Mendilaharsu *et al.* 2009). These turtles used the same foraging areas as post-nesting females from Gabon, including waters off SW Africa, in the south equatorial Atlantic, and waters off SE Brazil and Uruguay (Almeida *et al.* 2011). One post-nesting female tagged in Brazil crossed the Atlantic Ocean into waters off Angola.

Evaluating all movement data in the Atlantic Ocean, we find that the NW Atlantic population is markedly separated from the two populations in the South Atlantic Ocean. Turtles of the NW Atlantic population nest and forage exclusively in the North Atlantic Ocean. Leatherback turtles of the SE and SW Atlantic populations cross the South Atlantic Ocean to forage, and they overlap in some foraging areas (see also Fossette *et al.* 2014). However, genetic data indicate that they return to their natal beaches to nest on opposite sides of the Atlantic Ocean (Dutton *et al.* 2013b; Vargas *et al.* 2017), and no flipper tag recoveries or satellite tracks contradict these data. Therefore, we conclude that movement data support the marked separation of three populations in the Atlantic Ocean: NW Atlantic, SW Atlantic, and SE Atlantic.

3.1.1.2.2 Indian Ocean

Leatherback turtles have been tracked from nesting beaches within the SW and NE 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 the Andaman Islands in the NE Indian Ocean (Namboothri *et al.* 2012a). No telemetry studies have been conducted at other known Indian Ocean nesting beaches in Sri Lanka, Thailand, and Sumatra and Java Islands, Indonesia (Wallace *et al.* 2010a). There are, however, flipper tag data to inform movements from those nesting beaches (Hamman *et al.* 2006) and from the functionally extinct nesting population at Terengganu, Malaysia, which we include in this Indian Ocean section because of its apparent genetic linkage to the NE Indian population (Shanker *et al.* 2011; Dutton and Shanker 2014).

Females nesting on South African beaches showed diverse movements that were highly influenced by complex oceanographic currents and features leading them to foraging destinations in the South Atlantic Ocean, SW Indian Ocean, and Mozambique Channel (Hughes *et al.* 1998, Luschi *et al.* 2006, Lambardi *et al.* 2008; Robinson *et al.* 2016). About half of the females tagged at nesting beaches in South Africa subsequently foraged in shallow nearshore waters of the Mozambique Channel (Robinson *et al.* 2016) and northern Madagascar. The other half of post-nesting females moved southward within the Agulhas Current, and then either moved eastward into oceanic waters of the Indian Ocean or westward into the South Atlantic Ocean. Movements into Atlantic foraging regions off South Africa, Angola, and Namibia overlap with areas used by females nesting in Gabon and Brazil. Thus, waters of the productive Benguela Current off southern Africa represent a mixed foraging area for leatherback turtles from the South Atlantic Ocean and SW Indian Ocean. However, there is no genetic evidence for contemporary interbreeding between these populations (Dutton *et al.* 2013b), and tagging data do not indicate movement between these distant nesting beaches.

The small number of satellite tracks (less than 10) available for post-nesting females in the NE Indian Ocean capture a similarly diverse range of foraging destinations spanning the entire width of the Indian Ocean (Namboothri *et al.* 2012a). About half of the post-nesting females tagged at the Andaman Islands moved westward, with two individuals reaching the same foraging areas off northern Madagascar and the Mozambique Channel used by post-nesting females tagged in South Africa (Robinson *et al.* 2016). 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. A second leatherback traveled as far east as the island of Timor before contact was lost.

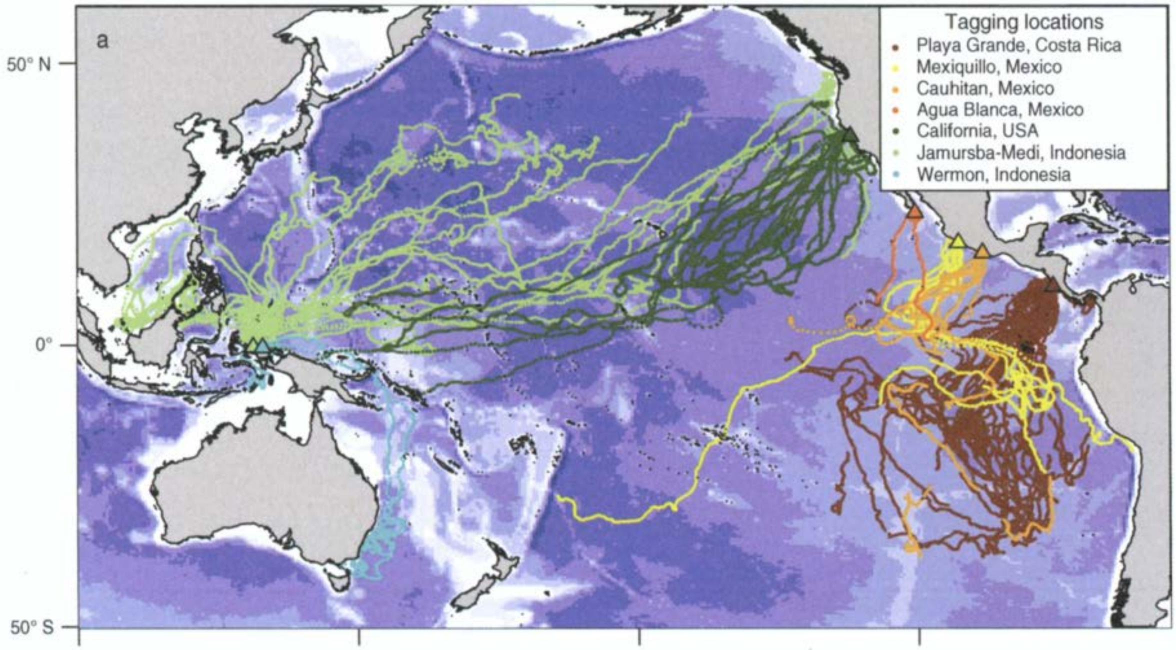
Flipper tag data provide additional linkages for NE Indian leatherback turtles. One tag recovery from a leatherback that nested in Java, Indonesia, revealed movement to a foraging area off NW Australia (Hamann *et al.* 2006). Flipper tag recoveries also provide the only information on movements of leatherback turtles from the former nesting beach at Terengganu, Malaysia, with tags recovered from coastal waters off Hainan (China), Philippines, Borneo (Malaysia), and Japan (Hamman *et al.* 2006). No tag recovery data are available for females nesting in Sri Lanka, Thailand, or Vietnam (Hamman *et al.* 2006).

Evaluating all movement data in the Indian Ocean, we find that the NE Indian population is markedly separated from the SW Indian population. Despite sharing a foraging area, there is no genetic evidence for contemporary interbreeding between the two populations (Dutton *et al.* 2013b), and tagging data do not indicate movement between the distant nesting beaches.

3.1.1.2.3 Pacific Ocean

Similar to the Atlantic, there have been extensive telemetry studies of leatherback turtles within the Pacific Basin, and these tracking data provide the most complete understanding of adult and subadult leatherback movements. Satellite tag deployments have been conducted at multiple nesting beaches in the eastern (Morreale *et al.* 1996; Eckert and Sarti 1997; Shillinger *et al.* 2008) and western Pacific Ocean (Benson *et al.* 2007a; Benson *et al.* 2007b; Benson *et al.* 2011), as well as foraging regions off California (Benson *et al.* 2007c; 2011; 2018a). Post-nesting females of the East and West Pacific populations show distinctly different movement patterns (Bailey *et al.* 2012a; Figure 6), which we review separately below.

Figure 6. Satellite tracking of 135 leatherback turtles in the Pacific Ocean. Individuals were tagged at their nesting beaches or at sea. Image: Figure 1a from Bailey *et al.* (2012a), PLOS ONE, *https://doi.org/10.1371/journal.pone.0036401*



Telemetry deployments at eastern Pacific nesting beaches have been conducted at multiple sites in Mexico (Eckert and Sarti 1997) and one location in Costa Rica (Morreale *et al.* 1996; Shillinger *et al.* 2008). Virtually all females nesting along the eastern Pacific coast moved southward across the Equator to forage in open-ocean waters of the SE Pacific Ocean, although a few individuals traveled to coastal waters of Central America, Peru, and Chile. Additionally, one female nesting in Mexico traveled southwestward into the western South Pacific before transmissions stopped north of New Zealand. Post-nesting eastern Pacific females forage broadly in the SE Pacific Ocean but do not appear to have well-defined foraging areas similar to other populations. Tracking data indicate little or no overlap with foraging areas used by females nesting in the West Pacific Ocean (Benson *et al.* 2011; Bailey *et al.* 2012a), except possibly near the Kermadec Islands north of New Zealand, where females nesting in Papua New Guinea and Solomon Islands are also known to forage (Benson *et al.* 2011, see below). However, a genetic study of bycatch off the coast of Chile and Peru indicated that 15 percent of leatherback turtles originated from West Pacific nesting beaches (Dutton *et al.* 2010), suggesting that foraging overlap may be more prevalent than estimated by existing telemetry data.

Data on movements for post-nesting females of the West Pacific population are markedly more complex and demonstrate a bimodal pattern of seasonal nesting in Indonesia, Papua New Guinea, and the Solomon Islands (Benson *et al.* 2011; Tapilatu *et al.* 2013), with peaks in January and July (boreal winter and summer, respectively). Telemetry deployments show dramatically different foraging destinations for post-nesting females tagged during each season (Benson *et al.* 2011), likely facilitated by seasonal changes in monsoon winds and ocean currents (Gaspar *et al.* 2012).

Telemetry deployments on winter-nesting females have been conducted in Papua New Guinea, Solomon Islands and at Papua Barat, Indonesia (Benson *et al.* 2007a; Benson *et al.* 2007b; 2011). Post-nesting movements predominantly tracked into the Southern Hemisphere, including diverse coastal and open-ocean ecosystems. Nearly all females nesting in the Solomon Islands and Papua New Guinea traveled southward into waters off SE Australia and northern New Zealand, except for one individual tagged in the Solomon Islands that traveled westward into the Gulf of Papua over the course of a 266-day track. In contrast, only 35 percent of winter-nesting females tagged at Papua Barat, Indonesia moved southward into waters off Australia and New Zealand, while the remaining 65 percent moved west and south into the Halmahera, Ceram, or Banda Seas, remaining in tropical island-associated Indonesian waters to forage.

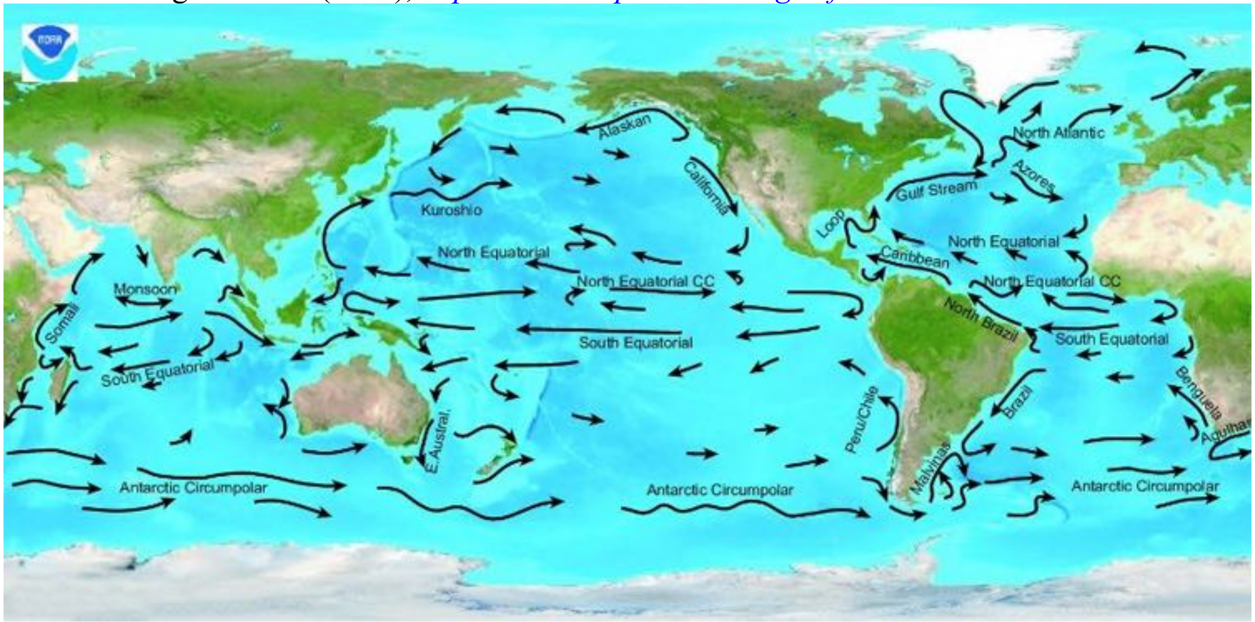
Summer nesting (July peak) is mostly confined to Papua Barat, Indonesia (northern Bird's Head Peninsula), although low levels of summer nesting are known or suspected for Papua New Guinea (Rei 2005) and the Solomon Islands (Benson *et al.* 2011; Jino *et al.* 2018). Post-nesting destinations for females from Papua Barat also include diverse coastal and oceanic ecosystems throughout the northern Indo-Pacific region (Benson *et al.* 2011). About one third of deployments resulted in movements westward into the Sulawesi, Sulu, and South China Seas to forage primarily off Malaysian Borneo and Palawan Island, Philippines. The other two thirds of post-nesting females traveled northeastward into waters of the western, central, or eastern temperate North Pacific, including the most distant foraging grounds off California, Oregon and Washington. Leatherback turtles of both sexes have also been satellite-tagged at one of the U.S. foraging areas, off central California (Benson *et al.* 2007c; 2011; 2018b). One individual subsequently nested during the summer in Papua Barat, Indonesia, and a second satellite-tagged leatherback traveled to Solomon Islands and nested during May (Benson *et al.* 2011). Multiple individuals tagged with passive integrated transponder (PIT) tags provide additional evidence of movement between Papua Barat and California (S. Benson, unpublished data). Many of the leatherback turtles tagged off California remained in the eastern North Pacific Ocean for two or more years, spending winter months in the warmer equatorial eastern Pacific Ocean before returning to cool-water foraging grounds in the California Current Ecosystem during spring through fall.

Evaluating all movement data in the Pacific Ocean, we find that the East Pacific population is markedly separated from the West Pacific population. Despite sharing a foraging area in the East Pacific Ocean, 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.

3.1.2 Physical Factors

The leatherback turtle has a global range, with foraging areas extending into high latitudes. Nesting however, mainly occurs on tropical or subtropical beaches. Post-hatchling dispersal is determined by the ocean currents encountered off nesting beaches (Fig. 7). While adults can 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.* 2013b; Jensen *et al.* 2013), creating reproductive isolation only among distant nesting sites, which may also be physically separated from one another via land masses and oceanographic barriers to gene flow.

Figure 7. Global ocean currents. Black arrows represent the location and direction of ocean currents. Image: NOAA (2019), *https://oceanexplorer.noaa.gov/facts/climate.html*



The most obvious physical barriers for leatherback turtles are land masses. Sea 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. The Isthmus of Panama, which emerged approximately 3 million years ago, existed long before the recent expansion of the species, which occurred less than one million years ago (Bowen and Karl 1996; Dutton *et al.* 1996, 1999). As a result, the evolutionary separation between the Atlantic and Pacific populations is likely the oldest within the species (Duchene *et al.* 2012); this corresponds to genetic data, which show the greatest mtDNA divergence between western Atlantic and eastern Pacific populations (Dutton *et al.* 1999; Duchene *et al.* 2012).

Within the Pacific Ocean, the East Pacific Barrier physically separates coastal marine populations by 4,000 to 7,000 km of uninterrupted deep water between the central Pacific Ocean and the Americas (Briggs 1974). However, leatherback turtles are clearly capable of crossing this barrier: tracking studies demonstrate that females nesting in the western Pacific forage off California (Benson *et al.* 2011). In a genetic study of 59 leatherback turtles caught in longline and gillnet fisheries off Peru and Chile, Dutton *et al.* (2010) revealed that approximately 15 percent had western Pacific origins. In addition, a simulation study demonstrated that western Pacific hatchlings may be advected by ocean currents to foraging areas in the Indonesian and China Seas, North and South Pacific Oceans and the Indian Ocean (Gaspar *et al.* 2012). Most eastern Pacific leatherback turtles rely on the eastern equatorial Pacific and South Pacific Gyre for foraging, and nesting peaks are associated with cool, highly productive La Niña events (i.e., cool phases of El Niño-Southern Oscillation, ENSO) or large-scale phytoplankton blooms following El Niño events, which are warm phases of ENSO (Saba *et al.* 2008). Their migration appears to be somewhat constrained by the South Pacific Gyre, though they use active movement to pass through the North Equatorial Counter Current, South Equatorial Current, Equatorial Undercurrent, and South Equatorial Current (Shillinger *et al.* 2008). Their movements, like females nesting in the western Pacific, are likely driven by prey availability and ocean currents (Gaspar *et al.* 2012; Schick *et al.* 2013). While females nesting in the eastern Pacific generally remain in the East Pacific Ocean, one post-nesting female tracked from Mexico traveled southwestward into the western South Pacific before transmissions stopped north of New Zealand. Considering all of these data together, we find that leatherback turtles cross the East Pacific Barrier to reach distant foraging areas. However, we could not find any data to support nesting interchange across the Pacific Ocean. Genetic data indicate that eastern and western Pacific populations are reproductively separated (Barragan *et al.* 1998; Dutton *et al.* 1999; Barragan and Dutton 2000), and no tagging data (i.e., females nesting on both sides of the Pacific Ocean) contradict these data. Therefore, nesting beach philopatry, combined with the long distance of the East Pacific Barrier, results in marked separation of the East and West Pacific populations.

The Pacific and Indian Oceans are not separated by existing land masses or long distances. However, within the past 6,000 years, low sea levels have exposed land barriers between Indonesia, New Guinea, and the Philippines (Barber *et al.* 2000). The most recent expansion of the species (less than a million years ago) appears to have originated from this area, which is known as a cradle of biodiversity (Bowen *et al.* 2013). These exposed land barriers created sharp biogeographic breaks in marine communities, including Wallace's line that separates fauna from eastern and western Indonesia and Huxley's extension that runs through the Philippines (Barber *et al.* 2000). These previous land barriers likely created physical separation between leatherback nesting beaches, which remain isolated as a result of philopatric behavior. Genetic data are not available from all recently active nesting beaches. However, available data indicate significant genetic partitioning between West Pacific nesting beaches in Papua New Guinea, Indonesia, and the Solomon Islands and NE Indian nesting beaches in Malaysia and Little Andaman Island (Dutton *et al.* 2007; Shanker *et al.* 2011; Dutton and Shanker 2014). Seasonal monsoons may also play a role by altering current directions and hatchling dispersal patterns (Benson *et al.* 2011; Gaspar *et al.* 2012). Therefore, the NE Indian and West Pacific populations appear to be reproductively isolated as a result of previous land barriers and contemporary philopatry.

Within the Indian Ocean, nesting sites are separated by distance (e.g., approximately 5,000 km between Mozambique and Sri Lanka) and ocean currents. The SW and NE Indian populations overlap at foraging areas in the SW Indian Ocean (Namboothri *et al.* 2012a). However, males and females are philopatric, at least regionally. Such large distances between Indian Ocean nesting sites likely results in reproductive isolation: while females have been shown to nest in locations separated by hundreds of kilometers, we have no data to support internesting over thousands of kilometers. Furthermore, the Somali Current runs between the distant Indian Ocean nesting sites. While leatherback turtles are strong swimmers, their movements often follow the currents or eddies they encounter off the nesting beaches (Luschi *et al.* 2003b; Luschi *et al.* 2006; Lambardi *et al.* 2008; Robinson *et al.* 2016). This is likely mediated through hatchling dispersal and subsequent adult fidelity to foraging destinations (Fossette *et al.* 2010b; Seminoff *et al.* 2012). Genetic data support the reproductive isolation between the South African and Malaysian nesting aggregations (Dutton *et al.* 1999). Therefore, the NE and SW Indian populations appear to be reproductively isolated as a result of distance and philopatry.

Indian and Atlantic Ocean nesting sites are separated by the SE coast of Africa. The fast-moving Benguela and Agulhas Currents meet at the Cape of Good Hope. The tropical Agulhas Current carries warm Indian Ocean water to the southwest, whereas the cold Benguela Current moves northward along the Atlantic coast. Both current systems provide suitable foraging habitat, but only the Agulhas Current provides the warm temperatures needed for nesting. Adults of the SW Indian population are known to cross from the Agulhas Current to the Benguela Current to forage in SE Atlantic waters (Luschi *et al.* 2006; Lambardi *et al.* 2008; Robinson *et al.* 2016), where they overlap with populations from the South Atlantic Ocean. However, they appear to return to the waters off the SW Indian Ocean nesting beaches to mate, creating a barrier to gene flow at the Cape of Good Hope (Dutton *et al.* 2013b). The cold-water upwelling has periodically subsided over evolutionary time periods (Peeters *et al.* 2004), and it is possible that leatherback turtles colonized the Atlantic Ocean via this route (Dutton *et al.* 2013b). Despite shared haplotypes, there is genetic partitioning between SE Atlantic and SW Indian populations (Dutton *et al.* 2013b). Therefore, the SW Indian and South Atlantic populations appear to be reproductively isolated as a result of cold ocean currents and philopatry.

The mid-Atlantic barrier is a stretch of open ocean, approximately 2,400 km in length, that separates the eastern and western Atlantic Ocean. Leatherback turtles are clearly capable of crossing this barrier. Telemetry and genetic studies demonstrate that females nest on one side of the South Atlantic and forage on the other (Almeida *et al.* 2011; Witt *et al.* 2011; Dutton *et al.* 2013b; Vargas *et al.* 2017). However, significant genetic partitioning occurs between SE and SW Atlantic nesting sites, indicating a lack of gene flow (Dutton *et al.* 2013b; Vargas *et al.* 2017). It is likely that while leatherback turtles of the South Atlantic Ocean share foraging areas, they do not overlap during the breeding season, when males and females return to the waters off their nesting beaches to mate. Thus, philopatry across the Atlantic Ocean separates the SE and SW Atlantic populations.

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 advected 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 Ocean (Fossette *et al.* 2010a; Chambault *et al.* 2017). Stewart *et al.* (2016) performed a genetic assignment test on 397 leatherback turtles incidentally captured in the U.S. pelagic longline fishery (i.e., the North Atlantic Ocean). They did not genetically assign any individuals to the SE Atlantic population; however, they assigned one individual to the SW Atlantic population (Stewart *et al.* 2016). Females nesting in Brazil primarily forage along the southeastern coast of Brazil (Lopez-Mendilaharsu *et al.* 2009), although some cross the South Atlantic Ocean to forage off the coast of Africa (Almeida *et al.* 2011). As with other leatherback populations, gene flow appears restricted ontogenetically, because turtle movements are influenced by ocean currents encountered when hatchlings depart the nesting beaches. Genetics show that the NW, SW, and SE Atlantic nesting populations are significantly differentiated (Dutton *et al.* 2013b; Vargas *et al.* 2017). Therefore, the NW, SW, and SE Atlantic populations appear to be reproductively isolated as a result of ocean currents and philopatry.

3.1.3 Ecological and Physiological Factors (Morphological Discontinuity)

Leatherback populations are recently derived, originating from a common lineage within the past one million years (Dutton *et al.* 1996; Dutton *et al.* 1999; Bowen and Karl 2007). As such, we would not expect large morphological differences, which may take millions of years to evolve. However, ecological and physiological factors may result in differential age and growth among populations. To address this possibility, we explored the best available age and growth data.

Somatic growth patterns (including the age and size at which individuals mature) are fundamental to understanding population dynamics and potential effects of management actions. However, collecting such data for in-water life stages of sea turtles is challenging. Leatherback turtles are especially challenging because they forage and migrate in remote, open-ocean areas. Because we cannot directly measure somatic growth rates for all life stages in the wild, inference must be made through indirect approaches. As a result, few age and growth data have been reported for leatherback turtles and existing estimates range widely, limiting the usefulness of this information (Avens *et al.* 2009; Wallace and Jones 2015).

Observation of vascularized cartilage in leatherback turtles, an anatomical feature corresponding with rapid growth in avian species, was thought to support rapid growth and possible maturation within a proposed 2- to 6-year time frame (Rhodin 1985). Captive growth studies have demonstrated the capacity for rapid early juvenile leatherback growth, exceeding that demonstrated for Cheloniid sea turtles (reviewed by Jones *et al.* 2011; Avens and Snover 2013). Early juvenile growth was reported as extremely rapid (averaging 32 cm annually) and growth models indicated that leatherback turtles might reach mean minimum nesting carapace lengths at around 16 years (range 13 to 20 years).

Estimation of generation time has also been used to indirectly infer age at maturation for a rapidly-expanding nesting population on St Croix. DNA fingerprinting analysis of tissue samples collected from nesting females provided information about potential mother-daughter relationships (Dutton *et al.* 2005). Together with the timing of observed increases in nesting females relative to conservation efforts and mark-recapture data, genetic data suggested that age at maturation for females in the population could be 12 to 14 years (Dutton *et al.* 2005).

In a study involving Pacific leatherback turtles, Zug and Parham (1996) evaluated skeletal growth marks in scleral ossicles, which are small bones that form a ring around the pupil of each eye in sea turtles. Counts of marks at the lateral edges of scleral ossicle sections were used to estimate age at stranding and fitting growth curves to the resulting data yielded mean age estimates of 13 to 14 years for adult-sized turtles in the eastern South Pacific Ocean. However, within the scope of that study, it was not possible to address the underlying assumptions for skeletochronological analysis, increasing uncertainty for the resulting age estimates. Subsequently, Avens and Goshe (2007) and Avens *et al.* (2009) conducted validation of annual mark deposition in Cheloniid and leatherback scleral ossicles from the NW Atlantic Ocean and found that mark retention and clarity was greater at the tips of ossicle sections than at lateral edges. Analysis of marks at scleral ossicle tips from leatherbacks in the NW Atlantic Ocean was then used to estimate age at stranding for each turtle and growth models were fit to the resulting size-at-age data (Avens *et al.* 2009). Age range estimates for carapace lengths associated with minimum and mean initial size of nesting females for this population were 16 to 22 and 24.5 to 29 years (Avens *et al.* 2009). However, while these data offered potential insight into how old adult-sized leatherback turtles might be at the time they died (Avens *et al.* 2009), data describing ages and sizes at which turtles actually attained maturity were still lacking (Wallace and Jones 2015).

Avens *et al.* (2020) analyzed Pacific samples using skeletal growth mark counts from section tips so that each mark could be assigned a corresponding age estimate, comparable to existing Atlantic data (Avens *et al.* 2009). Both Pacific and NW Atlantic samples were then evaluated to identify the transition skeletal growth mark potentially corresponding with maturation (Avens *et al.* 2015, 2017). A proportional relationship was also established between the radius measured for each ossicle used for analysis and carapace length of the turtle from which the sample was collected, allowing estimation of size associated with each measurable growth mark. Avens *et al.* (2020) then used Generalized Additive Mixed Models (GAMM) that could account for multiple data points for individual turtles to fit-smoothing splines to the size-at-age data to characterize the mean relationship and 95 percent credible intervals for the two sample populations.

As part of the updated skeletochronological analysis, for those samples retaining a measurable core mark denoting time of hatching, it was possible to estimate growth during the first year of life. Rates calculated for the NW Atlantic population were 39.3 cm CCL per year (range 18.3 to 50.2; n = 13), which were greater than those estimated for East Pacific turtles at 18.6 cm CCL per year (range 12.5 to 30.2; n = 5). However, this discrepancy could result from differences in time frames encompassed by the two data sets and/or the smaller East Pacific sample size. Furthermore, based on models fit to growth trajectories for captive NW Atlantic juveniles, Jones *et al.* (2011) proposed an age estimate of 2.8 years for juveniles of approximately 70 cm SCL. Similarly, GAMM results from the updated skeletochronological analysis yielded a mean age estimate of 3 years for wild NW Atlantic juveniles 73 cm CCL and 5 years for wild Pacific juveniles at the same size (Avens *et al.* 2020).

For both NW Atlantic and East Pacific leatherback turtles, minimum and mean age and CCL at maturation were estimated using the values assigned to the transition growth mark denoting a decrease in spacing and inferred from the GAMM smoothing spline fit to the size-at-age data (Avens *et al.* 2020). The NW Atlantic minimum maturation CCL (i.e., size at maturity) was 112.8 cm and corresponded with age estimates of 12 to 13 years (95 percent CI = 10.5 to 13.5 years) and mean maturation CCL for the population of 129.2 cm was associated with age estimates of 19 to 19.5 years (95 percent CI = 17 to 21.5 years). Despite apparent differences in early juvenile growth trajectories, analysis of Pacific samples yielded similar results, with minimum maturation CCL and ages of 110.7 cm and 11.5 to 12 years (95 percent CI = 10 to 14 years) and mean maturation CCL and ages of 129.7 cm and 17 to 20 years (95 percent CI = 17 to 24 years; Avens *et al.* 2020).

Identification of the putative maturation growth mark also allowed estimation of adult stage duration through counts of marks deposited at the tips of scleral ossicles subsequent to the transition mark. Mean longevity after maturation (i.e., reproductive longevity) was estimated to be 11 years for NW Atlantic turtles (range 3 to 22 years) and 8 years for Pacific turtles (range 3 to 18 years) for the disparate time frames represented by the two sample sets (Avens *et al.* 2020). The upper ends of ranges for these estimates are consistent with observed tag returns over maximum spans of 10 to 19 years for nesting females of different populations around the world (reviewed by Eckert *et al.* 2012) but lower than recent maxima of 31 years in St. Croix, USVI (K. Stewart, NMFS, and C. Lombard, USFWS, pers. comm., 2019).

From these studies of age and growth, we conclude that leatherback turtles from the East Pacific and NW Atlantic Oceans have the potential to attain maturity at similar minimum (111 to 113 cm CCL) and mean (approximately 130 cm CCL) carapace lengths. These sizes at maturation also correspond with similar estimates of possible minimum (12 years) and mean (17 to 19 years) ages at maturation for the two populations. This result is interesting, as females nesting in the East Pacific, where the majority of the Pacific samples originated, are on average smaller than those of other populations including the NW Atlantic (Saba *et al.* 2015) and these two populations are the most divergent genetically (Dutton *et al.* 1999; Duchene *et al.* 2012), suggesting the possibility of regional differences in age at maturation. Detection of potential differences may have been hindered by (1) integration of individuals from western and eastern populations in the Pacific sample and (2) temporal discrepancy between collection of Pacific and NW Atlantic samples. However, based on the currently available skeletochronlogy data, we find no discontinuity in potential age and size at maturation among the analyzed populations.

3.1.4 Summary of Discreteness Analyses

As discussed above, the main factor separating these populations is behavior: leatherback turtles return to the waters off their natal nesting beaches to mate. This results in reproductive isolation between distant nesting beaches, which are separated by physical features, such as land masses, oceanographic features, and currents. The best available data provide evidence of this separation, as indicated by significant genetic discontinuity among the seven populations (Dutton *et al.* 1999; Dutton *et al.* 2007; Wallace *et al.* 2010a; Dutton *et al.* 2013b). Tagging and telemetry data indicate overlap among foraging areas; however, nesting sites remain discrete and isolated. We did not find evidence for morphological discontinuity as a result of physiological or ecological factors; however, this may reflect the relatively recent divergence of the populations. Without gene flow to connect them, each of the seven populations is evolving on an independent trajectory (Wallace *et al.* 2010a). We conclude that the leatherback turtle comprises seven discrete populations, markedly separated from each other as a result of behavioral and physical factors:

- NW Atlantic
- SW Atlantic
- SE Atlantic
- SW Indian
- NE Indian
- West Pacific
- East Pacific

3.2 Significance

Because we considered each population segment to be discrete, the DPS Policy requires us to consider their biological significance to the species. The DPS Policy states that a population segment may be considered biologically or ecologically significant if it is important to the species; this consideration may include, but is not limited to:

1. Evidence that loss of the discrete population segment would result in a significant gap in the range of a taxon;
2. Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon;
3. Evidence that the discrete population segment 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 discrete population segment differs markedly from other populations of the species in its genetic characteristics.

We did not consider the third factor because leatherback turtles have not been introduced outside of their current range. For each population segment, we considered whether the other factors rendered it biologically or ecologically significant to the species. The following information provides more details regarding the significance of each population, as summarized in Table 1.

Table 1. Summary of each population's significance to the species.

Population	Loss would result in significant gap in range	Unique or unusual ecological setting	Markedly different genetic characteristics
NW Atlantic	Gap in nesting range of species throughout NW Atlantic Ocean; gap in foraging range of species throughout North Atlantic Ocean.	Highest latitude foraging. Most widely distributed nesting, geographically.	Significantly contributes to the genetic diversity of species; local adaptations likely exist but yet to be identified.
SW Atlantic	Gap in nesting range of species throughout SW Atlantic Ocean.		Significantly contributes to the genetic diversity of species; local adaptations likely exist but yet to be identified.
SE Atlantic	Gap in nesting range of species throughout SE Atlantic Ocean.		Significantly contributes to the genetic diversity of species; local adaptations likely exist but yet to be identified.
SW Indian	Gap in nesting range of species throughout SW Indian Ocean.	Highest latitude (temperate) nesting.	Significantly contributes to the genetic diversity of species; local adaptations likely exist but yet to be identified.
NE Indian	Gap in nesting range of species throughout NE Indian Ocean.		Significantly contributes to the genetic diversity of species; local adaptations likely exist but yet to be identified.
West Pacific	Gap in nesting range of species throughout West Pacific Ocean.	Summer and winter nesting seasons; foraging in both hemispheres.	Significantly contributes to the genetic diversity of species; local adaptations likely exist but yet to be identified.
East Pacific	Gap in nesting range of species throughout East Pacific Ocean.	Smallest nesting females, eggs, and clutch size, which may reflect unique foraging conditions.	Significantly contributes to the genetic diversity of species; local adaptations likely exist but yet to be identified.

Females nesting on beaches of the NW Atlantic forage in the North Atlantic Ocean. The loss of this population would result in a gap of the nesting and foraging range of the species (the entire North Atlantic Ocean). 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 (tens to hundreds of years), leaving a significant gap in the range of the species. The extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by its possession of several unique haplotypes. Local adaptations likely exist, as a result of thousands of years of reproductive isolation, but have yet to be identified (because all genetic studies have involved neutral markers). Leatherback turtles of the NW Atlantic Ocean also occur in a unique ecological setting: they regularly forage at higher latitudes than any other population. 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. Leatherback turtles of the NW Atlantic population also exhibit the most diverse nesting habitat along both islands and mainland beaches, as well as the most widespread and numerous nesting beaches. We conclude that the NW Atlantic population is biologically significant to the species.

In the SW Atlantic Ocean, leatherback turtles only nest along a small area of the SE coastline of Brazil. All other known, regular nesting sites in South America occur above the Equator (and these turtles forage in the North Atlantic Ocean) or on the Pacific Coast. Therefore, the loss of this population would result in a gap of the nesting range of the species (the SW Atlantic coast). Though SE Atlantic leatherback turtles forage off the coasts of Brazil, Argentina, and Uruguay, they do not breed there, but rather 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 its small population size (Vargas *et al.* 2017). Local adaptations likely exist, as a result of thousands of years of reproductive isolation, but have yet to be identified (because all genetic studies have involved neutral markers). SW Atlantic leatherback turtles also occur in a unique ecological setting: they forage in the Rio de la Plata estuary, which provides a stable, highly productive foraging area near Brazilian nesting beaches (Miazan *et al.* 2001; Prosdocimi *et al.* 2014; Lopez-Mendilaharsu *et al.* 2009). We conclude that the SW Atlantic population is biologically significant to the species.

Leatherback turtles of the SE Atlantic population nest in West Africa and forage throughout the South Atlantic Ocean. This population includes a large nesting aggregation in Gabon that is two orders of magnitude higher 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 (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 its possession of unique haplotypes. Local adaptations likely exist, as a result of thousands of years of reproductive isolation, but have yet to be identified (because all genetic studies have involved neutral markers). We conclude that the SE Atlantic population is biologically significant to the species.

In the SW Indian Ocean, leatherback turtles only nest along the shared South African and Mozambican coastlines. Leatherback turtles do not nest in eastern Africa or in other areas throughout the western Indian Ocean. Therefore, the loss of this population would result in a gap of the nesting range of the species (the SW Indian Ocean). The extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by its possession of unique haplotypes. Local adaptations likely exist, as a result of thousands of years of reproductive isolation, but have yet to be identified (because all genetic studies have involved neutral markers). SW Indian leatherback turtles also occur in a unique ecological setting: they nest at higher latitudes than any other population. The warm Agulhas Current, adjacent to the nesting beaches, likely facilitates their temperate nesting. We conclude that the SW Indian population is biologically significant to the species.

Leatherback turtles nest in small numbers throughout islands in the NE Indian Ocean. These nesting sites are separated from SW Indian Ocean nesting sites by at least 5,000 km. Though western Pacific nesting sites are closer, male and female philopatry prevents interbreeding. Therefore, the loss of this population would result in a gap of the nesting range of the species (the NE Indian Ocean). The extirpation of this population would also significantly reduce the genetic diversity of the species, as reflected by its possession of unique haplotypes. Local adaptations likely exist, as a result of thousands of years of reproductive isolation, but have yet to be identified (because all genetic studies have involved neutral markers). We conclude that the NE Indian population is biologically significant to the species.

West Pacific leatherback turtles nest in small numbers primarily in Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu. 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 (the West Pacific Ocean). The loss of this population would also result in a gap of the foraging range of the species (the North Pacific Ocean). The extirpation of this population would significantly reduce the genetic diversity of the species, as reflected by its possession of unique haplotypes. Local adaptations likely exist, as a result of thousands of years of reproductive isolation, but have yet to be identified (because all genetic studies have involved neutral markers). 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. We conclude 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 (the East Pacific Ocean). Though West Pacific leatherback turtles may forage off the coasts of Peru and Chile, they do not breed there (Dutton *et al.* 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 reflected by its possession of several unique haplotypes. Local adaptations likely exist, as a result of thousands of years of reproductive isolation, but have yet to be identified (because all genetic studies have involved neutral markers). 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., ENSO). We conclude that the East Pacific population is biologically significant to the species.

3.3 DPS Analysis Summary

As described above, we found seven leatherback populations that meet the DPS Policy definition for discreteness. These populations are markedly separated as a result of the behavioral factors of movement (as demonstrated by tracking and 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. These seven populations are significant to the species because the loss of any one of them would result in a significant gap in the range of the species and 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, we identified seven populations that meet the discreteness and significance criteria of the DPS Policy:

- NW Atlantic
- SW Atlantic
- SE Atlantic
- SW Indian
- NE Indian
- West Pacific
- East Pacific

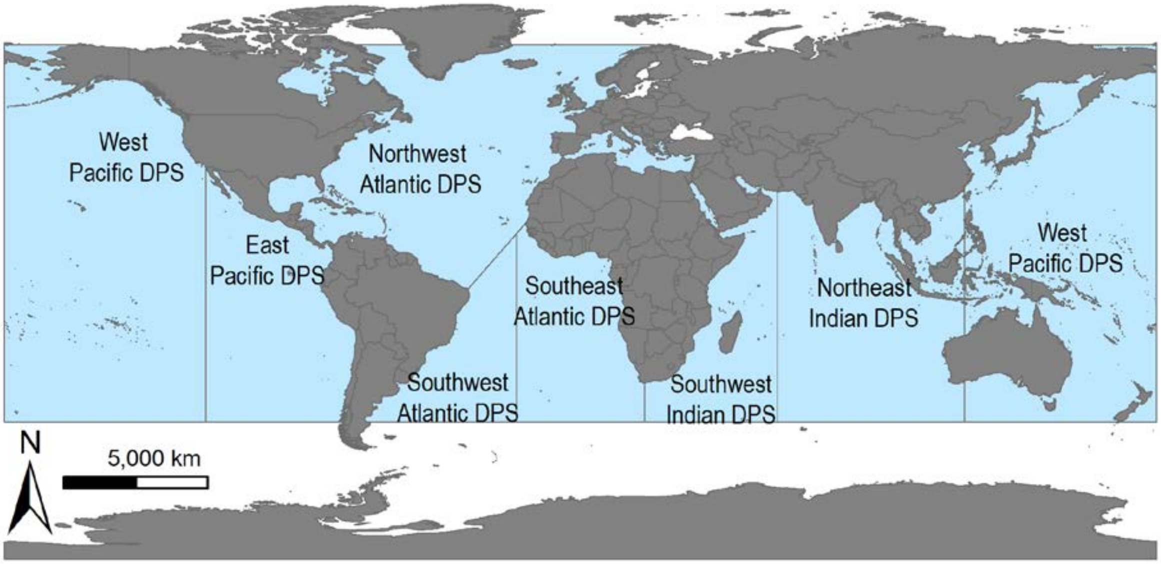
Under the ESA, the identification and status of a DPS is a legal determination, made by the Services and promulgated by regulation (16 U.S.C. 1533(a)(1)). For the remainder of this document, we refer to the seven populations as DPSs; however, we stress that the populations are not DPSs under the ESA unless identified and listed as threatened or endangered by the Services through official rulemaking, which would be published in the *Federal Register* and *Code of Federal Regulations*.

We defined the DPSs as leatherback turtles originating from nesting beaches within the boundaries of each DPS (Figure 8). The range of each DPS, which also includes foraging areas, thus extends beyond the DPS boundaries for most DPSs. We created the boundaries using the best available genetic, tracking, and observational data. When such data were not available, we used information on possible barriers to gene flow, such as oceanographic features. For ease of use, we created boundaries at existing political boundaries when this did not conflict with biological or oceanographic data. Additional information on the boundaries is available in the following sections for each DPS. Here we provide the definition for each DPS:

1. NW Atlantic DPS: 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
2. SW Atlantic DPS: 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
3. SE Atlantic DPS: leatherback turtles originating from the SE Atlantic Ocean, north of 47° S, east of 20° W, and west of 20° E; the northwestern boundary is a diagonal line between 12.084620° N, 20° W and 16.063° N, 16.51° W
4. SW Indian DPS: leatherback turtles originating from the SW Indian Ocean, north of 47° S, east of 20° E, and west of 61.577° E

5. NE Indian DPS: leatherback turtles originating from the NE Indian Ocean, south of 71° N, east of 61.577° E, and west of 120° E
6. West Pacific DPS: 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
7. East Pacific DPS: 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

Figure 8. Leatherback turtle DPS boundary map. Lines represent boundaries between DPSs.



4. General Information Applying to All DPSs

In this section, we provide information that applies to the entire species or several DPSs.

4.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the species.

4.1.1 Abundance

For each DPS, the indices of nesting female abundance range from under 100 to nearly 10,000 females. Because DPSs are discrete as a result of behavioral separation (nesting beach fidelity), genetic discontinuity, and physical isolation, abundance is most relevant at the DPS level. Species-level abundance is less relevant because gene flow and movement among DPSs are minimal.

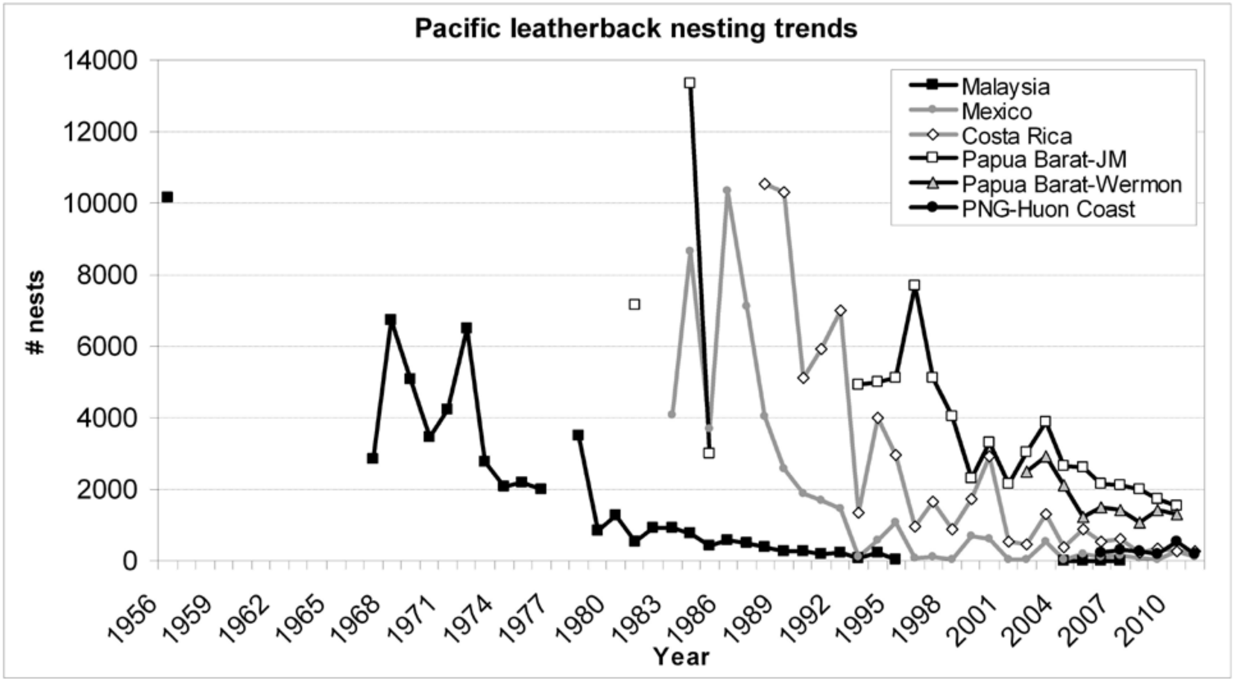
Within a DPS, long-term trends in nesting female abundance are complicated by the discovery of new nesting beaches over time, changes in remigration intervals and/or clutch frequency, and modified observational effort. Abundance estimates for even large nesting beaches were not available prior to 1950 (Rivalan *et al.* 2006), several large nesting beaches were not discovered until the 1960s or later (NMFS and USFWS 2013), and even when abundance estimates were available, the monitoring efforts were variable over time. Pritchard's (1971) global estimate of 29,000 to 40,000 nesting females assumed that large nesting aggregations had yet to be discovered. At that time, the nesting aggregation at Terengganu, Malaysia 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). Regardless, 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. In 2002, the nesting aggregation in Gabon was identified as the largest in the world at that time, with tens of thousands of nesting females (Witt *et al.* 2009), but 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.

4.1.2 Productivity

Species-level nest trends are uninformative because the DPSs are discrete; therefore, we must consider the nest trends of each DPS independently. All except the least abundant DPS (SW Atlantic) have exhibited declines in nests in recent years; however, it is not clear whether the declines represent the loss of nesting females or less frequent nesting activity.

Declines in nesting can occur rapidly in leatherback populations. For example, in the Pacific, nesting has declined precipitously in recent decades (Figure 9; Benson *et al.* 2015). Aerial surveys of nesting beaches in Mexico detected declines from 70,000 nesting females in 1982 to fewer than 250 in 1998, with an annual mortality rate of 22.7 percent (Spotila 2000). The Terengganu, Malaysia nesting population was reduced to less than one percent of its original size between the 1950s and 1995 (Chan and Liew 1996) and is now considered functionally extinct. Significant declines in nesting have been documented for other nesting aggregations, such as Gabon, French Guiana, and Indonesia.

Figure 9. Nest trends of leatherback turtles in the Pacific. Figure 10-2 from Benson *et al.* (2015); used with permission from S. Benson, NMFS, 2019.



Productivity metrics vary among DPSs and are reviewed by Eckert *et al.* (2012). For each DPS, we described the available productivity metrics, which were generally available only from consistently monitored nesting beaches. Important metrics include remigration interval, clutch frequency, clutch size, and hatching success (i.e., the percentage of eggs within a clutch that completed development and pipped from the egg).

4.1.3 Spatial Distribution

The species occurs over a broad spatial range, in tropical and temperate waters worldwide, from 71° N to 47° S (Figure i; Goff and Lien 1988; Carriol and Vader 2002; McMahon and Hayes 2006; Shillinger *et al.* 2008; Wallace *et al.* 2010a; Benson *et al.* 2011; Eckert *et al.* 2012). The DPSs are reproductively isolated with little to no gene flow connecting them. However, within some DPSs there is fine-scale population structure (Figure 3; also see Dutton *et al.* 1999; Dutton *et al.* 2003; Dutton *et al.* 2013b; 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.* 2013b). However, the reduction of nesting aggregations within a DPS has likely reduced or removed this structure, and the associated resilience, in some DPSs.

4.1.4 Diversity

Relative to other sea turtle species, the leatherback turtle has low genetic diversity and shallow mtDNA coalescence (Dutton *et al.* 1999). As a species, it uses diverse and widely distributed nesting and forage areas. Differences in size at maturity, remigration interval, 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 likely indicate that the species has some resilience to stochastic and environmental changes.

4.2 ESA Section 4(a)(1) Factors

The following information addresses the threats that are common to most or all DPSs.

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

Most DPSs are threatened by the destruction or modification of their nesting habitat, as a result of coastal development or erosion. By the year 2025, the United Nations (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 ocean. This is likely to result in increased development and erosion of coastal habitats.

Coastal development and associated activities cause accelerated erosion rates and interruption of natural shoreline migration (National Research Council 1990). Numerous nesting beaches are eroding due to both natural (e.g., storms, sea level changes, waves, shoreline geology) and anthropogenic (e.g., coastal development, construction of armoring structures, groins, jetties, marinas, inlet dredging) factors. Such shoreline erosion leads to a loss of nesting habitat for leatherback turtles and loss of nests from inundation. Erosion or inundation and accretion of sand above incubating nests appear to be the principal abiotic factors that may negatively affect incubating egg clutches in some areas (USFWS 1999; Dow *et al.* 2007; NMFS and USFWS 2013). Shoreline structures can also physically prevent females from reaching suitable nesting habitat or prevent post-nesting females and hatchlings from reaching the sea (Witherington *et al.* 2011).

Low hatching success is characteristic of many leatherback populations despite high fertility rates (reviewed by Bell *et al.* 2003; Eckert *et al.* 2012), and nest relocation occurs 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 influence the sex ratio of hatchlings produced (Sieg *et al.* 2011).

Coastal development and expansion also contribute to habitat degradation via artificial lighting (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; Nelson Sella *et al.* 2006; Witherington *et al.* 2014). As hatchlings head toward lights or meander along the beach, their exposure to predators and the 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.

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

Historically, the harvest of turtles and eggs (i.e., the direct killing of turtles and collection of eggs for consumption or other human use) was the known primary threat to the species (Congdon *et al.* 1993; Spotila *et al.* 1996). 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 some DPSs, legal harvest of turtle and eggs continues (Humber *et al.* 2014), as reviewed in the sections on the NW Atlantic, NE Indian, and West Pacific DPSs. Despite laws in many countries, the poaching of eggs continues in locations worldwide. Nesting females, and turtles caught at sea, continue to be poached for their meat and fat in many locations (Eckert *et al.* 2012).

4.2.3 Disease and Predation

Little is known regarding the effects of disease on the species, but we provide information on this threat when it is available for each DPS. Numerous species, both native and introduced, 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, (*Dorylus spininodis* and unidentified species), ghost crabs (*Ocypode spp.*), monitor lizards (*Varanus niloticius*), crows (*Corvus albus*), mongoose *(Atilax paludinosus)*, domestic and feral dogs (*Canis familiaris*), and feral pigs (*Sus domesticus*; Eckert *et al.* 2012). For hatchlings, common predators include the terrestrial predators listed above as well as numerous species of carnivorous fish. In addition, there have been reports of sharks and killer whales preying on subadult and adult leatherback turtles.

4.2.4 Inadequacy of Regulatory Mechanisms

Numerous regulatory mechanisms have been promulgated to protect sea turtles at international, regional, national, and local levels. For example, the harvest of sea turtles and their eggs is prohibited by regional conventions and national laws, as summarized below and in the sections for each DPS. Fisheries bycatch is also addressed, although not comprehensively, by several international and national instruments and/or governing bodies.

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 prohibiting the harvest of turtles and eggs are inadequate due to a lack of enforcement. In some areas, the regulation of fisheries bycatch is inadequate to sufficiently reduce associated mortality. Fishery observer coverage is often inadequate to accurately estimate leatherback bycatch. Details and examples are provided in the sections on each DPS.

Given their worldwide distribution and highly migratory nature, combined with nesting site fidelity, leatherback turtles require coordinated international, national, regional, and local protection. We discuss relevant regulations, treaties, conventions, and agreements in the DPS-specific sections.

4.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

Other threats that affect all DPSs include fisheries bycatch, pollution, and climate change. Additional threats are identified in the sections on each DPS.

4.2.5.1 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.* 2013a; Angel *et al.* 2014). Leatherback turtles are susceptible to bycatch in a wide range of fisheries, from large scale commercial to artisanal. Gear types include: longlines, purse seines, 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).

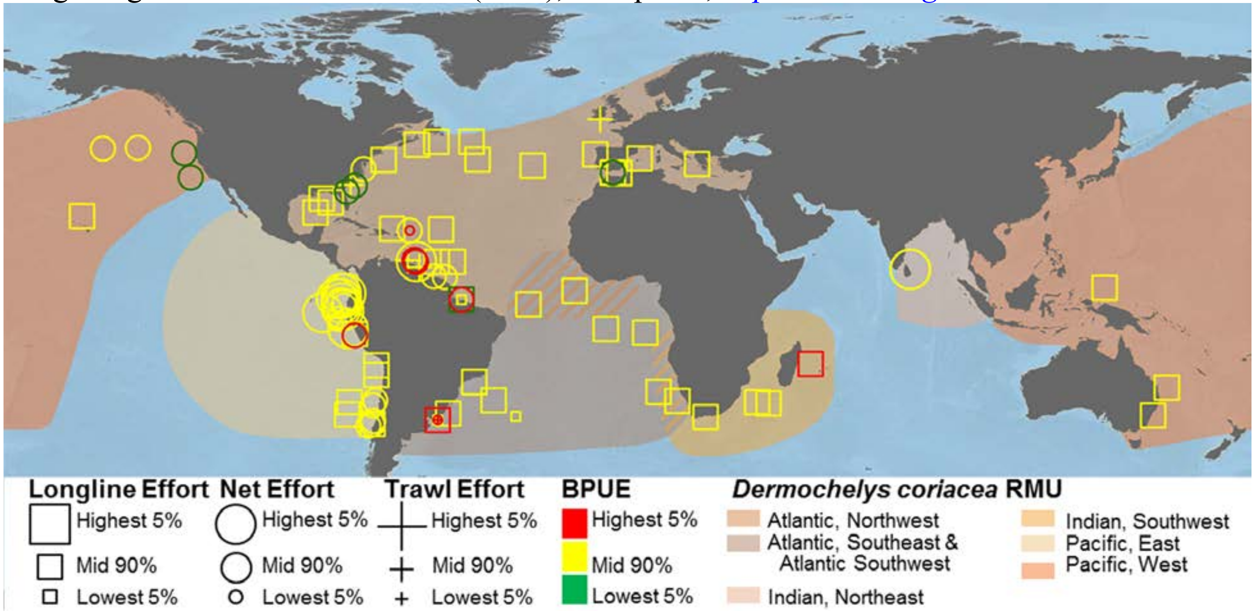
Bycatch data are most commonly collected by trained observers on fishing vessels or via surveys or interviews (Lewison *et al.* 2015). Though often the best available data on bycatch, observer data generally cover less than five percent of fisheries' total effort (Finkbeiner *et al.* 2011; Clarke *et al.* 2014) and are rarely available for small-scale fisheries (Wallace *et al.* 2013a; 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, illegal, unreported, and unregulated (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, valued at $10 to 23 billion U.S. dollars (USD; *http://www.fao.org/iuu-fishing/en/*). We have no estimates of impacts to leatherback turtles, though interaction and mortality rates are likely high because of the magnitude of this additional fishing pressure and because it is unregulated.

Leatherback turtles typically do not attempt to consume bait, as do other sea turtles, but more commonly become entangled in fishing gear (Lewison *et al.* 2015). Longline fisheries involve the deployment of a horizontal mainline 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 (and incidentally turtles) in a vertical wall of netting that can be closed at the bottom (*https://www.fisheries.noaa.gov/national/bycatch/fishing-gear-purse-seines*); generally, purse seine bycatch rates are lower than longline bycatch rates (Angel *et al.* 2014). Leatherback turtles also become entangled and drowned in gillnets, which include set and drift 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 major 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 turtles; though turtle excluder devices (TEDs) may mitigate this threat, they are not 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.

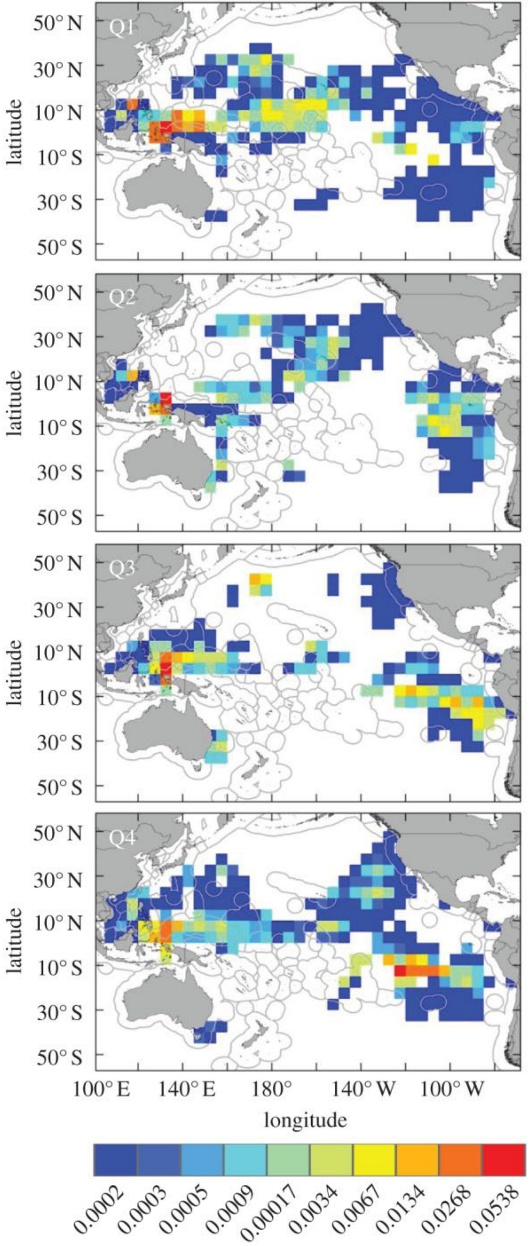
Longline and net fisheries are often the greatest threats to leatherback turtles. In a global study of sea turtle bycatch, Wallace *et al.* (2013) compiled data (n = 239 records) published between 1990 and 2011 to compare gear types (longline, net, and trawl) and their impacts to leatherback RMUs, which are similar to the DPSs identified in this Report, though their exact boundaries differ (Figure 10). 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.* 2013a).

Figure 10. Bycatch summary by gear type, effort, and BPUE for leatherback turtle RMUs.
Image: Figure 6 from Wallace *et al.* (2013), Ecosphere, *http:// dx.doi.org/10.1890/ES12-00388.1*.



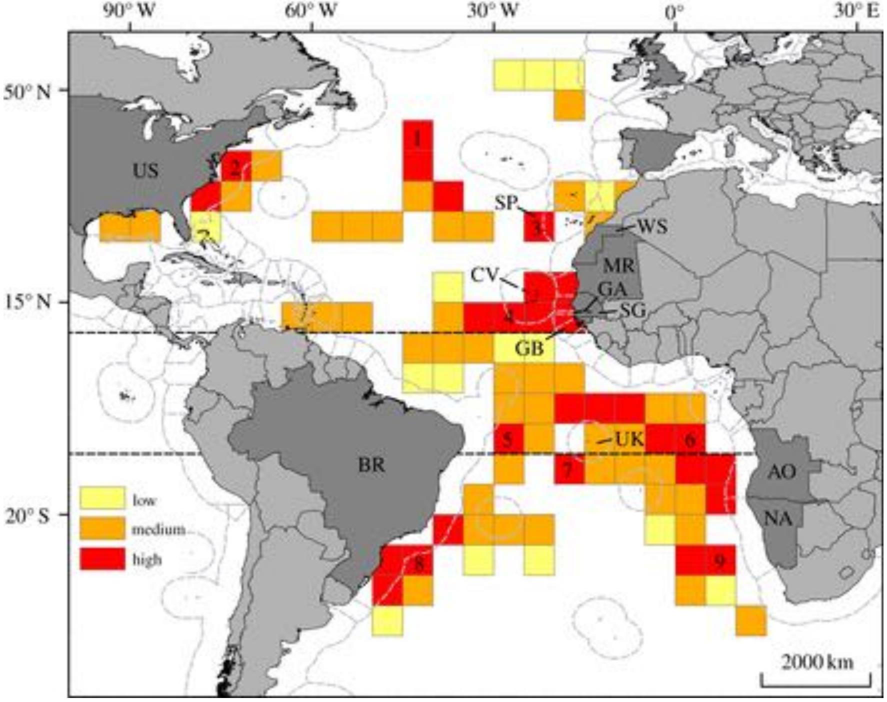
Integrating catch data from over 40 nations and bycatch data from 13 international observer programs, Lewison *et al.* (2004) estimated the numbers of leatherback turtles taken by pelagic longliners to be more than 50,000 leatherback turtles in one year (2000). Over half of the total fishing effort (targeting tuna, *Thunnus* spp., and swordfish, *Xiphias gladius*) occurred in the Pacific Ocean, where an estimated 20,000 leatherback turtles interacted with longline fishing, and 1,000 to 3,200 died in that year (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. The estimates of Beverly and Chapman (2007) may be more realistic, considering the abundance of Pacific leatherback turtles, and because they 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. Roe *et al.* (2014) predicted leatherback turtle bycatch hotspots in the Pacific Ocean, by comparing the satellite tracks of 135 adult turtles with longline fishing effort within 5° × 5° grid cells (Figure 11). 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. A review of Pacific observer data from 34 swordfish-targeting shallow-set longline fleets found 331 leatherback turtle interactions observed between 1989 and 2015 and identified bycatch hotspots in the central North Pacific Ocean and eastern Australia (Clarke 2017).

Figure 11. Exposure of Pacific leatherback turtles to bycatch in longline fisheries, by quarter. The quarters are represented from top to bottom; red boxes represent areas exposed to high fishing pressure. Image: Figure 3 from Roe *et al.* (2014), Proceedings of the Royal Society B, *https://doi.org/10.1098/rspb.2013.2559*.



In the Atlantic Ocean, Fossette *et al.* (2014) compared leatherback telemetry data to longline fishing effort data from the International Commission for the Conservation of Atlantic Tunas (ICCAT) to identify nine areas in which leatherback turtles are exposed to bycatch risk associated with high longline fishing pressure (Figure 12; Fossette *et al.* 2014). 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.

Figure 12. Exposure of Atlantic leatherback turtles to bycatch in longline fisheries. Red boxes represent areas exposed to high fishing pressure, with the main high-susceptibility areas numbered one through nine. Each nation's EEZ is enclosed in a grey, dashed line. Image: Figure 2 from Fossette *et al.* (2014), Proceedings of the Royal Society B, *https://doi.org/10.1098/rspb.2013.3065*.



In some areas, there is a significant lack of sea turtle bycatch information from coastal small-scale fisheries (Lewison *et al.* 2014). Trap and pot fisheries are generally understudied but evidence from the U.S. lobster fishery in the northeast United States indicates this gear is problematic for leatherback turtles (NMFS, unpublished data). While global data are lacking, leatherbacks are likely bycaught in this gear type wherever fishing effort and turtles overlap.

4.2.5.2 Pollution

We define pollution as including contaminants, marine debris, and ghost or derelict fishing gear. Information describing the impacts of contaminants on leatherback turtles is rare, and for most DPSs, we were only able to find information on contaminants associated with oil and gas activities, which include exploration, extraction, spills, and other associated activities. 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 Wallace *et al.* 2020). 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 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, leading to a case of mistaken identity and 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 (Juárez-Cerón 1998). 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. Marine debris affects leatherback turtles via ingestion or entanglement and can reduce food intake and digestive capacity, cause injury 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). The impact of marine debris on leatherback turtles is unquantified, but likely severe, given the increase of debris entering the marine environment over the past 30 years, which is approximately 5.2 to 19.3 million tonnes per year (Lebreton *et al.* 2018). Furthermore, the impact of marine debris may be underestimated because data on this threat generally come from stranded turtles, which likely represent a fraction of affected individuals. Ghost or derelict fishing gear include discarded or lost nets, line, and other gear. Ghost fishing gear can drift in the ocean, 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).

4.2.5.3 Climate Change

Climate change is an increasing threat to all DPSs. In its Fifth Assessment Report, the Intergovernmental Panel on Climate Change (IPCC 2014) stated that the globally averaged combined land and ocean surface temperature data has shown a warming of 0.85 °C over the period of 1880 to 2012. The mean rate of globally averaged sea level rise was 1.7 millimeters/year between 1901 and 2010, 2.0 millimeters/year between 1971 and 2010, and 3.2 millimeters/year 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 sea surface temperatures are expected to be larger in the tropics and Northern Hemisphere subtropics (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/year (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 due to:

(1) Warming temperatures and increased precipitation at nesting beaches affect reproductive output including hatching success, hatchling emergence success, and hatchling sex ratio;
(2) Sea level rise results in a reduction or shift in available nesting beach habitat, an increased risk of erosion and nest inundation, and reduced nest success (Fish *et al.* 2005; Fuentes *et al.* 2010; Fonseca *et al.* 2013);

(3) Changes in productivity affect the abundance and distribution of forage species, resulting in changes in the foraging behavior and distribution of leatherback turtles as well as changes in leatherback fitness and growth;
(4) 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; and
(5) Increased frequency and severity of storm events impact nests and nesting habitat, thus reducing nesting and hatching success.

There is uncertainty regarding the response of the species to climate change. Females may shift nesting geographically or temporally to compensate for warming sand conditions. Turtles could shift their foraging area or duration, based on prey availability. Despite such uncertainty, in recent years, the threats due to climate change have become apparent (Spotila *et al.* 2015), as described in the paragraphs below, and in later sections, specific to each DPS. Increasing sand temperatures alter the thermal regime of incubating nests, resulting in altered sex ratios and reduced hatching output (Hawkes *et al.* 2009). Like all sea turtles, 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) and temperatures over 32 °C are likely to result in death. Increases in precipitation might cool beaches thereby mitigating some impacts relative to increasing sand temperatures. 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.

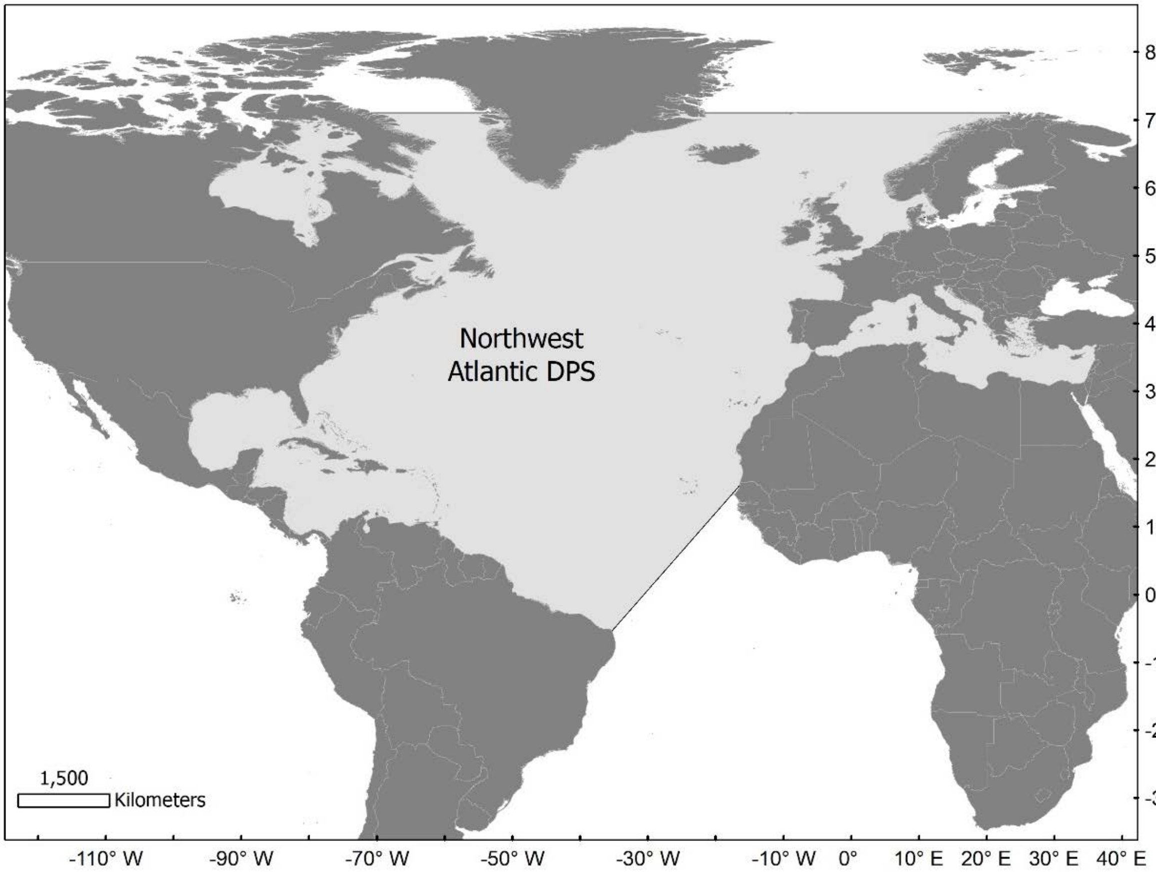
Beach erosion and nest inundation already threaten leatherback nesting habitat globally. Sea level rise is likely to increase the number of nests lost to erosion and inundation. This is especially problematic in areas prone to storm events, which are likely to increase in intensity and duration, and in areas where coastal development impedes natural shoreline migration.

Climate change is also likely to alter productivity in some marine environments, which could affect leatherback movement and prey availability. Models indicate that overheating presents a high risk to leatherbacks in Southeast Asia, a slight risk to leatherbacks in the West Atlantic and a low risk to leatherbacks in the East Atlantic (Dudley *et al.* 2016).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. In this document, we will focus on the climate changes impacts caused by increases in sand temperatures, sea level, and storm events.

5.0 Northwest Atlantic DPS

We define 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 (Figure 13). The northern boundary reflects a straight latitudinal line based on the northern-most occurrence of leatherback turtles (Brongersma 1972; Goff and Lien 1988; Carriol and Vader 2002; McMahon and Hays 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).

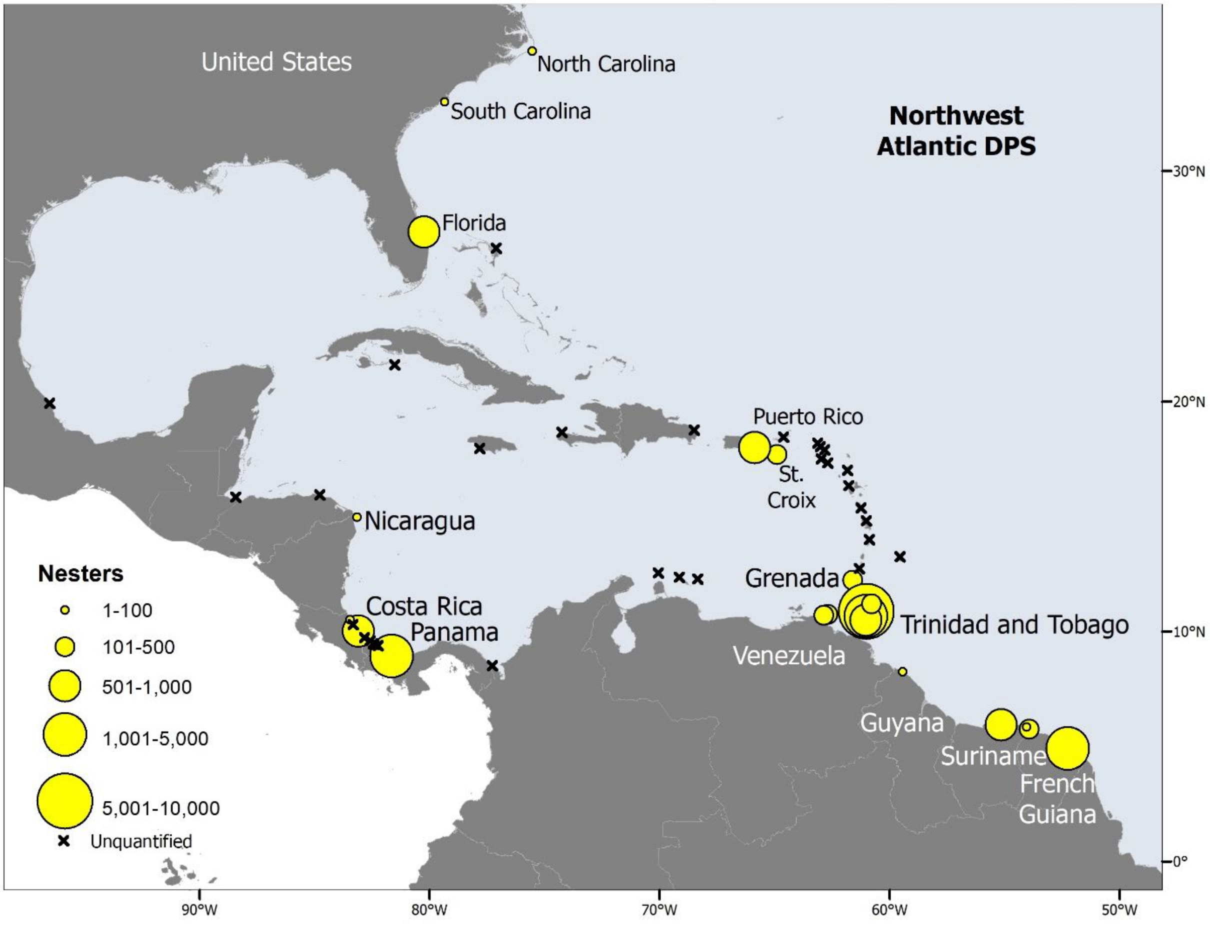
Figure 13. NW Atlantic DPS boundary map.



The range of this DPS includes all areas of occurrence, and extends throughout the North Atlantic Ocean, including the Caribbean Sea, 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, Puerto Rico, 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, USVI, Venezuela, and Western Sahara.

All nesting in this DPS occurs in the NW Atlantic Ocean (Figure 14), concentrated from the southeast United States throughout the Wider Caribbean Region (Dow *et al.* 2007). Leatherback nesting in the NW Atlantic may be grouped into several broad geographical areas, including the U.S. mainland (primarily Florida, but occasional nesting occurs in North and South Carolina), 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 most northern 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).

Figure 14. Nesting sites of the NW Atlantic DPS. Size of circle represents the index of nesting female abundance. An "X" indicates that nesting was documented but not quantified. Quantified nesting beaches are identified by nation (and State or Territory within the United States).



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 Balasingham 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 (review in 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 (reviewed by Eckert *et al.* 2012; Saba 2013; Shillinger and Bailey 2015). These waters include the GOM, North Central Atlantic Ocean, northwestern shelf waters of the United States and Canada, waters along the southeastern U.S. coast, the Mediterranean Sea, and northeastern shelf waters of Europe and northwestern Africa (TEWG 2007). Some post-nesting females also remain in tropical waters to forage (Fossette *et al.* 2010b). This DPS is mostly commonly associated with open ocean and coastal shelf foraging areas off Nova Scotia (Canada), the northeastern United States, GOM, northwestern Europe, and northwestern Africa (James *et al.* 2005a; James *et al.* 2006b; 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 (see Figure 5. Satellite tracking of 106 leatherback turtles in the 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 in the North Atlantic appear to have a diverse array of foraging habitat available.

5.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the NW Atlantic DPS.

5.1.1 Abundance

The total index of nesting female abundance for the NW Atlantic DPS is 20,659 females, based on the best available data, summarized in Table 2 and Table 5, which include unpublished nesting data graciously provided by numerous monitoring programs and published, recent nesting data. Table 3 also includes nesting beaches that were not included in our total index of nesting female abundance, largely because the data were not recent or available; nonetheless, we present the best available information for those nesting beaches to provide a comprehensive picture of all nesting areas for this DPS. For example, we included data published by Dow *et al.* (2007), who provided maps of each area surveyed, binned by crawls of less than 25, 25 to 100, 100 to 500, 500 to 1000, or unknown abundance; crawls represent 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). For some areas, published nesting data were available but did not meet our recent data criteria (i.e., last year of data in 2014 or more recently). Because these data may not be representative of recent nest trends, we did not include them in our total index of nesting female abundance and identified them as unquantified in Table 3. We were able to estimate the index of nesting female abundance for 24 nesting sites in 10 nations within the NW Atlantic DPS. To calculate these indices of nesting female abundance, we added the number of nests over the most recent remigration interval, which is approximately 3 years (see Table 4). Then, we divided by the average clutch frequency, 5.5 clutches per season (which represents the average of available NW Atlantic data) or site specific values if available (see Table 4).

Table 2. Available nesting data for the NW Atlantic DPS. We also include the highest and lowest number of recorded nests (or other units, as identified). We calculated the index of nesting female abundance by summing the number of nests over the most recent remigration interval (3 years; see Table 4) divided by the clutch frequency (5.5 clutches per season, the average of all available NW Atlantic sites, unless site specific values were available; Table 4). We also provide the index of nesting female abundance for each nation when available data met our criteria. We considered the nesting aggregations that did not meet our data criteria to be unquantified.

Nation, nesting beach, and length	Number of nests (first and last years monitored)	High and low number of nests (years)	Reference	Index of nesting female abundance
Colombia				Unquantified
Capitancito, Acandi, Playona, Playeta, Pueblo Nuevo, 24.6 km	1,639 (2006) 2,858 (2007)	High: 2,858 (2007) Low: 1,639 (2006)	Patiño-Martínez *et al.* 2008a	Unquantified
Costa Rica				1,306
Parismina, 6.2 km	243 (2012)	N/A	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	Unquantified

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (years)**	**Reference**	**Index of nesting female abundance**
Playa Pacuare, 7.4 km	375 (2004) 279 (2017)	High: 542 (2007) Low: 137 (2014)	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	(388+212+279)/)/5.5 = 160
Pacuare Nature Reserve, 7.2 km	380 (1991) 536 (2017)	High: 1,206 (2012) Low: 150 (1992)	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	(793+434+536)/5.5 = 321
Estacion La Tortuga, 2.65 km	419 (2002) 297 (2017)	High: 725 (2012) Low: 206 (2003)	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	(348+258+297)/5.5 = 164
Moín, 14.5 km	1,425 (2012) ~1,000 (2017)	High: 1,425 (2012) Low: ~1,000 (2017)	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	(1,242+1,000+1,000)/5.5 = 589
Playa Negra (Cahuita), 9 km	212 (2000) 267 (2012)	High: 350 (2001) Low: 73 (2004)	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	Unquantified
Playa Gandoca, 11 km	286 (1990) 288 (2012)	High: 1,135 (1997) Low: 226 (1992)	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	Unquantified
Playa Tortuguero, 24 km	1,623 (1995) 127 (2017) (estimated)	High: 1,623 (1995) Low: 111 (2014)	R. Valverde, STC, pers. comm., 2018	(154+116+127)/5.5 = 72
Dominica				Unquantified

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (years)**	**Reference**	**Index of nesting female abundance**
Rosalie, Bout Sable, Londonderry Beaches	278 (2007) 29 (2008)	High: 278 (2007) Low: 29 (2008)	Harris and George 2008	Unquantified
Dominican Republic				Unquantified
Jaragua National Park, 22.4 km	17 (2006) 134 (2010)	High: 210 (2009) Low: 17 (2006)	Revuelta *et al.* 2012	Unquantified
Saona Island, 15 km	1 (2007) 8 (2010)	High: 11 (2009) Low: 1 (2007)	Revuelta *et al.* 2012	Unquantified
French Guiana				2,519
Cayenne and Remire Montjoly, 7.5 km	947 (1999) 1,106 (2017)	High: 9,517 (2009) Low: 947 (1999)	KWATA data in Berzins 2018	(3,953+2,807+1,106)/3.55 = 2,216
Awala-Yalimapo, 2 km	3,445 (2002) 156 (2017)	High: 6,792 (2009) Low: 156 (2017)	Réserve Naturelle de l'Amana data in Berzins 2018	(483+437+156)/3.55 = 303
Grenada				566
Levera, St Patrick, 750 m	196 (2002) 623 (2017)	High: 1,529 (2014) Low: 196 (2002)	K. Charles, Oceans Spirits Inc., pers. comm., 2018	(1,237+1,252+623)/5.5 = 566
Guyana				76
Shell Beach Protected Area, 120 km	377 (2010) 45 (2017)	High: 377 (2010) Low: 45 (2017)	S. Edghill, WWF, pers. comm., 2018	(170+204+45)/5.5 = 76
Honduras				Unquantified

Nation, nesting beach, and length	Number of nests (first and last years monitored)	High and low number of nests (years)	Reference	Index of nesting female abundance
La Reserva del Hombre y la Biosfera del Río Plátano (RHBRP)	38 (1995) 35 (2015)	High: 50 (1999) Low: 2 (2003 and 2008)	PROTEP report 2015	Unquantified
Nicaragua				10
El Cocal, 36.1 km	132 (2009) 17 (2017)	High: 132 (2009) Low: 17 (2017)	C. Lagueux and C. Campbell, pers. comm., 2018	(19+17+17)/5.5 = 10
Panama				2,251
Chiriquí Beach, 24 km	2,922 (2004) 3,146 (2017)	High: 6,665 (2009) Low: 2,922 (2004)	C. Ordoñez, STC, pers. comm., 2018	(5,299+3,933+3,146)/5.5 = 2,251
Bluff Beach, Soropta Beach, Long Beach, Red Beach	2016 and 2017	Red Beach (18 in 2016); Long Beach (25 in 2016); Soropta Beach (691 in 2016 and 72 in 2017)	STC unpublished report 2018	Unquantified
Changuinola, 12 km	471 (2009) 546 (2013)	High: 601 (2012) Low: 259 (2011)	D. Chacón-Chaverri, Asociación LAST, pers. comm., 2018	Unquantified
Suriname				698
Galibi, 12 km	11,394 (2001) 72 (2017)	High: 11,394 (2001) Low: 72 (2004)	WWF Guianas, Stinasu (Foundation for Nature Conservation in Suriname), pers. comm., 2018	(134+109+72)/4.5 = 70

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (years)**	**Reference**	**Index of nesting female abundance**
Braamspunt, 1.5 km	3,588 (2001) 770 (2017)	High: 5,033 (2009) Low: 770 (2017)	2001 – 2011: WWF Guianas, Stinasu (Foundation for Nature Conservation in Suriname); 2012 – 2017: WWF and the Nature Conservation Division of the Ministry of Spatial Planning, Land, and Forest Management	(1,163+894+770)/4.5 = 628
Trinidad and Tobago				11,324
Grande Riviere, 1.2 km	17,485 (2009) 6,448 (2017)	High: 23,792 (2013) Low: 6,448 (2017)	Turtle Village Trust (2018)	(15,138+11,058+6,448)/5.5 = 5,935[1]
Matura, 8.8 km	4,791 (2009) 5,749 (2017) TVT; 7,237 (2006) 5,805 (2017) WIDECAST	High: 10,442 (2010) Low: 5,749 (2017) TVT; High: 19,109 (2007) Low: 4,766 (2012) WIDECAST	Turtle Village Trust (2018); Eckert and Mitchell (2018)	(7,097+9,581+5,749)/5.5 = 4,078 TVT[2]; (7,273+9,908+5,805)/5.5 =4,179 WIDECAST
Fishing Pond, 10 km	8,991 (2009) 1,396 (2017)	High: 8,991 (2009) Low: 1,396 (2017)	Turtle Village Trust (2018)	(2,473+2,013+1,396)/5.5 = 1,069[1]
Turtle Beach, Grafton Beach, and Mt. Irvine Back Bay, Tobago, 3.2 km	380 (2009) 284 (2017)	High: 641 (2013) Low: 284 (2017)	Turtle Village Trust (2018)	(422+623+284)/5.5 = 242[1]

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (years)**	**Reference**	**Index of nesting female abundance**
United States				1,694
Florida, 1,349 km	728 (2008) 663 (2017)	High: 1,747 (2009) Low: 663 (2017)	B. Brost, FWC, pers. comm., 2018	(1,493+1,054+663)/4.2 = 764
South Carolina, 303 km	0 (1980) 0 (2017)	High: 5 (2008) Low: 0 (2017)	SCDNR letter to NMFS/USFWS 2018	(2+0+0)/4.2 = 1
North Carolina, 531 km	4 (1998) 0 (2017)	High: 8 (2007) Low: 0 (2017)	NCWRC letter to NMFS/USFWS 2018	(0+0+0)/4.2 = 0
Sandy Point National Wildlife Refuge, St. Croix, 3 km	86 (1982) 120 (2017)	High: 1,008 (2001) Low: 82 (1986)	C. Lombard, USFWS, pers. comm., 2018	(382+216+120)/5.1 = 141
Puerto Rico (Rio Grande, Luquillo-Fajardo, Humacao, Maunabo, Yabucoa, Dorado, Aniasco, Arecibo, Isabela, Loiza-Piniones, Rincon, Mayaguez-Cabo Rojo, Barceloneta, Guanica, Manati, Hatillo, San Juan-Carolina, Culebra, Vieques), 70 km	1,385 (2011) 1,187 (2017)	High: 2,167 (2016) Low: 1,187 (2017)	C. Diez, PRDRNA, pers. comm., 2018	(1,845+2,167+1,187)/6.6 = 788
Venezuela				215
Cipara Beach, 2.06 km	102 (2000) 107 (2017)	High: 252 (2016) Low: 102 (2000)	H. Guada, IZET-UCV	(138+252+107)/4.4 = 113

Nation, nesting beach, and length	Number of nests (first and last years monitored)	High and low number of nests (years)	Reference	Index of nesting female abundance
			CICTMAR-WIDECAST, pers. comm., 2018	
Querepare Beach, 2.06 km	47 (2002) 99 (2017)	High: 254 (2008) Low: 47 (2002)	H. Guada, IZET-UCV CICTMAR-WIDECAST, pers. comm., 2018	(140+208+99)/4.4 = 102
Martinique, St. Kitts and Nevis		100 to 500 crawls annually	Dow *et al.* 2007	Unquantified
Anguilla, Aruba, Barbados, British Virgin Islands, Guadeloupe, St. Lucia		25 to 100 crawls annually	Dow *et al.* 2007	Unquantified
Antigua and Barbuda, Bahamas, Bonaire, Cuba, Guatemala, Mexico, Sint Eustatius, Sint Maarten		<25 crawls annually	Dow *et al.* 2007	Unquantified
Curacao, Haiti, Jamaica, St. Barthelemy, St. Vincent and the Grenadines		Unquantified	Dow *et al.* 2007	Unquantified

[1] Census data from Trinidad and tagging data from Tobago provide the best estimate of nesting activities for this nesting aggregation (TVT, pers. comm., 2018). These data were provided as extrapolated nest counts calculated by counting nesting females. The effort increased in 2013, by adding additional hours of monitoring each night, but this did not affect our index of nesting female abundance for these locations, which was based on data from 2014 to 2017.

[2] Two groups (Turtle Village Trust (TVT) and WIDECAST) provided nesting data for Matura Beach, Trinidad, so both numbers are presented here. The TVT nesting numbers were used in the Trinidad and Tobago country total and the DPS total, as the TVT index of nesting female abundance is lower than the WIDECAST estimate and thus more conservative to the species.

Table 3. The number of nesting sites by the index of nesting female abundance. We estimated the total index of nesting female abundance (for the DPS) by summing all indices of nesting female abundance from Table 2. We calculated the percentage at the largest nesting site by dividing that site's index of nesting female abundance (Table 2) by the total index of nesting female abundance for the DPS.

Index of nesting female abundance	**Number of nesting sites***
Unquantified**	31
1–10	3
11–50	0
51–100	3
101–500	8
501–1,000	5
1,001–5,000	4
5,001–10,000	1
10,001–20,000	0
>20,000	0
Total number of sites	55
Total index of nesting female abundance	20,659
Confidence in total index of nesting female abundance	Moderate (with the exception of Colombia, the largest aggregations are quantified, but many smaller ones are not)
Largest nesting site, percentage of total index	Grande Riviere, Trinidad, 29 percent

*Nesting sites are listed in Table 2. Sites may be represented by a whole country (with multiple nesting beaches included in one country) or sites may refer to individual beaches within a country.
**Sites included in Table 2 but not included in the total index for the DPS because recent data are not available over one remigration interval. These sites may host additional nesting females, but data are unavailable, not recent, or the survey methodology is unknown.

The total index of nesting female abundance for this DPS only includes available nesting data from recently and consistently monitored nesting beaches and assumes a 3-year remigration interval. In addition, for some sites, the clutch frequency was not based on data from that nesting beach, but rather the average for the DPS (5.5 clutches per season). Often, the nesting data provided to the Team only included the main nesting areas; data on scattered nesting were not provided and therefore not included in the index. Our total index does not include data from beaches where we were unable to calculate the index of nesting female abundance, either due to the lack of recent or available nesting data or because only crawl data were reported (often on smaller nesting beaches). Given these caveats and based on the data summarized in Table 2, our total index of nesting female abundance (20,659 females) represents the best available data for this DPS.

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 (Table 3). Relatively large nesting aggregations are also found in Matura, Trinidad, Chiriqui Beach, Panama, and Cayenne/Remire Montjoly, French Guiana (Table 2). In contrast, most known nesting beaches support a small number of nesting females; 71 percent of the total nesting sites record annual crawls of less than 100 (Dow Piniak and Eckert 2011). At the majority of nesting sites (n = 31), the number of nesting females is unquantified, 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). While remote beaches in western French Guiana hosted leatherback nesting up to 2013 (e.g., a high of 4,670 nests was found in 2003), more recent data were not available due to significant beach erosion and disappearance of previously monitored beaches.

As with other sea turtle species and other areas, the NW Atlantic DPS experiences high and low years in terms of the number of nests (Table 2). The difference between high and low years is large in some areas, and often reflects the decrease in nests over time. For example, on Galibi Beach in Suriname, a low of 72 nests was recorded in 2017 with a high number of 11,394 nests documented in 2001. For many beaches, the lowest number of nests was recorded in 2017.

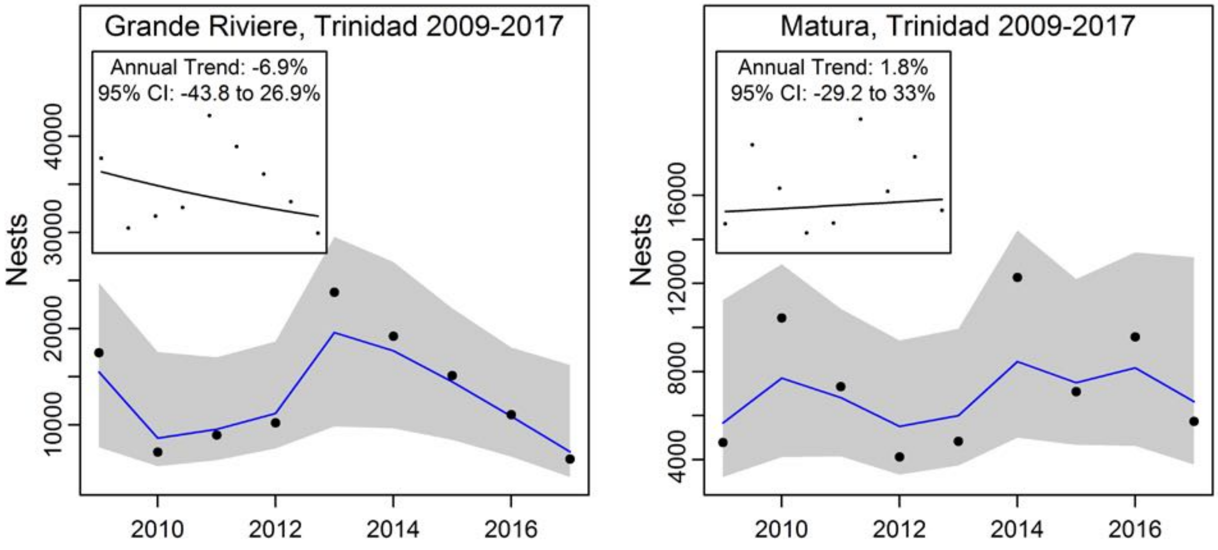
Our total index of nesting female abundance is similar in comparison to other published estimates. TEWG (2007) estimated the abundance of NW Atlantic leatherback turtles using nesting data from 2004 and 2005. At that time, the number of adult females (equating to nesting female abundance in our analysis) was estimated to be approximately 18,700 (range 10,000 to 31,000; TEWG 2007). 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 recently published IUCN Red List assessment for the NW Atlantic Ocean subpopulation estimated 20,000 mature individuals in 2019 (The Northwest Atlantic Leatherback Working Group 2019). This estimate is lower than our index, which only includes nesting females. The difference is likely a result of using a different remigration interval, which has been increasing in recent years (see Section 5.1.2 Productivity).

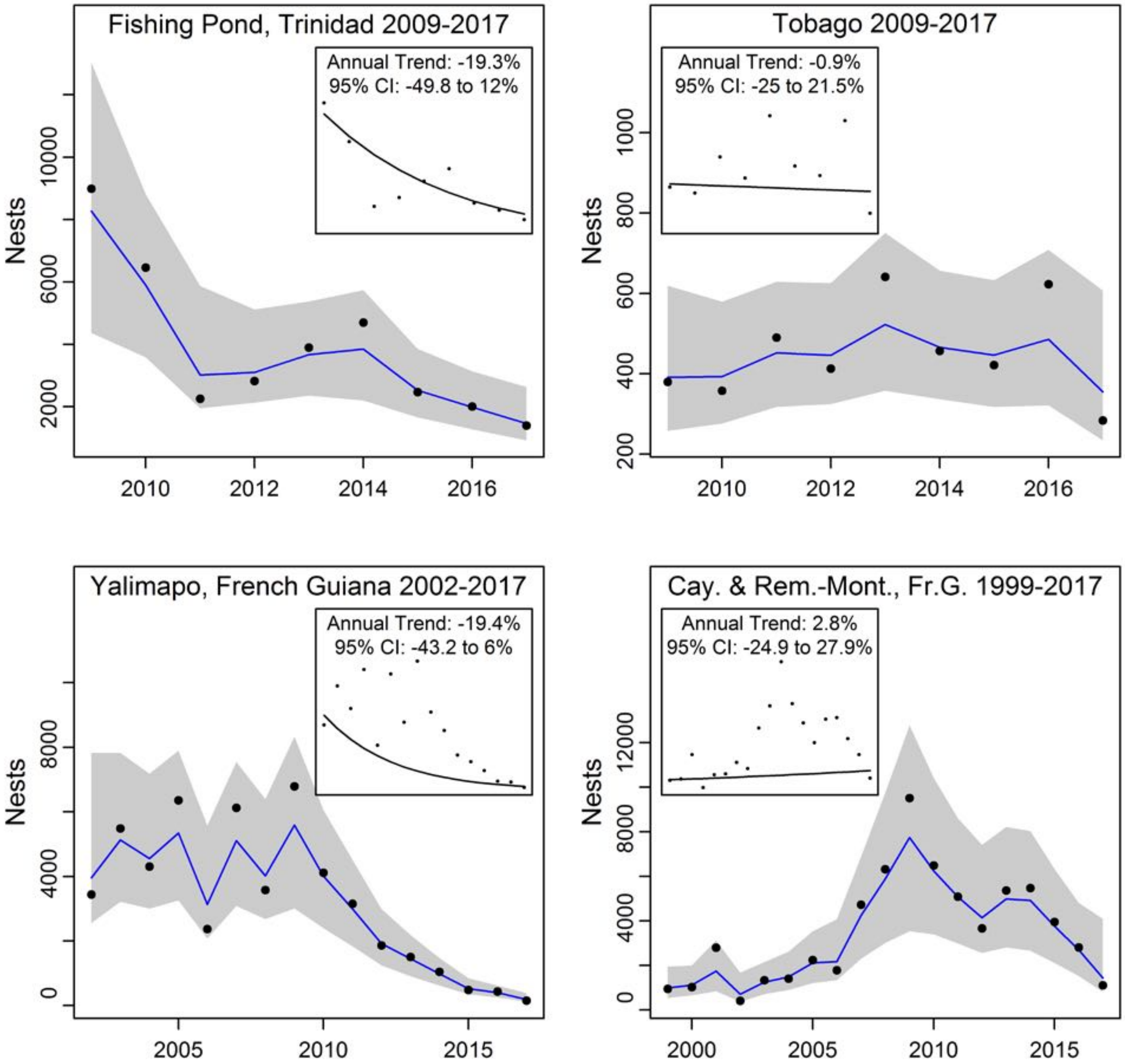
We conclude that the NW Atlantic DPS has a total index of nesting female abundance of 20,659 females. We have a moderate level of confidence in our total index of nesting female abundance because the nesting beaches with the greatest abundance have been included in our estimate, although many smaller beaches are not included. In addition, females nesting in Colombia, which hosted over 2,000 nests in 2007, were not included in our index because recent data were not available. Most beaches with an unquantified number of nests likely host few nesting females. While nesting female abundance is not at a level where stochastic changes would have catastrophic impacts, abundance at several nesting sites with previously high density has declined drastically. Widespread threats are likely to continue and further reduce the overall abundance.

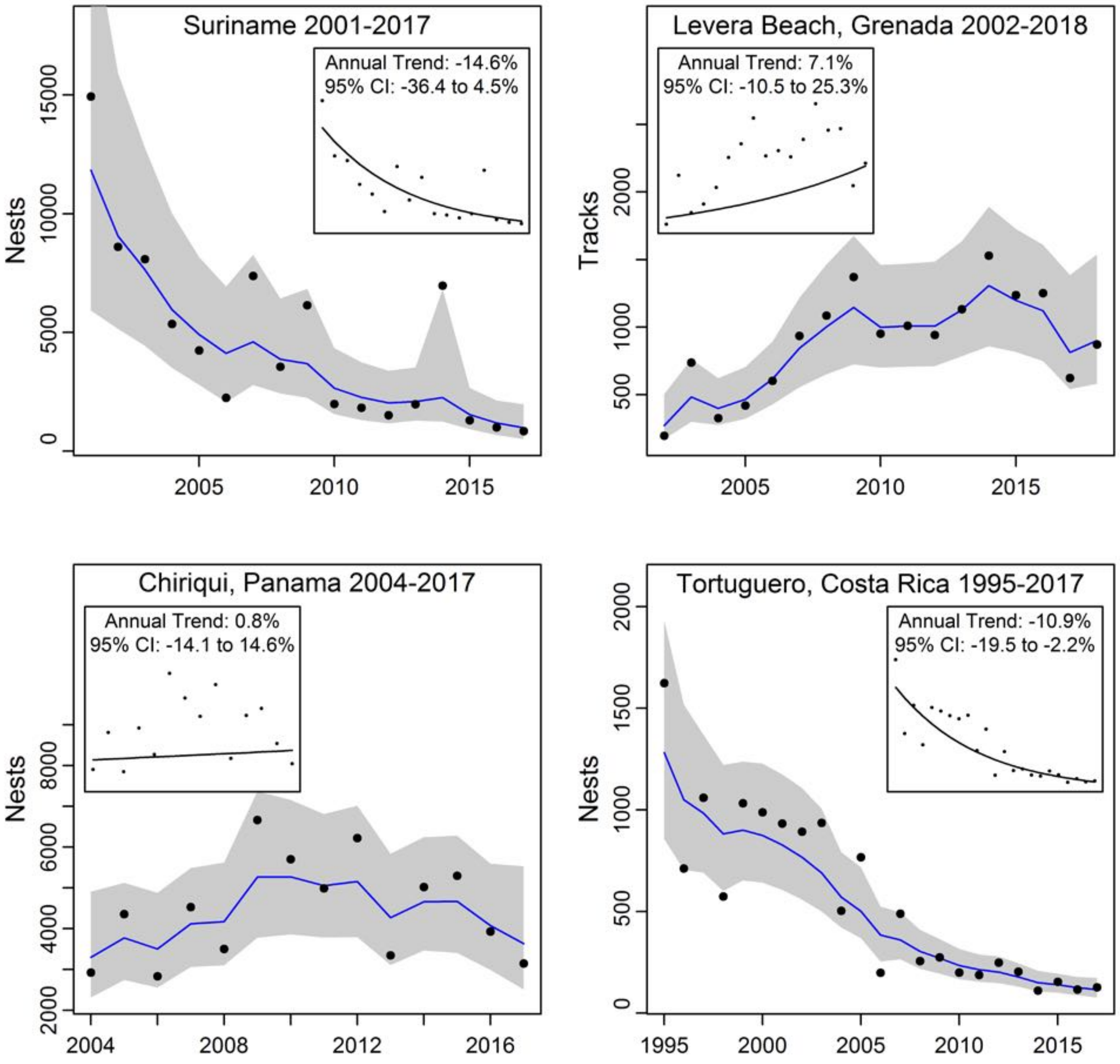
5.1.2 Productivity

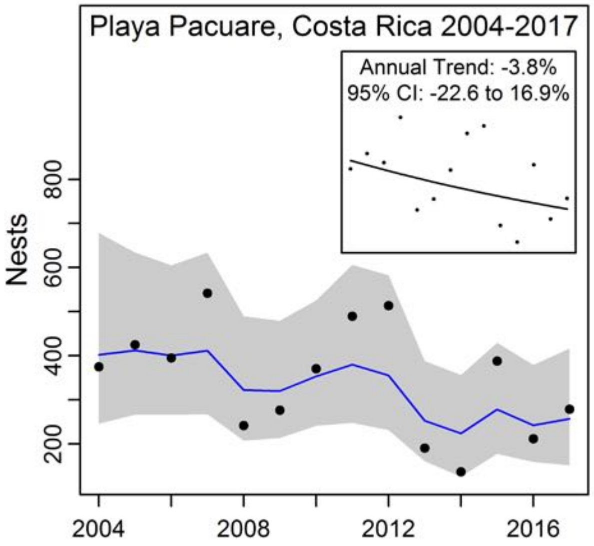
The NW Atlantic DPS exhibits a decreasing nest trend at nesting beaches with the greatest known nesting female abundance (Figure 15). We base our conclusion on nest count data consistently collected using a standardized approach for at least 9 mostly consecutive years (with exceptions explained below), with the last year of data in 2014 or more recently, and with an average of more than 50 nests annually. These data are representative of the DPS because they include the largest nesting aggregations, with the possible exception of the Colombian nesting aggregation. We acknowledge that these trend analyses are indices of nesting (not necessarily nesting females) and that turtles nest in multiple locations, such that nesting at unmonitored beaches was not included. We did not perform a trend analysis for several nesting beaches because we did not have 9 years of data collected in a consistent and standardized manner from these sites. Instead, we include bar graphs for any available data from these sites (Figure 16). Because we analyzed all of the data provided to the Team, the time periods for the different sites vary. As with all DPSs, we report the BSSM trend analysis results as the median and CI, which reflects that there is a 95 percent chance that the trend falls between the low and high CI values. The wider the CI, the less confident we are in the estimated median trend. The higher the “f statistic” the more confident we are in the sign (positive or negative) of the estimated median trend.

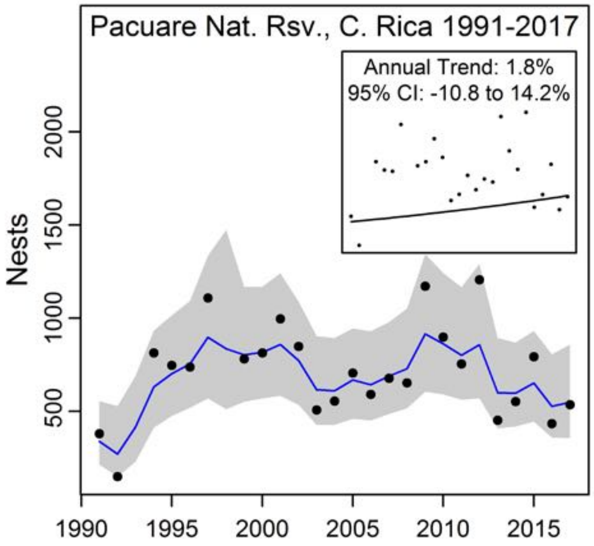
Figure 15. Nest trends at NW Atlantic beaches. The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (nests or tracks). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the inter-annual variability. Abbreviations used for Cayenne (Cay.) and Remire Montjoly (Rem. Mont.), French Guiana (Fr. G.), and Costa Rica (C. Rica).

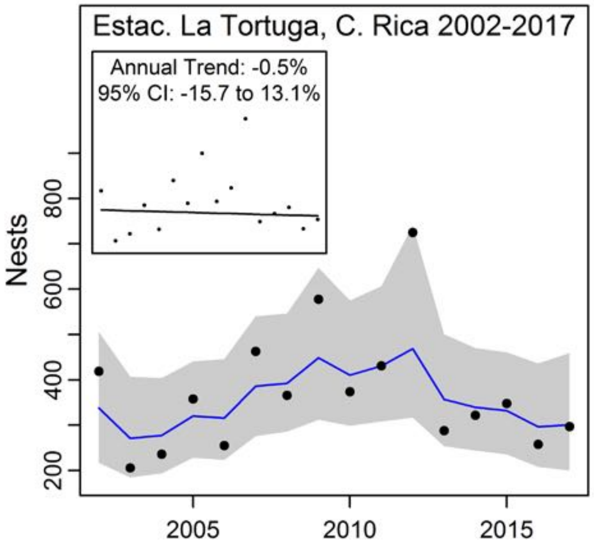


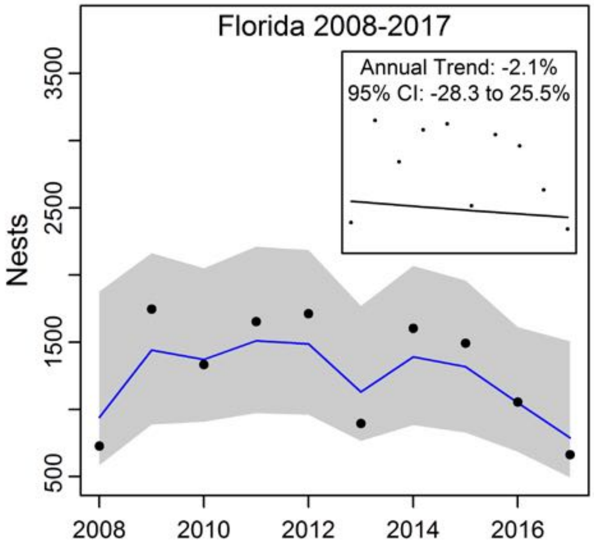

Fishing Pond, Trinidad 2009-2017
Annual Trend: -19.3%
95% CI: -49.8 to 12%
Nests
2000
6000
10000
2010
2012
2014
2016
Tobago 2009-2017
Annual Trend: -0.9%
95% CI: -25 to 21.5%
Nests
200
400
600
800
1000
2010
2012
2014
2016
Yalimapo, French Guiana 2002-2017
Annual Trend: -19.4%
95% CI: -43.2 to 6%
Nests
0
4000
8000
2005
2010
2015
Cay. & Rem.-Mont., Fr.G. 1999-2017
Annual Trend: 2.8%
95% CI: -24.9 to 27.9%
Nests
0
4000
8000
12000
2000
2005
2010
2015

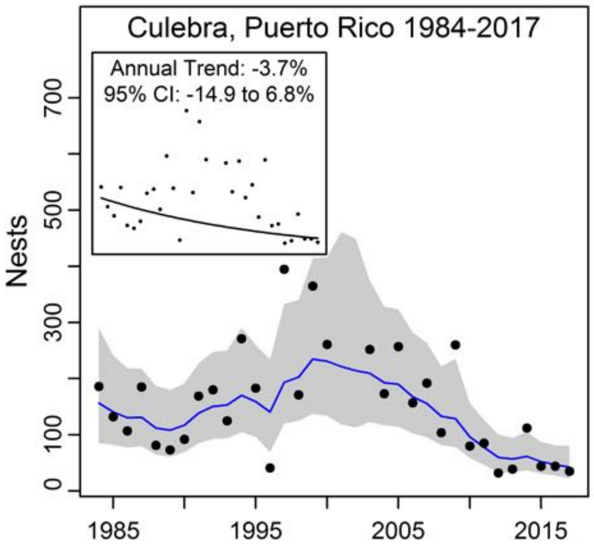

Suriname 2001-2017
Annual Trend: -14.6%
95% CI: -36.4 to 4.5%
Nests
0
5000
10000
15000
2005
2010
2015
Levera Beach, Grenada 2002-2018
Annual Trend: 7.1%
95% CI: -10.5 to 25.3%
Tracks
500
1000
2000
2005
2010
2015
Chiriqui, Panama 2004-2017
Annual Trend: 0.8%
95% CI: -14.1 to 14.6%
Nests
2000
4000
6000
8000
2004
2008
2012
2016
Tortuguero, Costa Rica 1995-2017
Annual Trend: -10.9%
95% CI: -19.5 to -2.2%
Nests
0
500
1000
1500
2000
1995
2000
2005
2010
2015

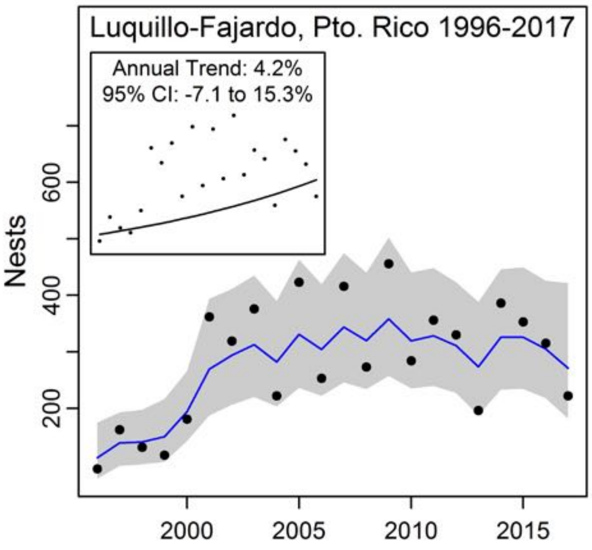

Playa Pacuare, Costa Rica 2004-2017
Annual Trend: -3.8%
95% CI: -22.6 to 16.9%
Nests
200
400
600
800
2004
2008
2012
2016

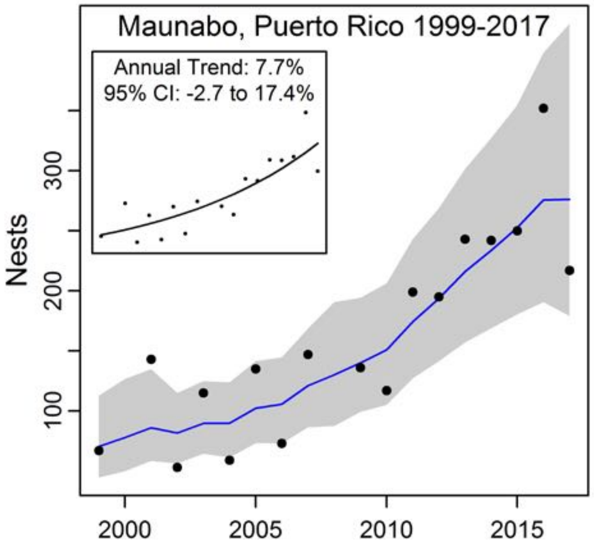

Pacuare Nat. Rsv., C. Rica 1991-2017
Annual Trend: 1.8%
95% CI: -10.8 to 14.2%
Nests
500
1000
1500
2000
1990
1995
2000
2005
2010
2015


Estac. La Tortuga, C. Rica 2002-2017
Annual Trend: -0.5%
95% CI: -15.7 to 13.1%
Nests
200
400
600
800
2005
2010
2015


Florida 2008-2017
Annual Trend: -2.1%
95% CI: -28.3 to 25.5%
Nests
500
1500
2500
3500
2008
2010
2012
2014
2016


Culebra, Puerto Rico 1984-2017
Annual Trend: -3.7%
95% CI: -14.9 to 6.8%
Nests
0
100
300
500
700
1985
1995
2005
2015


Luquillo-Fajardo, Pto. Rico 1996-2017
Annual Trend: 4.2%
95% CI: -7.1 to 15.3%
Nests
200
400
600
2000
2005
2010
2015


Maunabo, Puerto Rico 1999-2017
Annual Trend: 7.7%
95% CI: -2.7 to 17.4%
Nests
100
200
300
2000
2005
2010
2015

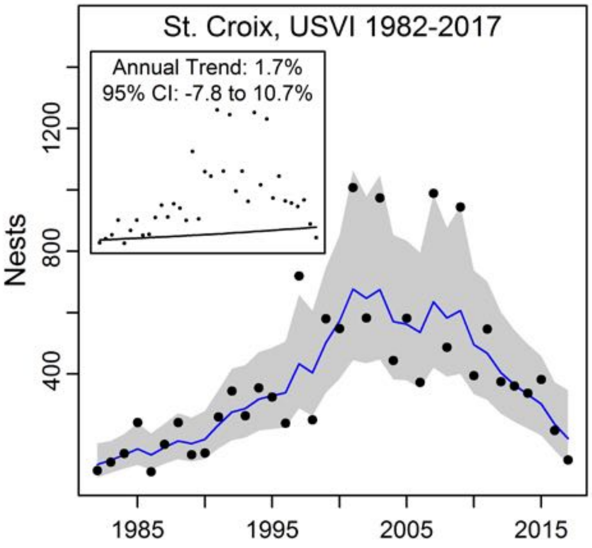
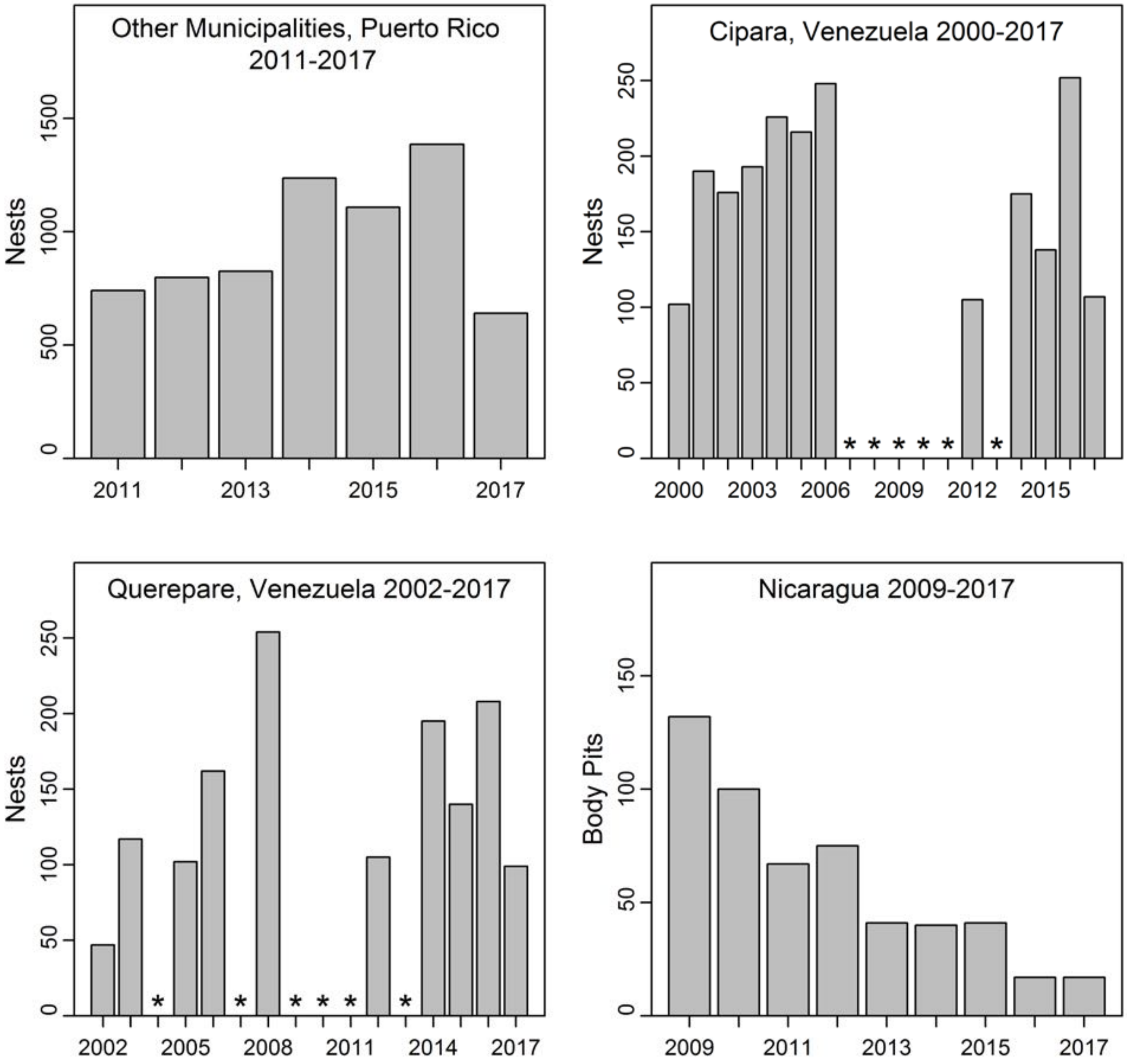

St. Croix, USVI 1982-2017
Annual Trend: 1.7%
95% CI: -7.8 to 10.7%
Nests
400
800
1200
1985
1995
2005
2015

Figure 16. Nesting data at other NW Atlantic beaches. We created bar graphs using data that did not meet our criteria for trend analysis. Asterisks indicate years when data were not available.



This DPS is relatively rich in long-term monitoring data, with seven nations having sufficient data for trend analyses, which provides wide geographic coverage of trends throughout the region. Based on our qualitative assessment of these trends (i.e., considering the relative abundance of each site and its trend), the DPS appears to exhibit an overall decreasing trend in annual nesting activity. Our conclusion is supported by significant declines that 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, and French Guiana (Awala-Yalimapo). In addition, the NW Atlantic Leatherback Working Group completed a region-wide trend analysis that showed an overall decline (NW Atlantic Leatherback Working Group 2018), which we discuss in detail below.

In Trinidad and Tobago, trends in annual nest counts were largely negative over the 2009-2017 period of data collection. We used the best available data provided from TVT, which increased its monitoring starting in 2013, limiting our confidence in these trends. 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 trends in nest counts for Awala-Yalimapo beach in the west (data for 2002-2017) and Cayenne and Remire Montjoly beaches in the east (data for 1999–2017). 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). The monitoring effort at Awala-Yalimapo was not uniformly consistent through years and within one nesting season, but most of the nesting season was covered each year. 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 4,670 nests was found in 2003, with 1,270 mean annual nests from 2002 to 2013), but we were unable to analyze trends or provide a bar plot given the inconsistent monitoring effort in recent years. These areas experienced significant beach erosion, and some previously monitored beaches disappeared. As a result, monitoring on these remote beaches has been reduced since approximately 2010.

Suriname, Grenada, and Panama each had a single time series sufficient for trend analysis. For Suriname, long-term nest count data come from two beaches, Galibi and Braamspunt, which were monitored from 2001-2017. Because combined data were provided for some years, we analyzed these beaches as a single time series. Total nests in Suriname declined over time, with a median trend of −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 from 2002 to 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) over the data collection period (2004–2017).

In Costa Rica, the four beaches for which we had sufficient data to analyze annual nest count trends mostly exhibited declining trends. Of these, 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; f = 0.989; mean annual nests = 547) for data collected from 1995 to 2017. Nest counts declined at Playa Pacuare (median = −3.8 percent; sd = 9.3 percent; 95 percent CI = −22.6 to 16.9 percent; f = 0.703; mean annual nests = 346) over 2004–2017, but increased by 1.8 percent annually (sd = 6.0 percent; 95 percent CI = −10.8 to 14.2 percent; f = 0.644; mean annual nests = 715) at the nearby Pacuare Nature Reserve from 1991-2017. Nest counts at Estacion la Tortuga

(ELT) decreased slightly, with a median trend of −0.5 percent annually (sd = 7.0 percent; 95 percent CI = −15.7 to 13.1 percent; f = 0.537; mean annual nests = 370) from 2002 through 2017.

For the United States, we analyzed annual nest count trends for Florida (statewide data collection from 2008–2017), three beaches in Puerto Rico, including Culebra (1984–2017), Luquillo-Fajardo (1996–2017), and Maunabo (1999–2017), and Sandy Point National Wildlife Refuge in St. Croix, USVI (1982-2017). The median trend for Florida was −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 4.2 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). Data were not collected during two years in Culebra and in Maunabo; however, a total of at least 9 years of data were available. 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). Additional data collected in other municipalities of Puerto Rico from 2011 to 2017 (mean annual nests = 963) were combined into a single bar graph, which shows no obvious pattern over seven years.

Data for Venezuela were available from two beaches, Cipara (2000–2017; mean annual nests = 177) and Querepare (2002–2017; mean annual nests = 143). Trend analyses were not possible due to gaps in the available data; thus, a bar graph is provided for each site.

In Nicaragua, the number of body pits were recorded annually from 2009 through 2017 (mean annual body pits = 59); however, due to inconsistencies in the geographic extent and magnitude of survey effort, the data are only presented in a bar graph. While interpretation of all bar graphs in this report should be done with caution, this one presents evidence for a possible decline.

Data from Honduras were available in a project report (2015); however, the leatherback turtle nesting data focused on nests that were relocated and protected. Due to uncertainty on how well those data represent the entirety of leatherback nesting in Honduras, we did not conduct a trend analysis or produce a bar graph.

Our trend data yield similar results to other recent findings. Given recent concerns with declining nest counts throughout the region, the National Fish and Wildlife Foundation convened a NW Atlantic Leatherback Working Group (the Working Group) to assess recent nesting data and complete a region-wide trend analysis (NW Atlantic Leatherback Working Group 2018). The trend analyses used leatherback nesting data from 23 sites from 14 different nations with at least 10 years of data during which consistent within-site methodology was used. The Working Group used a Bayesian regression model (Sauer *et al.* 2017) to estimate trends for all nesting sites, nesting aggregations, and for the regional population (which is equivalent to the NW Atlantic DPS) during three temporal scenarios: 1) 1990 to 2017, 2) 1998 to 2017, and 3) 2008 to 2017.

Overall, regional abundance-weighted trends were negative across the three temporal scenarios, and became more negative with the more recent time series (NW Atlantic Leatherback Working Group 2018). Overall nest trends were as follows:

- –4.21 (95 percent CI = −6.66 to −2.23) percent annually from 1990 to 2017
- −5.37 (95 percent CI = −8.09 to −2.61) percent annually from 1998 to 2017
- −9.32 (95 percent CI = −12.9 to −5.57) percent annually from 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 “significantly” positive nest trends from 1990 to 2017, but there were no sites with significantly positive trends from 2008 to 2017; significant was defined as when zero was not included in the 95 percent credible intervals around the mean annual trend estimate. 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, −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) had a 7.44 percent annual increase in nesting from 1990 to 2017 and an 8.19 percent annual increase from 1998 to 2017; however, it exhibited a −14.21 percent annual decrease for the most recent time period (2008 to 2017). While nesting increased over time at Cayenne, this increase has apparently not resulted from females shifting from Awala-Yalimapo, as Cayenne turtles 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 more recently, 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. While these results were taken into consideration by the Team when evaluating the extinction risk of the NW Atlantic DPS, we performed our 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 the same result—the NW Atlantic DPS is experiencing a significant decline in nesting (NW Atlantic Leatherback Working Group 2018). Note that we did not receive exactly the same dataset provided to the Working Group and the two trend projects were conducted independently.

In-water abundance studies of leatherback turtles are rare. However, the relative abundance of turtles at a foraging area off Nova Scotia, Canada, from 2002 to 2015 was recently assessed (Archibald and James 2016). This study evaluated opportunistic sightings per unit effort and found a mean density of 9.8 turtles per 100 km^2, 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.* 2011b; Rivas *et al.* 2016). 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 of the study area (Archibald and James 2016).

Despite the declining trend in nesting, productivity parameters for the DPS (Table 4) are similar to the species' averages. While there is some variation, most productivity parameters are relatively consistent throughout the DPS. Mean CCL for NW Atlantic nesting females throughout the region has been reported as 151.8 to 157.6 cm, with minimum and maximum observations spanning a wide range, from 105 (Stewart 2007) to 264.1 cm (converted from 252 cm SCL (Girondot and Fretey 1996) using equation from Tucker and Frazer (1991)). Adults in the NW Atlantic are generally considered to be greater than 145 cm CCL, subadults (not sexually mature) range between 100–145 cm CCL, and juveniles are less than 100 cm CCL (TEWG 2007). 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), 0.89 in St. Croix (Dutton *et al.* 2005), and 0.89 to 0.96 on the Atlantic coast of Florida (Stewart *et al.* 2007b; Stewart *et al.* 2014). Remigration intervals may range anywhere 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 changing (4.5 years; K. R. Stewart, The Ocean Foundation and C. Lombard, USFWS, pers. comm., 2019; K. M. Stewart, Ross University School of Veterinary Medicine and St. Kitts Sea Turtle Monitoring Network, pers. comm., 2019). Considering all available data (Table 4Table 4), we estimated the remigration interval to be 2.7 years, which we rounded to 3 years for use in our calculation of the index of nesting female abundance. Average clutch frequency per nesting season is estimated to range from 3.6 to 8.3 throughout the region, with an overall mean of 5.5 nests each 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 (in addition to a variable number of shelled albumen gobs); however, an early study by Carr and Ogren (1959) reported only 67 eggs per clutch. The incubation period for leatherback nests may last 59 to 67 days, on average. Hatching success is highly variable for nests that remain *in situ*, even for those that are viable and do not experience significant inundation and/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), with an overall estimate of ~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.* 2012b). 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).

Table 4. Productivity parameters for the NW Atlantic DPS.

Productivity metric	**Variable by nation**	**Reference**
Size of nesting female	French Guiana (Awala-Yalimapo): 154.6 cm SCL (range 127–252 cm SCL)	Girondot and Fretey 1996

Productivity metric	**Variable by nation**	**Reference**
	Suriname, annual mean: 154.1 cm CCL (minimum) 155.6 cm CCL (maximum) (range 128–184 cm CCL)	Hilterman and Goverse 2007
	Venezuela (Playa Cipara, Playa Querepare): 151.8 cm CCL	Eckert *et al.* 2012
	Trinidad (Matura Beach): 157.6 cm CCL (range 139.7–210.0 cm CCL)	Chu Cheong 1990
	Trinidad (Matura Beach): 154.5 cm CCL (range 115–196 cm CCL)	Eckert *et al.* 2012
	Costa Rica (Gandoca): 153.2 cm CCL (range 135–198 cm CCL)	Chacón- Chaverri and Eckert 2007
	Costa Rica (Tortuguero): 156.2 cm CCL (range 124.0–180.3 cm CCL)	Leslie *et al.* 1996
	St. Croix: 153.6 cm CCL (range 131.0–177.4 cm CCL)	Boulon *et al.* 1996; Stewart *et al.* 2007a
	Juno Beach, Florida: 151.8 cm CCL (range 125.0–173.5 cm CCL)	Stewart *et al.* 2007a
	Culebra, Puerto Rico: 147.0 cm SCL (range 134.2–160.5 cm SCL)	Tucker and Frazer 1991
Annual female survivorship (percent)	French Guiana: 0.91 (95 percent CI 0.75 0.97)	Rivalan *et al.* 2005
	St. Croix: 0.893 (95 percent CI 0.87–0.92)	Dutton *et al.* 2005
	Juno Beach, Florida: 0.956	Stewart *et al.* 2007b
	Juno Beach, Florida: 0.889	Stewart *et al.* 2014

Productivity metric	**Variable by nation**	**Reference**
	NW Atlantic synthesis: 0.85	Pfaller *et al.* 2018
Remigration interval (years) (*used in index of nesting female abundance)	DPS average: 2.7*	
	French Guiana: 2*	Eckert *et al.* 2012
	French Guiana: 2 (range 1–5)	Chevalier and Girondot 1998
	French Guiana: 3*	Rivalan *et al.* 2005
	Suriname: 2	Schulz 1975
	Suriname: 2* (range 1–5)	Hilterman and Goverse 2007
	Venezuela (Playa Cipara, Playa Querepare): 2.5*	Eckert *et al.* 2012
	Costa Rica (Pacuare): 2.5* (range 1–6)	Rivas *et al.* 2016
	St. Croix: 4.5 years*	K. Stewart (NMFS) and C. Lombard (USFWS), pers. comm., 2019
	St. Croix: 2–3 (2.5*); (range 1–11)	Dutton *et al.* 2005, Garner *et al.* 2017
	St. Croix: 2	Boulon *et al.* 1996
	Juno Beach, Florida: 2.7*	Stewart *et al.* 2014
Clutch size (eggs)	French Guiana (Silébâche Beach): 87.7	Pritchard 1971 (in Eckert *et al.* 2012)
	French Guiana (Awala-Yalimapo): 87.8	Caut *et al.* 2006
	Suriname: 86	Pritchard 1969 (in Eckert *et al.* 2012)

Productivity metric	**Variable by nation**	**Reference**
	Suriname (Bigi Santi): 84.1	Schulz 1975
	Suriname (Babusanti): 86.6	Hilterman and Goverse 2007
	Venezuela (Playa Parguito): 80.6	Hernández *et al.* 2007
	Trinidad (Matura Beach): 85.7	Maharaj 2004
	Trinidad (Grande Riviera Beach): 83.1	Maharaj 2004
	Costa Rica (Matina and Tortuguero): 67.3	Carr and Ogren 1959
	Costa Rica (Gandoca): 81.2	Chacón and Eckert 2007
	Costa Rica (Tortuguero, 1990/1991): 83	Leslie *et al.* 1996
	Costa Rica (Tortuguero, 1995): 80.2	Campbell *et al.* 1996
	Costa Rica (Pacuare): 81.3	Rivas *et al.* 2016
	Dominican Republic: 71.1 (2008) 83.6 (2009)	Revuelta *et al.* 2014
	St. Croix: 82	Eckert *et al.* 1989b
	Humacao, Puerto Rico: 78.4	Matos 1986 (in Eckert *et al.* 2012)
	Culebra Island, Puerto Rico: 77.1	Tucker 1988 (in Eckert *et al.* 2012)
	Brevard County, Florida: 87.3	Maharaj 2004

Productivity metric	**Variable by nation**	**Reference**
	Florida: 73	Stewart and Johnson 2006
	All Western Atlantic: 82	Sotherland *et al.* 2015
Clutch frequency (number of nests per nesting season)	French Guiana: 8	Rivalan *et al.* 2006; Eckert *et al.* 2012; Eckert *et al.* 2015
	French Guiana: 3.55[1]	KWATA data in Berzins 2018 Réserve Naturelle de l'Amana data in Berzins 2018
	Suriname: 4.5	Eckert *et al.* 2015
	Venezuela: 4.4	Eckert *et al.* 2015
	St. Croix: 5.1[1]	Eckert *et al.* 2015
	St. Croix: 5.4 (highest 6.17 in 2003; lowest 3.6 in 2010)	Garner *et al.* 2017
	Culebra, Puerto Rico: 6.6	Eckert *et al.* 2015
	Florida: 4.2[1]	Eckert *et al.* 2015
	Florida: 4.4	Stewart *et al.* 2014
Internesting interval (days)	St. Croix: 9–10	Eckert *et al.* 2015
	Culebra, Puerto Rico: 9	Eckert *et al.* 2015
	French Guiana: 9–10	Eckert *et al.* 2015
	Florida: 10	Eckert *et al.* 2015

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[1] We used this value as the most recent data in our calculation of the index of nesting females.

Productivity metric	**Variable by nation**	**Reference**
	Suriname (Babusanti): 61–65	Hilterman and Goverse 2007
	Suriname (Matapica): 63–67	Hilterman and Goverse 2007
Mean incubation period (*in situ*) (days)	Venezuela: 59	Hernandez *et al.* 2007
	Trinidad (Matura): 66	Maharaj 2004
	Costa Rica (Gandoca): 60	Chacón and Eckert 2007
	Dominican Republic: 65 (2008); 63 (2009)	Revuelta *et al.* 2014
	St. Croix: 64 days	Eckert and Eckert 1990a
	Florida: 67 days	Stewart and Johnson 2006
Hatching success[2] (*in situ*) (percent)	French Guiana: 33–39	Girondot *et al.* 2007
	Suriname (Babusanti): 10.6–56.0	Hilterman and Goverse 2007
	Suriname (Matapica): 58.3–63.7	Hilterman and Goverse 2007
	Trinidad: 65.3 (does not include eroded nests)	Maharaj 2004
	Venezuela: 47.2	Hernandez *et al.* 2007
	Colombia: 43.8–69.4	Patiño-Martínez *et al.* 2008a
	Panama: 75.7	Patiño-Martínez *et al.* 2008a

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[2] May include failed nests.

Productivity metric	**Variable by nation**	**Reference**
	Panama: 46.0	Godfrey 2018
	Costa Rica (Tortoguero 1998–2005): 13.8–46.5	Troëng *et al.* 2007
	Costa Rica (Tortuguero 2015): 8.9	STC unpublished data
	Costa Rica (Pacuare 1992–2012): 55.2 (42–62.1)	Rivas *et al.* 2016
	Costa Rica (Pacuare 2013–2017): 15.5–59.1	Pacuare Reserve Ecology Project International unpublished data
	St. Croix: 58.5 percent (40.3–67.8)	Garner *et al.* 2017
	Culebra, Puerto Rico: 75.1 (72.2–76.8)	Tucker 1988
	Florida 2002–2016: 42.6 (30.7–62.6)	Brost *et al.* 2015, FWC/FWRI unpublished data
	Suriname: 30–100	Dutton *et al.* 1985
Sex ratio (hatchling, percent female)	Suriname: 69.4	Godfrey *et al.* 1996
	Tobago: 100	Mickelson and Downie 2010
	Colombia: 83–97	Patiño-Martínez *et al.* 2012b
	Costa Rica (Tortuguero): 63–70.8	Leslie *et al.* 1996
	Dominican Republic: 53.6 (2008); 72.9 (2009)	Revuelta *et al.* 2014

Productivity metric	**Variable by nation**	**Reference**
	Canada (foraging): 69	James *et al.* 2007
	South Carolina (strandings): 69.6	Murphy *et al.* 2006
Sex ratio (large juvenile and adult, percent female)	US Atlantic and GOM (sub-adult strandings): 61	TEWG 2007
	US Atlantic and GOM (adult strandings): 57	TEWG 2007

Whereas the productivity metrics of the DPS are similar to the species' averages, the overall trend in nests laid annually is decreasing. In addition, there are some 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. The declining trends reflect reduced nesting female abundance due to known threats; in addition, longer remigration intervals and/or reduced clutch frequencies may play a role in this decline. We conclude that the declining nest trend places the DPS at elevated extinction risk.

5.1.3 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 (Figure 9), 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, Colombia, Panama, Costa Rica, Puerto Rico, St. Croix, and the mainland United States (Florida Atlantic) (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 763 bp mtDNA sequences in combination with an array of 17 nuclear microsatellite DNA loci (Roden and Dutton 2011, Dutton *et al.* 2013b). The microsatellite data revealed fine-scale genetic differentiation among neighboring subpopulations (Dutton *et al.* 2013b): 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 nDNA microsatellite markers in their study. The mtDNA homogeneity between Costa Rica and Florida, taken with the nDNA differentiation, 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.* 2013b). 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 between small French Caribbean Island nesting aggregations (Martinique and Guadeloupe) and the mainland French Guiana rookery (Molfetti *et al.* 2013). St. Croix likely represents a broader Northern Caribbean subpopulation that includes multiple neighboring island nesting aggregations in the USVI and Puerto Rico, however sampling and analysis are required to determine extent of fine scale structuring (NMFS unpublished data). The Costa Rica (Tortuguero and Gandoca) and Guianas (French Guiana and Suriname) nesting aggregations are distinct subpopulations based on microsatellite and mtDNA results (Dutton *et al.* 2013b), 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, on the level of philopatry to a region, not a specific beach. In adjacent nesting sites in French Guiana and Suriname, five to six percent of females were observed to shift nesting 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 the 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 mixing of nesting aggregations at foraging areas. Mixed stock analysis and microsatellite assignment of longline bycatch of 684 samples from across the NW Atlantic, 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

Exhibit 8 to Declaration of Adam Keats

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 (Stewart *et al.* 2016). 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 generally represent the relative population sizes for these breeding populations. Microsatellite DNA assignment of wild captured or stranded male leatherback turtles (n = 122) throughout the NW Atlantic and Mediterranean found that all males originated from NW Atlantic nesting aggregations (55 percent from Trinidad, 31 percent from French Guiana, and 14 percent from Costa Rica; 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 continental 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; Large Pelagics Research Center 2014; Dodge *et al.* 2014, 2015; Aleksa *et al.* 2018), Canada (James *et al.* 2005a; James *et al.* 2005b; James *et al.* 2005c; James *et al.* 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; Ferraroli *et al.* 2004a; Hays *et al.* 2004; Eckert 2006; Eckert *et al.* 2006; Hays *et al.* 2006; TEWG 2007; Evans *et al.* 2008; Sonmez *et al.* 2008; Fossette *et al.* 2010a; Fossette *et al.* 2010b; Bailey *et al.* 2012a,b; Richardson *et al.* 2012; Fossette *et al.* 2014; Stewart *et al.* 2014; Horrocks *et al.* 2016; Chambault *et al.* 2017). For example, turtles from Nova Scotian foraging grounds were tracked to nesting areas off Colombia, Trinidad, Guyana, and French Guiana (James *et al.* 2007; Stewart *et al.* 2013; 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.* 2005a), 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, Florida (Eckert *et al.* 2006), Panama (Evans *et al.* 2008), and Anguilla (Richardson *et al.* 2012). The eastern and western GOM also provides 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.* (2004a) tracked adult 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. Including turtles captured at sea provides a bit more insight into leatherback migrations; Fossette *et al.* (2010a, 2010b) discovered 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) round-trip migrations from where they started through the North Atlantic Ocean heading northwest to fertile foraging areas off Canada, in the Gulf of Maine 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 (TEWG 2007).

These studies show that NW Atlantic nesting females migrate and forage throughout most of the North Atlantic from the Equator to more northern latitudes (as far north as 71° N latitude). There is considerable mixing of leatherback turtles among NW Atlantic nesting beaches and multiple foraging areas throughout the North Atlantic Ocean (Figure 5; Fossette *et al.* 2014). The range of juvenile leatherback turtles may be more restricted than subadult and adult leatherback turtles. 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 leatherback turtles and found that juveniles less than 100 cm CCL 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. Thus, we conclude that the spatial distribution and structure reduces the extinction risk of the NW Atlantic DPS.

5.1.4 Diversity

The NW Atlantic DPS exhibits spatial diversity, as demonstrated by insular and continental nesting, diverse foraging habitats, multiple 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), which 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 (Figure 5. Satellite tracking of 106 leatherback turtles in the

Atlantic Ocean; Fossette *et al.* 2014). Foraging habitats include temperate waters of the GOM, North Central Atlantic, 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.* 2010b). 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.* 2013b). 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 the DPS some level of resilience against short-term spatial and temporal changes in the environment; however, high-abundance nesting occurs only at 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. Its moderate genetic diversity may provide the DPS with the raw material necessary for adapting to long-term environmental changes. We conclude that such diversity reduces the extinction risk of this DPS by providing it with some resilience to threats.

5.2 ESA Section 4(a)(1) Factors

The following sections describe and assess the 4(a)(1) factors or threats. For each, we evaluate the best available information on the threat. We also describe the exposure and impact of each threat.

To assess these threats, we reviewed the best available information from throughout the range of the DPS. We focus on current threats for which data are available. We also considered previous reviews when primary information was not available for that area. For example, Dow *et al.* (2007) compiled data from a questionnaire sent to 43 Wider Caribbean Region nations and territories, which asked for information on nesting, protection policies, and nesting and foraging threats within their jurisdiction. Information was collected for all species of sea turtles combined. See Tables 4, 5 and 6 in Dow *et al.* (2007) for a complete accounting of national policy, threats to turtles on the beach, and threats to turtles at sea, respectively.

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

Destruction and modification of leatherback turtle nesting habitat results from: 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; FWS 1999; Dow *et al.* 2007; Eckert *et al.* 2012; NMFS and USFWS 2013). As a result, most nesting beaches are severely degraded from a variety of activities. This is a historical and continuing problem throughout the range of the DPS. In the following paragraphs, we describe the individual habitat threats in greater detail.

In many areas, nesting habitat is under constant threat from coastal development and construction (Lutcavage *et al.* 1997; 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; Dow *et al.* 2007; Hernandez *et al.* 2007; Forestry Division (Government of the Republic of Trinidad and Tobago) *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 occur (Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). Although some important nesting beaches in Trinidad remain largely free of commercial development, the same cannot be said of Grande Riviere, the largest nesting aggregation of this DPS. Likewise, several Tobago beaches are densely developed for commercial tourism, resulting in reduced sea turtle access to potential nesting sites due to buildings, umbrellas, and other recreational equipment (Forestry Division (Government of the Republic of Trinidad and Tobago) *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). Ecosystems continue to be threatened by coastal development in Puerto Rico, even though the coastal zone is protected by the Maritime-Terrestrial Zone designation (Coastal Public Trust Lands; Flores and Diez 2016).

Leatherback nest placement and subsequent nest success are likely influenced by coastal development. 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 (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.

While erosion is often intensified due to anthropogenic influences, such as coastal armoring, 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; since 2010, previously monitored beaches have disappeared or been reduced, reducing or preventing monitoring (and likely nesting). 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 Braamspunt 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; Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). At some Trinidad and Tobago nesting sites (e.g., Fishing Pond, Matura, Grande Riviere, 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; Forestry Division (Government of the Republic of Trinidad and Tobago) *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.* 2008a).

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 (range = 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 than in non-overwashed (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.* 2008a). While levels of inundation and resulting declines in hatching success have been noted at multiple sites throughout 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; FDEP unpublished data).

Naturally dynamic areas make it difficult to protect nesting beach habitat and accurately assess leatherback nest 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; Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). However, leatherback turtles seem to have adapted to this constant geomorphological change of beaches in the Guianas. 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).

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). These assessments of armoring do 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). That said, 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 of the DPS.

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 of nests 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, nesting success, nest construction, 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 natural sand transport has been altered and little or no suitable nesting habitat previously existed, beach nourishment has been found to result in increased nesting at the nourished site compared to the beach prior to nourishment (Ernest and Martin 1999). However, on most beaches in the southeastern United States, nesting success typically declines for the first year or two following construction, even though more beach 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; Dow *et al.* (2007) found that only four nations (Anguilla, Cuba, Mexico, and United States) reported frequent or occasional beach nourishment.

Sea level rise as a result of climate change is likely to intensify erosion and loss of nesting beach habitat. Further, with the threat of sea level rise, additional armoring of the coast is anticipated, creating a further reduction of nesting habitat (Hawkes *et al.* 2009). The impacts of erosion and subsequent habitat loss are anticipated to continue under the future climate change projections. Additional climate change impacts are discussed in Section 5.2.5.4 Climate Change).

Coastal development also contributes to habitat degradation by increasing light pollution. As noted in Section 4.2.1, light pollution 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 2018 unpublished data). 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 occur in sufficiently remote areas, and large-scale development is currently less of an issue there (Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010; NMFS and USFWS 2013). That said, even in the same country, light pollution is variable. Fossette *et al.* (2008a) reported that light pollution from residential areas is a problem at Cayenne Beach in French Guiana, but it is not an issue at Awala-Yalimapo. Similarly, lighting is not a significant problem in Trinidad, but is a concern on nesting beaches in Tobago (Forestry Division (Government of the Republic of Trinidad and Tobago) *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 is a threat that has been managed to some extent in some areas (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 lighting ordinances is unregulated, resulting in urban glow. Although outreach and conservation programs to reduce the impacts of lighting exist 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.

Extracting sand from nesting beaches for construction projects has an obvious 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 (Forestry Division (Government of the Republic of Trinidad and Tobago) *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.

In some nations, upland deforestation and the resultant deposition of debris and garbage on the nesting beaches can destroy or modify nesting beaches. The debris can block access of gravid 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).

Beach driving also occurs in most nations throughout the range of this DPS (Chacón-Chaverri and Eckert 2007; Dow *et al.* 2007; Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). In the U.S., vehicular driving is allowed on certain beaches in Florida where leatherback turtles nest at varying levels (e.g., Duval, St. Johns, and Volusia counties). Beach driving has been found to reduce 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 has been found to hinder 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 on nesting beaches contributes to erosion, especially during high tides or on narrow beaches where driving is concentrated on the high beach and foredune.

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). Dense stands have overtaken many coastal areas throughout central and south Florida. Australian pines cause excessive shading of the beach that would not otherwise occur. Studies on 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).

While non-native vegetation can affect nesting habitat throughout the range of the DPS, native vegetation can also affect productivity. For instance, on Sandy Point, St. Croix, changing erosion-accretion cycles led to native *Ipomoea pescaprae* 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 pescaprae* decreased nest productivity by reducing leatherback hatching and emergence success rates (Conrad *et al.* 2011).

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 Prohibited Area status was granted to the nesting beach at Grande Riviere in 1997. In 1998, the Amana Nature Reserve was established in French Guiana, which includes Awala-Yalimapo beach and a 30 m wide marine fringe. 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 critical habitat for leatherback turtles of this DPS, which protects the nesting beaches at Sandy Point, St. Croix (43 FR 43688; September 26, 1978) and surrounding marine waters (43 FR 17710; March 23, 1979); however, assuming no ESA protection, these critical habitat protections would be lost.

In the marine environment, habitat threats include anthropogenic noise and offshore lighting. We discuss other threats to marine habitat and prey (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.

We conclude that large numbers of nesting females, hatchlings, and eggs are exposed to the reduction and modification of nesting habitat, as a result of activities such as coastal development and armoring, erosion, and artificial lighting. This threat impacts 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.

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

Overutilization for commercial, recreational, scientific or educational purposes is a threat to this 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 in the range of the NW Atlantic DPS protect leatherback turtles from direct harvest, in some Caribbean and Central American nations, the harvest of leatherback turtles is legal (Brautigam and Eckert 2006; Dow *et al.* 2007; Richardson *et al.* 2013; Humber *et al.* 2014; Horrocks *et al.* 2016). 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 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. Humber *et al.* (2014) reviewed literature and contacted country experts to assess the global level of legal sea turtle take. In the NW Atlantic DPS, the estimated annual legal take as of January 2013 was approximately 89 leatherbacks, with legal takes in Colombia (48.1), Panama (34), St. Vincent and the Grenadines (3), Guyana (2.8) and Antigua and Barbuda (1) (Humber *et al.* 2014). However, Bräutigam and Eckert (2006) and Dow *et al.* (2007) both reported that Panama and Guyana have full sea turtle protection, and since the Humber *et al.* (2014) assessment, new legislation has been implemented to protect sea turtles in some of the other countries (e.g., Antigua and Barbuda, St. Vincent and the Grenadines). Regardless, these allowable lethal captures are significant because they often target nesting females, removing the most important individuals from the population. We are unable to quantify the current amount of legal harvest at this time, so the exact magnitude of this threat is less clear. See Section 5.2.4 Inadequacy of Regulatory Mechanisms for more information on allowable lethal take. Leatherback eggs, rather than turtle meat, are more commonly harvested (TEWG 2007; Patiño-Martínez *et al.* 2008a).

Poaching of turtles and eggs occurs throughout the NW Atlantic DPS and Dow *et al.* (2007) identified this as a threat for all turtle species on the beaches in the Wider Caribbean Region. In Panama, interviews with local people 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; Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). Most poaching, however, targets eggs, and the level of poaching often is determined by how much monitoring and activity (to deter poachers) occurs 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 were illegally collected in Tortuguero from 2000 to 2005. 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 in 2006 to 2010, with an average 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 (22 to 31 percent of clutches at Playona in 2006 and 2007; Patiño-Martínez *et al.* 2008a) and, to some extent, in most other Caribbean nations (Guyana and Grenada). Poaching is likely more prevalent, and occurs at higher levels, on unmonitored or unprotected beaches (Dow *et al.* 2007; Troëng *et al.* 2007; TEWG 2007; Forestry Division (Government of the Republic of Trinidad and Tobago) *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 issue in Trinidad, but levels have been reduced with more people monitoring the beach (Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). 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). Poaching in Panama outside the monitored areas still occurs, with the sale of eggs widespread but clandestine (Brautigam and Eckert 2006). The Marine Turtle Conservation Act funds activities in Panama to reduce poaching. Similarly, 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. Better estimates of the amount of adult and egg poaching are needed throughout the DPS, as 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. Fewer adults and eggs are exposed to legal harvest. 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. Egg harvest reduces productivity. Thus, we conclude that overutilization poses a threat to the DPS given the severity of the impact and exposure.

5.2.3 Disease and Predation

Disease and predation are threats to the NW Atlantic DPS. The information on leatherback turtle diseases is limited, but predation has been more extensively documented.

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 nesters on beaches in Costa Rica (Santoro *et al.* 2008) and St. Kitts (Dutton *et al.* 2013a); 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 7 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. Entangled turtles exhibited blood chemistry results indicating stress and possibly reduced food and seawater intake associated with entanglement. In addition, Perrault *et al.* (2012) examined baseline blood chemistry metrics (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.

From these data, we estimate that the exposure of eggs, juveniles and adults to disease is low. We are unable to quantify the impact of disease as we are unaware of any deaths or reductions in productivity directly related to disease. However, the existence of disease may be more noteworthy given other threats to the species and related physiologic derangements. We conclude that disease is likely a threat to the DPS. There is uncertainty regarding the consequences of disease on the DPS because of the lack of information available on long term population level impacts.

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, fly larvae, locust larvae, mole crickets, ghost crabs, vultures, dogs, cattle (trampling), armadillo, opossum, coati, and raccoons (see Eckert *et al.* 2012). In particular, dog predation of eggs is a particular concern 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 (Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). On Chiriqui Beach, Panama, 54 percent of the monitored leatherback nests were depredated by dogs in 2003, and approximately 8 percent in 2004 (Ordoñez *et al.* 2007). Such predation may have been reduced as a result of protection efforts funded by the Marine Turtle Conservation Act of 2004. 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; assuming no ESA protection, it is likely that predation rates in Puerto Rico would be higher.

Egg predation by other species, such as dipteran larvae, 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 increasing poaching as well as logistical difficulties in these areas of high-density nesting. Nest loss to predators was found to be the 7th ranked threat to turtles (all species, not specific to leatherback turtles) on nesting beaches in the Wider Caribbean Region, and 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, carnivorous fishes, gray snapper, tarpon, vultures, hawks, gulls, night herons, frigate birds (Fregatidae), dogs, mongoose, coati, and raccoons (see 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 of leatherback subadults and adults, presumably because of their large size and pelagic behavior. Predation by sharks and killer whales have 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 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 predator origin.

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 of predation is reduced productivity (reduced emergence success). Predation on nesting females reduces abundance and productivity. We conclude that predation is a threat to the NW Atlantic DPS.

5.2.4 Inadequacy of Regulatory Mechanisms

Many regulatory mechanisms (including State, Federal and international) have been promulgated to protect leatherback turtles, their 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 (e.g., poaching, bycatch, etc.). The effectiveness of many international regulations was evaluated by Hykle (2002), and those findings apply here when appropriate. Often international instruments 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 regulations are summarized in Section 13. Appendix II: NW Atlantic DPS Regulatory Mechanisms. Although these regulatory mechanisms provide some protection to the species, most are inadequate to sufficiently reduce the threats they were designed to address, generally as a result of poor implementation or incomplete enforcement. Specifically, the inadequacy of existing regulatory mechanisms for impacts to nesting beach habitat and overutilization (harvest of turtles and eggs) is a continued threat to this DPS (see Sections 5.2.1 and 5.2.2 above). In addition, in Section 5.2.5, we describe the insufficiency of regulatory mechanisms with respect to several threats including bycatch in fishing gear, vessel strikes, and marine debris. Despite existing regulatory mechanisms, bycatch, incomplete nesting habitat protection, and poaching remain significant threats to the DPS.

5.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

Other threats affect the NW Atlantic DPS, with fisheries bycatch being the primary threat. Additional natural or manmade factors that impact this DPS include vessel strikes, pollution, climate change, oil and gas activities, natural disasters, and oceanographic regime shifts.

5.2.5.1 Fisheries Bycatch

Bycatch in fisheries 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 through 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 turtle interactions, of which 2,300 were lethal, occurred in U.S. Atlantic Ocean and GOM commercial fisheries annually. After implementing bycatch reduction measures, 1,400 leatherback turtles, 40 of those dead, were estimated to be bycaught annually in the Atlantic Ocean/GOM. The Atlantic/GOM pelagic longline fishery was responsible for the most 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 not included in these estimates, interactions are difficult to observe if gear modifications are in place, and the methods used are conservative (Finkbeiner *et al.* 2001).

In the Wider Caribbean Region, reports of leatherback turtle bycatch in fisheries are common. In a survey of Caribbean nations by Dow *et al.* (2007), fisheries bycatch ranked among the highest threats in the water. 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 is leatherback turtle bycatch in artisanal nearshore and offshore gillnet, longline and trawl fisheries (Barrios-Garrido and Montiel-Villalobos 2016).

While some information on fishery bycatch is collected from fisher surveys (Moncada *et al.* 2003; Delamare 2005; Madarie 2006; Madarie 2010; Madarie 2012), often much of what is known about potential anthropogenic impacts, including fishery bycatch, comes from stranding records and anomalies (e.g., injuries, scars, and gear) observed on 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 injuries that showed evidence of being fisheries related, whereas in 2003, at least 18.3 percent had such injuries, and 9 percent in 2005 (Hilterman and Goverse 2007). Further, from 2000 to 2003, an average of 28 leatherback turtles stranded on the Suriname survey beaches. While no cause of death was immediately apparent, Hilterman and Goverse (2007) indicated that the mortalities were fisheries related. On the western oceanic nesting beaches of French Guiana, injuries related to interactions with fisheries (scars, hooks, net present) were recorded on 8 to 10 percent of nesting females (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 field research sites in Atlantic Canada foraging grounds and

Trinidad nesting beaches found 19 percent of turtles exhibited injuries indicative of entanglement in lines or nets, and 17 percent of turtles showed evidence of hooks; 62 percent of turtles exhibited a minimum of one external injury (Archibald and James 2018).

In the Mediterranean and eastern North Atlantic Ocean, leatherback interactions with fisheries also occur. 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 necessarily specific to 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. Below we review information on specific gear interactions.

5.2.5.1.1 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.* 2013a; 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 rates in gillnets throughout the NW Atlantic Ocean at 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 gillnet 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 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, 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 (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, WIDECAST, pers. comm., 2018). The effort and turtle bycatch in this fishery are lower, but mortality rates are higher (~70 percent; Gass 2006). As such, the bottom set gillnet fishery is thought to have a comparable level of mortality as estimated in 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 the nation's most significant cause of sea turtle mortality (Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010). Bond and James (2017) tracked a female leatherback 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 Trinidad are estimated to kill well over 1,000 leatherback turtles annually, off the beaches hosting the largest nesting aggregation in the DPS, comprising a significant threat to the DPS.

High levels of gillnet bycatch occur in other Caribbean and South American nations, again off major nesting beaches. In French Guiana, bycatch was confirmed to be high in the Maroni estuary (Girondot 2015). In 2003, 26 leatherback turtles were caught in coastal gillnets and released off the Cayenne/Montjoly nesting sites (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 (70 and 30 percent, respectively). In Suriname, a World Wildlife Fund survey of fishermen estimated leatherback bycatch in drifting gillnets at 584 individuals 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 in gillnet fisheries (Patiño-Martínez *et al.* 2008). In Brazil, 200 leatherback turtles are captured annually in gillnets; however, mortality rates are not available (Marcovaldi *et al.* 2005 in Lewison and Crowder 2007). Longline and driftnet fisheries in Moroccan waters off northwestern Africa capture approximately 100 leatherback turtles annually (Benhardouze *et al.* 2012).

Although not as elevated as in the Caribbean (based upon observed interactions), gillnet bycatch occurs in U.S. and Canadian waters. South Carolina, Georgia, Florida, Louisiana, and Texas have prohibited gillnets in their State waters, but active gillnet fisheries remain in other States, and U.S. Federal waters. No estimates of leatherback bycatch in gillnet fisheries are available in U.S. waters due to the limited observed interactions. However, from 2003 to 2017, fishery observers recorded lethal and non-lethal leatherback bycatch in fixed sink, drift sink, and drift floating gillnets throughout the U.S. Atlantic EEZ and GOM (NMFS unpublished data). From 1989 to 1998, U.S. drift pelagic gillnets captured 54 leatherback turtles; that gear is no longer used. Hamelin *et al.* (2017) reviewed leatherback entanglement records reported 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 nets (gillnets or trammel nets) and drift gillnets. The UN established a worldwide moratorium on driftnet 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 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, 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), 8 were caught in the NE Atlantic tuna driftnet fishery (with 25 percent mortality) and 1 was caught in a hake gillnet (Pierpoint 2000).

Historically, driftnet fishing in the Mediterranean Sea caught large numbers of sea turtles. There are an estimated 600 illegal driftnet vessels operating in the Mediterranean, including fleets based in Algeria, France, Italy, Morocco, and Turkey (Environmental Justice Foundation 2007). Of 411 records of leatherback turtles 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 documented leatherback bycatch, and occasional leatherback bycatch occurs in Croatian artisanal gillnet fisheries (Camiñas 2004; Ergene and Uçar 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 (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.* 1997). Further, a review by Casale (2008) found that leatherback turtles are bycaught in the drift gillnet fishery in Spain at a rate of 0.065 turtles/day-boat. Considering data throughout the entire Mediterranean, as well as a conservative approach, Casale (2008) considered loggerhead mortality in set nets to be 60 percent; the mortality rate for leatherback turtles in set gillnets may be similar.

Throughout the range of the NW Atlantic DPS, leatherback bycatch reduction measures in gillnets have not been required, but measures to reduce leatherback bycatch have been presented 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 will be 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 U.S., gillnets with stretched mesh 7 inches and larger are prohibited at certain times off North Carolina and Virginia to protect sea turtles (71 FR 24776, April 26, 2006). Regardless of some of these protective measures, gillnet bycatch (especially off nesting beaches) is a major threat to this DPS, resulting in the loss of thousands of mature individuals annually.

5.2.5.1.2 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.* 2010b; Wallace *et al.* 2013a). There is significant concern over the effects of pelagic longline fishing, which extends globally throughout temperate and tropical waters (Fossette *et al.* 2014; Gray and Diaz 2017). High seas longlines in international waters, set by varying international vessels, frequently catch leatherback turtles (Lewison *et al.* 2004). Individuals are primarily 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/set, classifying the gear as a relatively low bycatch impact relative to other sea turtle populations (Wallace *et al.* 2013a; Lewison *et al.* 2015). However, because longline fisheries are widespread across leatherbacks' distribution and use millions of hooks each year, they still 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 (ICCAT 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 mortality rate of 21.4 percent, as seen in other high seas pelagic longline gear (Huang 2015), results in an average estimated mortality of 1,296 leatherback turtles from 2012 to 2014. This is likely an underestimate of total mortality however, as the high seas mortality rate in Huang (2015) was based upon the disposition of the turtle when boarded and 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.* 2010b) has high-moderate vulnerability to longline gear, encountering as much 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; Figure 12). These areas partly occurred in the EEZs of eight nations (Cape Verde, Gambia, Guinea Bissau, Mauritania, Senegal, Spain/Canaries, United States, and Western Sahara). The species' flexible diving behavior suggests that turtles are likely to encounter pelagic longlines throughout the Atlantic Ocean, whether they are predominantly engaged in foraging or migratory behavior (Fossette *et al.* 2014).

Bycatch in U.S. Atlantic/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 1,000 leatherback turtles annually (NMFS 2001; Yeung 2001; NMFS 2018a), with bycatch rates of about 0.15 to 0.5 turtles per 1000 hooks (Watson *et al.* 2005). In 2005, the fleet was estimated to have interacted with 351 leatherback turtles outside of experimental fishing operations (Walsh and Garrison 2006). NMFS (2018a) 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 2018a). 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). Considering the average leatherback interaction estimate for the entire U.S. pelagic longline fleet from 2011 to 2016 (351; NMFS 2018a), the estimated 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 reported interactions 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 when applied to the observer-based encounter estimates (DFO 2012). 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 to 190 leatherback turtles annually from 2002 to 2006, to 60 to 90 leatherback turtles annually from 2006 to 2010, largely as a result of gear modifications (discussed below; Hanke *et al.* 2012).

In the Mediterranean Sea, longlining is prevalent. Drifting longlines targeting swordfish, albacore (*T. alalunga*), and bluefin tuna (*T. thynnus*), are considered to be the most dangerous fishing gear for turtles in the Mediterranean Sea (Lucchetti and Sala 2011). 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 (Camiñas 1998, 2004). Casale *et al.* (2003) reviewed bycatch rates for longlines 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 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 (Casale 2008), leatherback bycatch may still be noteworthy. 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 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 leatherback distribution overlap.

Large circle hooks (non-offset) have been found to reduce leatherback bycatch when compared to traditional J-style hooks, by as much as 55 percent (Andraka *et al.* 2013; Coelho *et al.* 2015). While some 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 and exchange of ideas on improving the handling and release of bycaught sea turtles (ICCAT 2005), but no legally binding measure to require circle hooks exists (Gilman 2011). Without the widespread use of non-offset, large 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 required 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 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 (50 CFR 635.21). These requirements were developed based on the results of a 2001 to 2003 experiment in the Northeast Distant area (Watson *et al.* 2004; Shah *et al.* 2004). Swimmer *et al.* (2017) recently analyzed pelagic longline interactions before and after these regulations. Pre-regulation data are defined as years 1992 to 2001, and post-regulation data start in mid-2004 after the fishery was reopened. Throughout the 1992 to 2015 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, leatherback bycatch rates in the Atlantic 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., United States, Canada, 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). The number of vessels that do not employ bycatch reduction measures is likely higher than the number of vessels that do, so leatherback bycatch in pelagic longline fisheries is still a significant problem (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 and/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 is aimed at attempting 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 in 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), but Caribbean nations still find leatherback turtles with longline hooks (R. Berzins, pers. comm., 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.

5.2.5.1.3 Trawl Fisheries

Leatherback turtles may interact with bottom and midwater trawl gear throughout the range of the NW Atlantic DPS. The highest reported trawl bycatch of leatherback turtles in the NW Atlantic DPS is likely from the southeastern U.S. shrimp trawl fishery. Epperly *et al.* (2002) 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 effort reductions (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 new bycatch estimate for the GOM and southeastern U.S. Atlantic coast shrimp otter trawl fishery found the leatherback captures to be lower. 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); as such, species-specific mortality estimates could not be calculated for leatherback turtles (Babcock *et al.* 2018). The lethal bycatch estimate for those turtles classified as unknown/other species ranged from 24 to 99 in the GOM and 13 to 168 in the South Atlantic area (Babcock *et al.* 2018). The unknown category of turtles includes sea turtles that were not identified by the observers, as well as leatherback and hawksbill sea turtles. However, most of the turtles in the unknown category were unidentified hardshell turtles (20 out of 26 observed captures), and not leatherbacks.

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.* 2004b; TEWG 2007), Guyana (Reichart *et al.* 2003), Suriname (Madarie 2010), Trinidad (Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010), and Venezuela (Marcano and Alio-M 2000).

Despite extensive observer effort in several other NE Atlantic fisheries, no bycatch of leatherback turtles was recorded during trawl observer programs in United Kingdom and Irish waters; however, since 1980, there were eight reports of leatherback turtles captured by trawl gear in these waters (Pierpoint 2000). In the Mediterranean Sea, leatherback bycatch in bottom trawls off Tunisia has also been reported (Camiñas 2004), as well as in Egypt (Casale 2008).

Trawl bycatch reduction measures (turtle excluder devices or 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 the 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). Modified TEDs that exclude leatherback turtles, as well as large benthic immature and sexually mature loggerhead and green sea turtles, are now required.

TEDs are also used outside of the United States. Shrimp harvested with commercial fishing technology that may adversely affect sea turtles cannot be imported into the United States per Public Law 101–162, Section 609, enacted on November 21, 1989 (16 U.S.C. 1537). 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). Approximately 40 nations are currently certified to export shrimp to the United States (83 FR 22739, May 16, 2018). Although most certifications are done on a national basis, the U.S. State Department guidelines allow individual shipments of TED-harvested shrimp from uncertified counties. Specifically, on May 8, 2018, the U.S. Department of State certified 13 nations on the basis that their sea turtle protection programs (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. The Department also certified 26 shrimp-harvesting nations and one economy as having fishing environments that do not pose a danger to sea turtles. 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 the 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.

5.2.5.1.4 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 ad hoc entanglement reports, a Sea Turtle Disentanglement Network (STDN) was established by NMFS in the northeastern U.S. (Maine through Virginia) in 2002. This program relies primarily on reports from the public, and subsequent documentation and disentanglement by trained responders. From 2008 to 2017, 247 leatherback entanglements were reported in vertical fishing line (STDN, unpublished data). Of those fisheries that could be identified, 78 were lobster, 18 were conch (or a combination of conch and lobster), 6 were crab gear, and 2 were finfish lines; 143 entanglements were from unidentifiable vertical line fishing gear. While most unknown vertical line entanglements likely involve pot/trap gear, this cannot be conclusively determined. The vast majority of the reports are from Massachusetts waters. Of the 247 leatherback entanglements, 204 turtles were released alive and 43 were found dead. Given the nature of their injuries, it is probable that not all animals released alive survived. Currently there are limited empirical data on leatherback turtle 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 documented entanglement and using NMFS post-interaction mortality guidance (NMFS 2017b), 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 lines (most of which were likely pot/trap gear) in 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 were released alive and 4 were found dead, but these southeastern U.S. numbers do not incorporate potential post-interaction mortality so the total lethal interactions are likely higher than reported here. This information is likely an underestimate of actual entanglements and mortality given the opportunistic reporting nature of the program, but it is clear that leatherback interactions with vertical fishing lines are a threat to this DPS. Further, the NE STDN was initiated because leatherback turtles are listed under the ESA. Thus, if ESA protections no longer existed, leatherback turtle disentanglements in pot/trap gear would likely not occur at the current level, and mortality would be much higher.

Entanglements in Canadian waters are also frequently reported in a similar nature 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 (a fixed net similar to a weir or pound net) fisheries were involved in 26 percent of the entanglements (n = 53). Of the overall 205 reports, the majority of the 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 leatherbacks were bycaught in British and Irish waters, with the method of capture identified in 58 cases (Pierpoint 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 and 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 turtle 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 turtle mortality rate due to entanglements in vertical lines is certainly 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 turtle bycatch in vertical fishing lines, but efforts to reduce the amount of pot/trap gear 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/*).

5.2.5.1.5 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 required offshore pound nets set in a portion of the lower Chesapeake Bay from May 6 through July 15 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 (71 FR 36024). 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 2018b). 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 Virginia Chesapeake Bay using this gear (approximately four to six sets) and the frequency of live interactions. From 2008 to 2017, the STDN also documented leatherback turtle captures in weirs set off Massachusetts; turtles were either entangled in the netting (n = 2) or free swimming in the weir (n = 4). These individuals were found alive.

Purse seines are used to catch a variety of fish species, and are commonly used in the ICCAT area to catch tunas (Angel *et al.* 2014). Leatherback turtle captures have occurred in Atlantic purse seine fisheries and this bycatch may have a minor impact to the DPS. In British and Irish waters, two leatherback turtles were reported from purse seine gear since 1980 (Pierpont 2000). Clermont *et al.* (2012) reported a total capture of 67 leatherback turtles in more than 9,000 observed Atlantic purse seine sets between 1995 and 2011, with four found dead (representing 10 percent observer coverage). Most of the interactions were adults (75 percent). However, not all of the purse seine effort 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 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 turtle and purse seine interactions may occur where the two overlap, the magnitude of the purse seine impacts on the NW Atlantic DPS is smaller 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.

5.2.5.1.6 Summary of Fisheries Bycatch

We conclude that most immature and adult leatherback turtles 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 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 a widespread threat throughout the range of the DPS and a primary source of leatherback turtle 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. Bycatch mortality of nesting females reduces productivity. We conclude that fisheries bycatch is the primary threat to this DPS. The magnitude of the threat is likely higher than what is presented here due to the lack of information and documentation in most fisheries throughout the DPS.

5.2.5.2 Vessel Strikes

Vessel strikes are a threat to the NW Atlantic DPS. Injuries from vessel strikes may include blunt force trauma and/or propeller parallel slicing wounds affecting the carapace, flippers, and/or head and 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; of those, 204 had probable vessel strike 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 (ante-mortem) or after (post-mortem) the turtle died (Stacy *et al.* 2015). However, a recent study estimated that approximately 93 percent of turtles stranded in Florida with vessel strike wounds were killed by those injuries (Foley *et al.* 2019). Based on that information, 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 Tunisia in the Strait of Sicily (Karaa *et al.* 2013; Caracappa *et al.* 2017). While there is very limited observational information on vessel strikes in Atlantic waters of Canada, at least one was documented (DFO 2012). More recently, an injury assessment of live 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 boat 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 also been observed on Costa Rican nesting beaches (de Haro and Troëng 2006).

Leatherback turtle 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 vessel strikes (Dodge *et al.* 2018). Based on 24 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 turtle activity occurs in the top 15 to 30 m 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).

In summary, observational data are limited, but based upon the best available information, mortality due to vessel strikes may occur wherever vessel traffic and leatherback turtle (juvenile and adult) distribution 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, 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. As such, vessel strikes have an impact on the NW Atlantic DPS. We conclude that vessel strikes pose a threat to the DPS.

5.2.5.3 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 highest 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. For example, in French Guiana, a nesting leatherback expulsed 2.6 kg of plastic debris from her cloaca while nesting (Plot and Georges 2010). As with other terrestrial and marine species, leatherback turtles are susceptible to adverse effects from pollution and chemical contaminants. 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 (Bishop *et al.* 1991; Bergeron *et al.* 1994; Bishop *et al.* 1998; Keller *et al.* 2004). These chemicals have been identified in 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. While French Guiana hatching success has been shown to be low and OCs are present in the sand (most likely originating from pesticide use in plantations and malaria prophylaxis (Guirlet 2005), 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. Those results also suggested maternal transfer of POPs in leatherback turtles, but found higher levels of PCBs and pesticides than in French Guiana (Guirlet *et al.* 2010). While leatherback contaminant concentrations were substantially lower than concentrations in other reptile studies that demonstrated toxic effects, Stewart *et al.* (2011) suggest 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 females nesting in French Guiana (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 United Kingdom strandings (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 were 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 levels of similar chemical concentrations may occur in other areas of this DPS.

Various non-essential elements have also been documented in leatherback turtles and their eggs. Heavy metals (arsenic, cadmium, chromium, mercury, lead, etc.) and contaminants 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 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 present with selenium deficiency, a concern as selenium is important to hatchling and emergence success (Perrault *et al.* 2011), but 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 than 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 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. These areas overlap with leatherback foraging habitats.

Ingestion of marine debris is a concern for leatherback turtles, especially given the similarity of their preferred prey (gelatinous zooplankton) to some plastics. In particular, plastic bags appear similar to jellyfish in the marine environment, leading to a case of mistaken identity and triggering the sensory cue to feed (Schuyler *et al.* 2014; 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 pollution, including plastic marine debris (Camiñas 2004; Cozar *et al.* 2015). Marine debris ingestion has been documented from leatherbacks 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, turtles stranded in the Bay of Biscay (France) were found to have a plastic ingestion rate of 30 to 73 percent annually from 1979 to 1999 (n=87 leatherback turtles necropsied; Duguy *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 may not cause mortality and is typically an incidental finding. Of 41 leatherback turtles necropsied from North Carolina to Texas from 2008 to 2017, 17 had ingested plastics or marine debris (STSSN unpublished data 2018). From Maine to Virginia during that same time period, 10 necropsies detected some level of marine debris ingestion; however, the total number of necropsied turtles, out of the 677 strandings, is currently unknown, and it is likely that many more stranded turtles ingested marine debris (STSSN unpublished data 2018).

Out of 33 leatherback turtles necropsied in the New York Bight (an area with dense population), 30 percent had ingested synthetic material, mostly consisting of thin, clear plastic (Sadove *et al.* 1989). Of two leatherback turtles stranded in North Carolina during 2017 whose GI 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 died have documented large pieces of plastic (e.g., 83 x 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. Between 2008 and 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 (STRAP) 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 (Government of the Republic of Trinidad and Tobago) *et al.* 2010).

As noted in Section 5.2.1 Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range, marine debris is also a problem on nesting beaches and can reduce nesting success. Marine debris is often 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 (Forestry Division (Government of the Republic of Trinidad and Tobago) *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/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 issue 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 poses a threat to this DPS.

5.2.5.4 Climate Change

Climate change is a threat to the NW Atlantic DPS. The impacts of climate change include: increases in temperatures (air, sand, and water); 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.

5.2.5.4.1 Sand Temperature

Modeling results show that global warming poses a “slight risk” to females nesting in French Guiana and Suriname relative to those in Gabon/Congo and West Papua, Indonesia (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 and 30.5 °C (Eckert *et al.* 2012 and references within), but there is variation in measurements (Girondot *et al.* 2018) and could be variation 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.* 2012b), 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.* (2012b) 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.* 2012b). In St. Eustatius, leatherback hatchling production was female biased from 2002-2012, with less than approximately 24 percent of males produced every year (Laloë *et al.* 2016). Future warming 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.* 2012b).

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 could be associated with more extreme precipitation and faster evaporation of water, leading to greater frequency of both very wet and very dry conditions (Patiño-Martínez *et al.* 2014; Santidrián Tomillo *et al.* 2015). These impacts may affect nests in different ways, but the result (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 nester abundance (Santidrián Tomillo *et al.* 2015; Garner *et al.* 2017). Santidrián Tomillo *et al.* (2015) assessed climatic conditions on hatchling productivity 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. This is of particular concern because 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, also 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).

5.2.5.4.2 Sea Level Rise

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 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 as nesting females are forced to deposit eggs seaward of shoreline armoring - potentially subjecting nests 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, thus reducing hatching success. On Playona Beach, Colombia, Patiño-Martínez *et al.* (2014) found that nests in wet sand suffered higher mortality (emergence success of 0 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.* 2008a; Caut *et al.* 2010), and will continue to occur (or worsen) with sea level rise.

However, leatherback turtles may be less susceptible to loss of nesting habitat as they exhibit lower nest-site fidelity compared to other sea turtle species (Dutton *et al.* 1999). Nesting beaches in the Guianas are already highly dynamic and interseasonally variable, and 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 may become used by leatherback turtles, which may offset some loss of accessibility to beaches in southern portions of the range. This behavioral flexibility may allow for opportunities to colonize new beaches, but whether turtles can colonize nesting areas made 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, remains to be seen (Hawkes *et al.* 2009).

5.2.5.4.3 Water Temperature

Observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. Ocean temperatures on 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 thus 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 *et al.* 2005a; James *et al.* 2006b; James *et al.* 2007; Dodge *et al.* 2014; Dodge *et al.* 2015).

Leatherback prey distribution and abundance may be affected by changes in marine systems and resulting shifts in ranges and changes in algal and plankton abundance (IPCC 2007b). Jellyfish populations may increase in abundance and change in distribution due to ocean warming and other factors (Purcell 2005; Attrill *et al.* 2007; Richardson *et al.* 2009). The positive phase and warmer sea surface temperature in the northeast Atlantic Ocean has been found to correspond with large-scale “blooms” of jellyfish populations (Licandro *et al.* 2010). Changes in the forage base could affect the foraging behavior and distribution of leatherback turtles. Modifications in foraging behavior could lead to changes in abundance across the range of the NW Atlantic DPS, depending on whether there was an increase or decrease in gelatinous zooplankton and/or a seasonal shift in water temperature. Changes in distribution could lead to changes in exposure to threats.

Global warming is expected to expand leatherback foraging habitats into and increase residency time in higher latitude waters (McMahon and Hays 2006; James *et al.* 2006b; 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 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 (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 (remigration intervals, timing and length of nesting season) may occur (Hawkes *et al.* 2009; reviewed by Hamann *et al.* 2013). Changes in food availability may require farther foraging migrations, requiring greater energy expenditure at the cost of reproduction. For example, leatherback turtles tracked from French Guiana in 2019 travelled almost twice the distance, in a similar amount of time, as those tracked over the past 10 years (Greenpeace International 2019). Robinson *et al.* (2014) found that the median nesting date at Sandy Point (St. Croix) occurred earlier, at a rate of ~0.17 days 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). This finding contrasts with reports for nesting loggerhead and green turtle populations, for which increased SST and air temperature at nesting areas did correspond with earlier onset of nesting (Weishampel *et al.* 2004; Hawkes *et al.* 2007; Mazaris *et al.* 2008; Mazaris *et al.* 2009; Pike 2009; Weishampel *et al.* 2010), although in one study of green turtles, no correlation was found (Pike 2009). Mazaris *et al.* (2008) reported that increased SST at a Mediterranean nesting site was correlated with decreased clutch size for loggerheads. Because the relation between temperatures (local sea surface and 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.

5.2.5.4.4 Storm Events

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 and/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 and nest success as subsequently described in Section 5.2.5.6 Natural Disasters.

5.2.5.4.5 Summary of Climate Change Impacts

In summary, climate change could affect multiple life stages of the NW Atlantic DPS. Likely impacts include altering sex ratios and 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. It may also alter spatial and temporal patterns. These impacts may be more severe in certain areas with more dynamic beach environments, or may 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 may be better able to cope with climate change than other sea turtle species due to their wide geographic distribution and relatively weak nesting site fidelity. Nevertheless, we conclude that climate change is a threat to the NW Atlantic DPS.

5.2.5.5 Oil and Gas Activities

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 also occur (e.g., displacement from primary foraging areas with accompanying energetic 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. In addition, the potential for oil spills is of particular concern in the Wider Caribbean Region. The biggest oil producing nations in Central and South America are Brazil, Mexico, Venezuela, and Colombia. Currently, only three Caribbean nations have exportable oil and natural gas reserves (Barbados, Cuba, and Trinidad and Tobago); Trinidad and Tobago is the only significant exporter. However, in 2017, a major oil field was discovered off Guyana, and this 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 the area of the Trans-Isthmus oil pipeline and the Panama Canal, is of particular concern (Ruiz *et al.* 2007 in Brautigam and Eckert 2006). While the impact from oil exists in the area, 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 Gulf of Mexico pose a direct threat for leatherback turtles on 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 Gulf of Mexico, 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). While leatherback turtles were exposed to Deepwater Horizon oil, and some portion of those exposed likely died, injuries caused by the spill could not be quantified (Deepwater Horizon NRDA Trustees 2016).

In Atlantic Canada, impacts from oil and gas may also occur. Several petroleum production projects occur offshore 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 MARPOL Convention, 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 are as high as 1,200,000 metric tons (Alpers 1993). Direct oil spill events also occur as happened 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 nesting habitats of the NW Atlantic DPS, potentially exposing all life stages to oil associated threats (e.g., direct miring in oil, oil ingestion, vessel strikes, 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 direct contact with oil on the nesting beach) and potentially lowering abundance (e.g., if nesting females are killed by vessel strikes or oil exposure results in mortality). As such, impacts from oil and gas activities are of concern and are a threat to this DPS.

5.2.5.6 Natural Disasters

Natural disasters, such as hurricanes/storms, and natural phenomenon, such as increased biomass of *Sargassum* on and 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 on the beach. In 2017, Hurricane Maria devastated the islands of Dominica, St. Croix and Puerto Rico, and while the leatherback nesting season was nearly over, many beaches were impacted, including Maunabo, Puerto Rico (one of the most important nesting beaches on the island; R. Espinoza, Conservación ConCiencia, pers. comm., 2017). Further, Dewald and Pike (2014) found that a lower level of leatherback nesting attempts occurred on sites 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 so would not be expected to affect the majority of incubating nests (USFWS 1999). That said, climate change may be increasing the frequency and patterns of hurricanes (IPCC 2014) so the impacts to nests may become more common in the future.

Increased amounts of *Sargassum* have the potential to interfere with nesting activities and impede hatchling access to the ocean (Maurer *et al.* 2015). In 2011 and 2015, large amounts of *Sargassum* were present in the Caribbean (e.g., Trinidad and Tobago, Grenada) and frequently washed ashore, covering large expanses of sandy shoreline on nesting beaches, and while adults still nested in these areas, hatchlings needed help to reach the ocean (Wang and Hu 2016; K. Audroing, TVT, pers. comm., 2018; K. Charles, pers. comm., 2018). Most recently, large amounts of *Sargassum* were found in 2018 on Caribbean beaches; Barbados declared a national emergency in June 2018. If this continues and *Sargassum* washes ashore in large quantities on leatherback nesting beaches throughout the Caribbean, hatching success and survival may be impacted.

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. Without the intervention that occurs on some beaches (assuming no ESA protection), this threat would be higher. Seasonal losses at individual beaches may be large, but we do not expect such impacts to be spatially or temporally widespread. Nevertheless, we conclude that natural disasters pose a threat to the DPS.

5.2.5.7 Channel Dredging

Periodic dredging of sediments from navigational channels is carried out in numerous nations; in the United States, hopper dredges are used at large ports to provide for the passage of large commercial and military vessels. Hopper dredging disrupts benthic habitat, but that is less of a concern for leatherback turtles given their forage preferences and behavior. Leatherback turtles are sometimes entrained in hopper dredges in channel dredging operations. From 2008 to 2017, five instances of leatherback turtles killed during channel dredging operations occurred in the southeast United States (STSSN unpublished data). To reduce entrainment of turtles in hopper dredges, the U.S. often requires trawlers to capture and move turtles away from dredging operations. Nineteen leatherback turtles were captured in relocation trawls associated with hopper dredging in the U.S. southeast and GOM waters from 2013 to 2018 (NMFS unpublished data). These turtles were alive and appeared uninjured. Based on this information, we conclude that the DPS experiences low exposure to and impact from dredging.

5.3 Extinction Risk Analysis

After reviewing the best available information, we conclude that the NW Atlantic DPS has a high extinction risk, as summarized in this section. Our total index of nesting female abundance is 20,659 females at consistently monitored beaches. We based this estimate on the most recent 3 years (the average estimated remigration interval for the DPS) of nest data available to the Team. It is an index because we did not have recent, consistently collected data from all nesting beaches. Some beaches are unmonitored, and we do not have recent data for several others (many of the sites included in Dow *et al.* 2007). At this magnitude of abundance, the DPS is not at a level that places its continued persistence in question as the result of stochastic changes or catastrophic impacts; however, threats could quickly reduce abundance to such levels.

The NW Atlantic DPS exhibits a decreasing nest trend. This decline has become more pronounced in recent years (2008 to 2017), and the available nest data reflect a steady decline for more than a decade (Eckert and Mitchell 2018; NW Atlantic Leatherback Working Group 2018). This decreasing trend is observed when all available nest data are combined and at several nesting beaches (NW Atlantic Leatherback Working Group 2018), including the largest nesting aggregation in Trinidad (Grande Riviere, which is declining at 6.9 percent annually; Section 5.1.2 Productivity). In terms of productivity, the DPS exhibits low hatching success (see Table 4), but other key parameters such as clutch size, remigration interval, and clutch frequency are similar to species' averages. There are some indications of decreased productivity within the DPS at one of the most intensively monitored nesting beaches (Sandy Point, St. Croix; Garner *et al.* 2017), but whether these changes in productivity parameters will be found at other sites within the DPS remains to be determined. The declining region-wide nest trend and potential changes in productivity are reason for concern. Continued declines would further reduce the abundance and productivity of this DPS to levels that would place its continued persistence in question.

The DPS exhibits broad spatial distribution and some diversity. Based upon genetics data, as well as tagging and tracking data, this DPS shows significant spatial structure with some connectivity among nesting and foraging areas. This DPS nests throughout the southeast United States and Wider Caribbean Region, extending from North Carolina to the Guianas. Nesting is widely distributed throughout the area, at numerous beaches with less than 100 crawls annually and at 10 beaches with more than 1,000 nesting crawls annually (Dow *et al.* 2007; Piniak and Eckert 2011). The highest concentration of nesting occurs in Trinidad, French Guiana, and Panama, where a catastrophic event could have a disproportionate impact on the DPS. Leatherback turtles also demonstrate colonization of new nesting beaches (e.g., Florida) and movement among dynamic beaches (e.g., Guiana region). 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 Gulf of Mexico, providing some resilience against potential reduced prey availability. The genetic diversity and numerous and diverse nesting and foraging locations buffer the DPS from stochastic and environmental events that could have catastrophic effects on the entire DPS.

Despite the DPS's abundance, spatial distribution, and diversity, the declining nest trends and productivity are of concern and place the DPS's continued persistence in question. Clear and present threats place this DPS at a high risk of extinction, as summarized in Table 5 and described below.

Table 5. Threats to the NW Atlantic DPS. Exposure refers to the individuals affected by the threat. Impact refers to how the threat affects the demographic factors. The primary threat is identified with an asterisk.

Threat	**Exposure**	**Impact**
Destruction or modification of habitat	Nesting females, eggs, hatchlings	Reduction of nesting and hatching success (productivity)
Overutilization	Eggs and nesting females	Loss of nesting females (abundance) and reproductive potential (productivity)
Predation	Eggs, hatchlings, and some nesting females	Reduction of nesting and hatching success (productivity) and loss of nesting females (abundance)
Disease	Infrequent eggs, juveniles, and adults	Unknown
Inadequate regulatory mechanisms	Eggs and turtles of all life stages	Some laws are poorly enforced
*Fisheries bycatch	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) including loss of nesting females (productivity)

Threat	**Exposure**	**Impact**
Vessel strikes	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) including loss of nesting females (productivity)
Pollution	Eggs and turtles of all life stages	Lethal (abundance) and sublethal (productivity) effects
Climate change	Eggs and turtles of all life stages	Reduction of nesting and hatching success (productivity)
Oil and gas activities	Few individuals	Loss of individuals (abundance) and reduced hatching success (productivity)
Natural disasters	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Channel dredging	Few juveniles and adults	Potential injury and mortality (abundance)

The primary threat to this 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 is estimated to kill 1,000 adult leatherback turtles each year (Lee Lum 2006; Eckert and Eckert 2008; Eckert 2013). Bycatch, and subsequent mortality, in Trinidad bottom set gillnets and surface gillnets in Suriname and French Guiana are also major threats to the DPS. As noted earlier, Trinidad and French Guiana host the highest number of nesters in this DPS, so the continued mortality of adults in that area is of significant concern. Further, there is not an adequate regulatory mechanism 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, likely capturing thousands of leatherback turtles annually. Longline gear modifications (circle hooks) are sometimes, but not consistently, used, and size of hook is critical to successful bycatch reduction. 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 nesters, it immediately reduces productivity as well. Given the lack of observer coverage and reporting, cumulative mortality due to fisheries bycatch is likely higher than estimates. Based on available data, we have high confidence that the exposure and impact of this threat are high. We conclude that fisheries bycatch is a major, and the primary, threat to the DPS.

Additional threats to the DPS include habitat loss, the legal and illegal harvest of turtles and eggs, predation, disease, vessel strikes, pollution, climate change, oil and gas activities, natural disasters, and channel dredging. Coastal development and shoreline armoring, erosion (natural and anthropogenic), and artificial lighting are some of the most significant stressors on nesting beach habitat, reducing nesting and hatching success (productivity). Habitat loss 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 a variety of 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, Panama, Puerto Rico, and Trinidad and Tobago. Intervention (e.g., nest screening) to reduce predation is not used in most places partially due to the concern of attracting poachers and the infeasibility at high-density or remote beaches. Egg predation reduces productivity, and the exposure to this threat is moderate. Episodes of disease have been found in leatherback turtles from various areas, but disease does not appear to be causing widespread leatherback mortality throughout the DPS; as such, disease is considered a threat, but its population-level impacts are unknown. Vessel strikes are also a threat, as vessel strikes kill numerous leatherback turtles each year. While exposure to vessels may be most severe in developed areas, the 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 effects data are limited. Climate change is another threat that is likely to increase in the foreseeable future. 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. Oil and gas development is a threat that has the potential to grow in some Caribbean areas, so the impact may increase in the future. Natural disasters (hurricanes) and phenomena (large *Sargassum* events) have an intermittent impact on the NW Atlantic DPS. Channel dredging results in potential injury and mortality when interactions occur; however, this is considered infrequent. Although many international, national, and local regulatory mechanisms are in place, they do not reduce the impact of these threats to negligible levels.

Considered as a whole, the cumulative impact of these multiple threats is large. Innis *et al.* (2010) reported that many individuals were simultaneously exposed to multiple threats, including: entanglement, injury, plastic ingestion, adrenal gland parasitism, diverticulitis, and burdens of environmental toxins. 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 nester abundance). Often, however, several threats contribute to demographic reductions. For example, reductions in hatching success may be caused by one or more of the following threats: erosion, poaching, predation, climate change, and pollution.

We find that the NW Atlantic DPS is affected by a high level of clear and present threats, which are likely to exacerbate the declining nest trend. Yet the abundance, spatial distribution, and diversity provide some resilience. To determine whether the DPS is at high or moderate risk of extinction, we evaluated whether the DPS is at risk of extinction now (imminent risk) or in the foreseeable future as described by NMFS' Guidance. To answer this question, we asked how long it would take for nester abundance to be reduced by 50 percent and whether this time period is imminent or in the foreseeable future. Using estimates of the species mean time to maturation for the population (n = 19 years; Avens *et al.* 2020) and mean nesting longevity (n = 11 years; Avens *et al.* 2020), we estimated a generation time of approximately 30 years. We considered three scenarios.

- First, we used data on a well-documented, known threat: gillnet bycatch mortality of 1,000 adult turtles annually off the largest nesting aggregation (Trinidad). Assuming that half of turtles killed are female, total DPS nester abundance would decrease by 50 percent in 28 years, which is approximately one generation.
- Second, we used regional nest trend data from the NW Atlantic Leatherback Working Group (2018). Because the most recent trends are generally used for projections, we first evaluated their nest trend from 2008 to 2017 (−9.32 percent per year, 95 percent CI: −12.9 to −5.57 percent). We found that nester abundance would fall by 50 percent within 8 years (95 percent CI: 6 to 13 years). However, sea turtle populations often exhibit nesting variation over time, which would be reflected in a longer data set. Therefore, we also considered their nest trend from 1990 to 2017 (−4.21 percent per year, 95 percent CI: −6.66 to −2.23 percent), in which nester abundance would to fall by 50 percent within 17 years (95 percent CI: 11 to 31 years).
- Third, using our nest trend for the highest abundance nesting area in the DPS, Trinidad (−7.3 percent per year, 95 percent CI: −34 to 18 percent), nester 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 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 nester abundance may not be an index of the remainder of population; assumes stable age distribution; and time series do not always span one generation (nor the multiple generations required to reach stable age distribution). Despite these caveats, all scenarios resulted in a 50 percent reduction in nester abundance in less than one generation. While the first scenario did not involve the use of nest trend data, it only considered one threat, and we know that the DPS faces many, large-impact threats.

The Team discussed whether this time period (8 to 28 years) is imminent or in the foreseeable future. We struggled with this decision, individually and as a Team. The 10 Team members present at the meeting voted as follows:

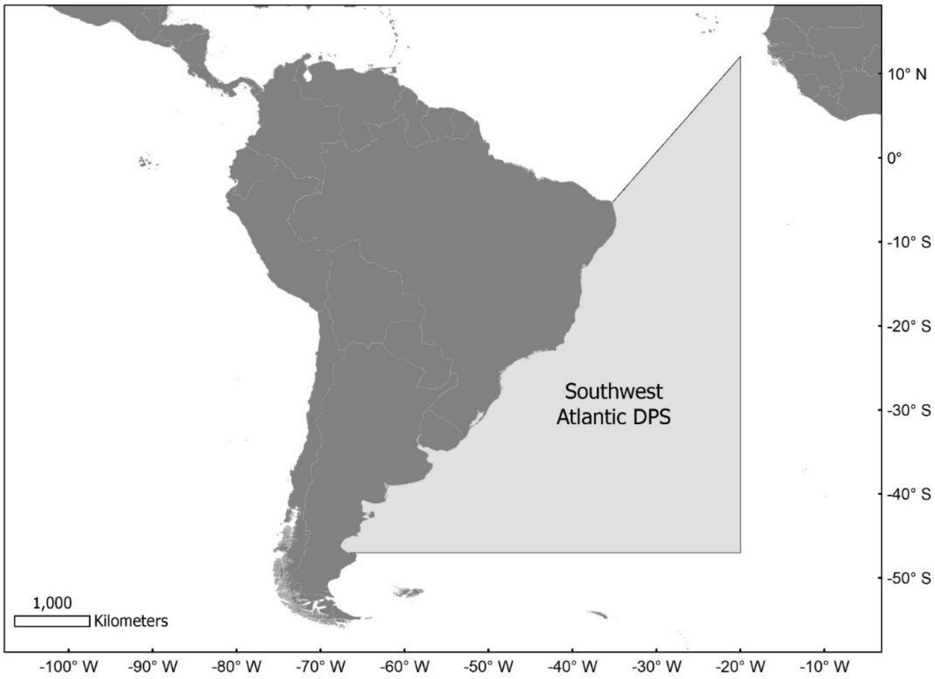
- Eight voted for high extinction risk with moderate confidence due to the declining trend that has accelerated in recent years (not high confidence due to the resilience provided by the abundance, spatial distribution, and diversity)
- Two voted for moderate extinction risk with low confidence due to the declining trend that has accelerated in recent years

We concluded that less than one generation (in particular 8 to 17 years to decline by 50 percent) is imminent. The DPS faces clear and present threats that are likely to create imminent and substantial demographic risks (declining trends and reduced abundance), consistent with the definition for high risk of extinction (see Extinction Risk Assessment). The moderate risk definition does not apply because it is at a high risk of extinction at present, rather than on a trajectory to become so in the foreseeable future. We have moderate confidence in this conclusion because while the declining trend has accelerated in recent years, the abundance, distribution, and diversity provide it with some resilience. However, other leatherback populations have quickly declined despite larger abundances (e.g., Mexico nesting aggregation declined from 70,000 nesters in 1982 to under 1,000 nesters by 1994; Spotila *et al.* 2000). Therefore, the Team concludes that the NW Atlantic DPS meets the definition for high risk of extinction because the decreasing nest trend (productivity) is at or near a level that places its continued persistence in question.

6.0 Southwest Atlantic DPS

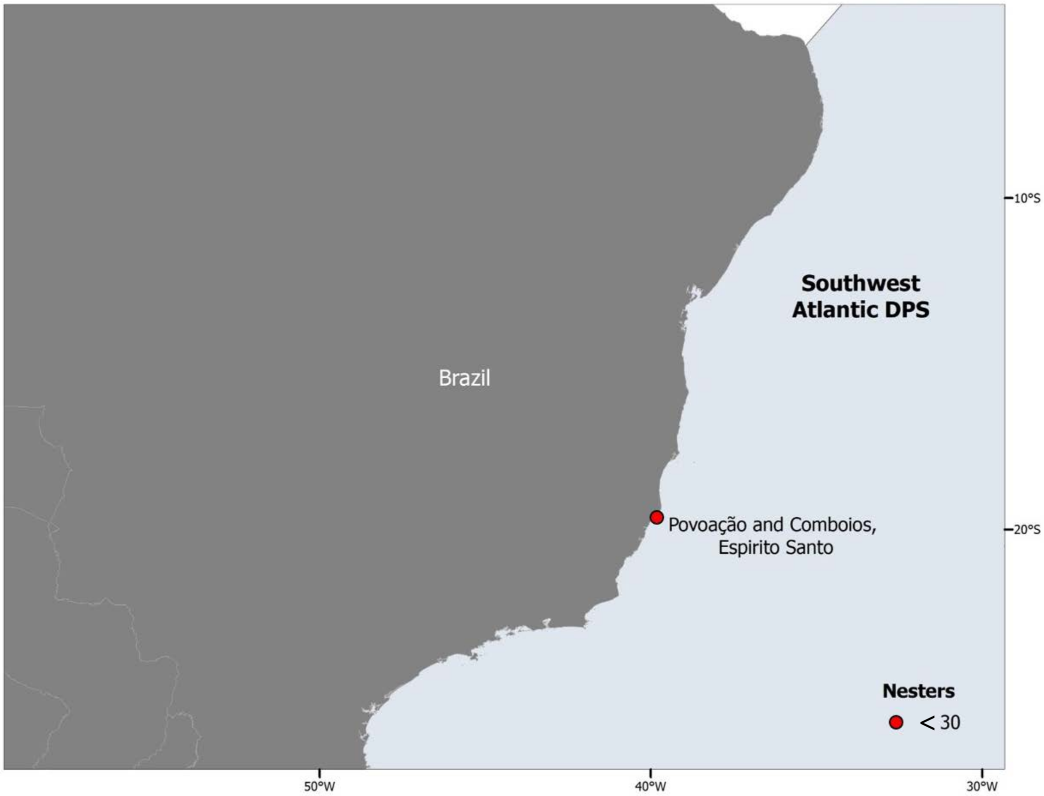
We define the SW Atlantic DPS as leatherback turtles originating from the SW Atlantic Ocean, north of 47° S, east of South America, and west of 20° W; the northern boundary is a diagonal line between 5.377° S, 35.321° W and 12.084620° N, 20° W (Figure 17). The southern boundary is based on the Antarctic circumpolar current which prevents sea turtles from nesting further south (Eckert 2013). 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. The eastern boundary was chosen at the 20° W meridian as an approximate midpoint between SW Atlantic and SE Atlantic (females nesting on beaches in western Africa) nesting beaches and to reflect the DPS's wide foraging range throughout the South Atlantic Ocean; however, the low abundance of this DPS makes it less likely to be encountered in these waters compared to individuals from the more abundant SE Atlantic DPS.

Figure 17. SW Atlantic DPS boundary map.



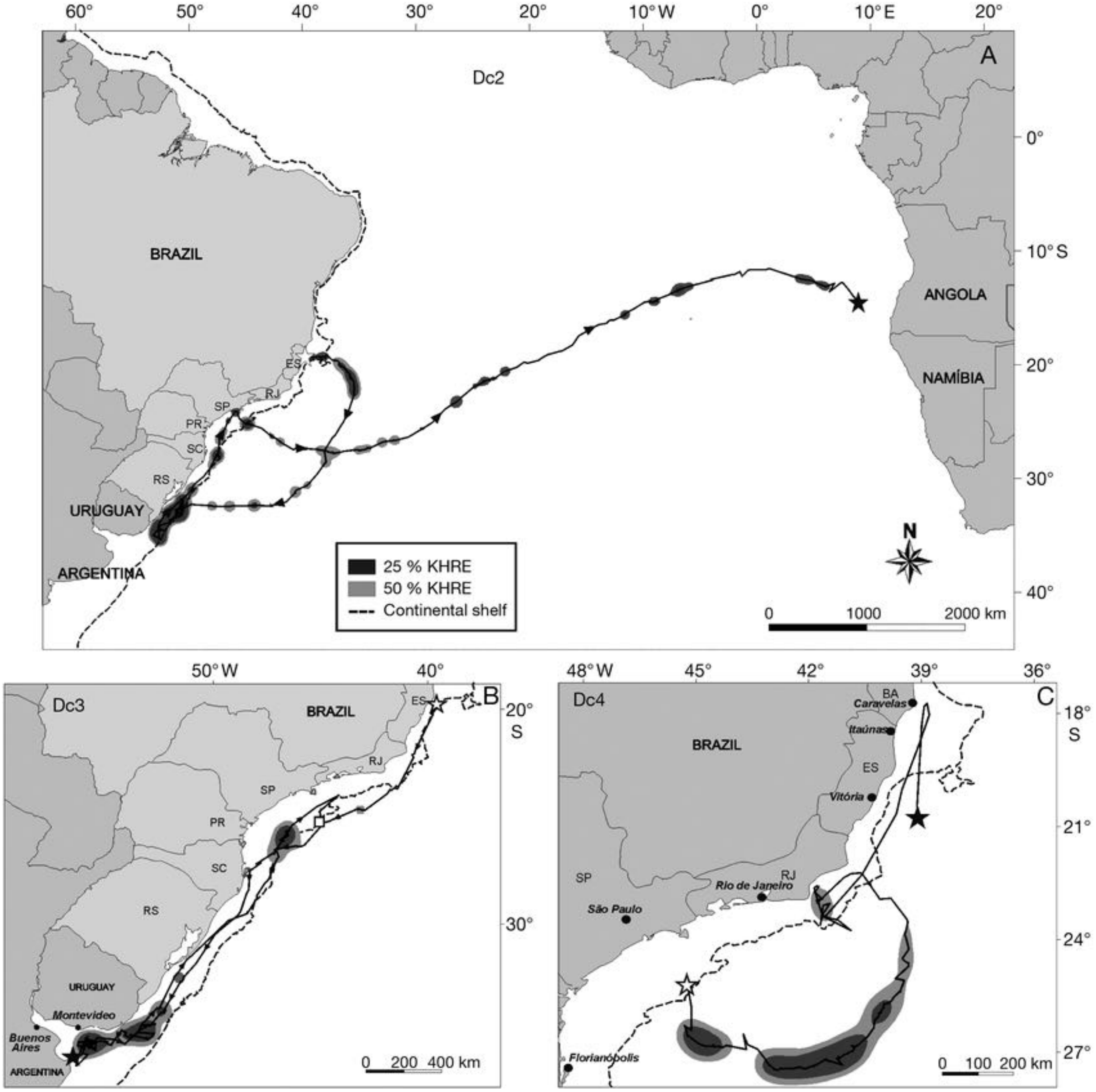
This DPS nests only on the southeastern coast of Brazil, primarily in the state of Espírito Santo (Figure 18). While there is occasional, limited nesting south of the primary nesting beaches, the sand becomes coarser further south and the excavation of nests becomes more difficult because the sand falls back into the holes (Thomé *et al.* 2007).

Figure 18. Nesting site of the SW Atlantic DPS.



While nesting is limited geographically, the overall range of this DPS (all areas of occurrence) is extensive, as demonstrated by individuals tracked to numerous foraging areas. Leatherback turtles 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 the open ocean foraging the species is known for (Almeida *et al.* 2011; Figure 19). Individuals of this DPS are also known to migrate to the waters off western Africa and use 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.

Figure 19. Satellite tracks of post-nesting females tagged in Brazil. Open stars reflect starting point; black stars reflect location of last transmission. Image: Figure 4 from Almeida *et al.* (2011), Endangered Species Research (*https://doi.org/10.3354/esr00359*).



6.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the SW Atlantic DPS.

6.1.1 Abundance

Our total index of nesting female abundance for the SW Atlantic DPS is approximately 27 females (Table 6 and Table 7). Almost all nesting occurs in the south-central Brazilian state of Espírito Santo with the exception of an occasional nest to the north or south. The majority of nesting occurs on the beaches of Povoação and Comboios, which are separated by the Doce

River but represent a continuous stretch of nesting beach less than 100 km in length. Projeto TAMAR (the Brazilian Sea Turtle Conservation Program) has established an index nesting survey area along 47 km of that stretch 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 1986/87 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). Additional minimal, scattered nesting has also been reported on beaches outside of Espírito Santo (Barata and Fabiano 2002; Thomé *et al.* 2007; Bezerra *et al.* 2014), but most of the beaches outside of the core area appear to be of suboptimal quality 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.

Table 6. Available nesting data for the SW Atlantic DPS. Number of nests (or other units, as identified) recorded for the first and last years monitored at surveyed nesting beaches. We also include the highest and lowest number of nests (or other units, as identified). We calculated the index of nesting female abundance by summing the number of nests over the most recent remigration interval (3 years; see Table 8) divided by the clutch frequency (5 clutches per season; see Table 8). This is the total index of nesting female abundance for the DPS.

Nation, nesting beach, length	**Number of nests (first and last year monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Brazil				
Povoação and Comboios Index Beaches, 47 km	20 (1986/1987) 38 (2016/2017)	High: 124 (2012/2013) Low: 6 (1993/1994)	Projeto TAMAR nesting data 2018	(30+64+38)/5 = 27 (index beach only, total nesting is unquantifiable at this time)

Table 7. The number of nesting sites by the index of nesting female abundance. We estimated the index of nesting female abundance (for the DPS) by summing the nesting female abundance from Table 6. We calculated the percentage at the largest nesting site by dividing that nesting female abundance (Table 6) by the index of nesting female abundance for the DPS.

Index of nesting female abundance	**Number of nesting sites**
Unquantified	
1–10	
11–50	1*
51–100	
101–500	
501–1,000	
1,001–5,000	
5,001–10,000	
10,001–20,000	
>20,000	
Total number of sites	1
Total index of nesting female abundance (DPS)	27
Confidence in total index of nesting female abundance	High (much of coastline not regularly monitored, but index site captures most nesting, [approximately 80 percent; Colman *et al.* 2019], and the expected overall trend)
Largest nesting site, percentage of total index	Espírito Santo, 100 percent

*Table 6 does not include nesting outside of the index nesting beach (and thus non-index nesting is not included in the total estimate for the DPS): All non-index beach nesting in Espírito Santo and scattered, occasional nesting in neighboring states are all part of the same continuous nesting area, but only the index beach is monitored regularly.

Our total index of nesting female abundance is similar to the IUCN Red List assessment that estimated 35 mature individuals (male and female) in the population using nesting data through 2010 (Tiwari *et al.* 2013c). Since then, nesting has increased overall, though the latest three years of nesting (2014/2015 to 2016/2017) were lower than the previous three years. To calculate the total index of nesting female abundance (n = 27), we divided the total number of nests (30 + 64 +38 = 132) counted at the index site over the most recent three nesting seasons (2014 to 2016, assuming a 3 year remigration interval) by a clutch frequency of 5. We used general estimates for the remigration interval and clutch frequency because no specific data were available for this

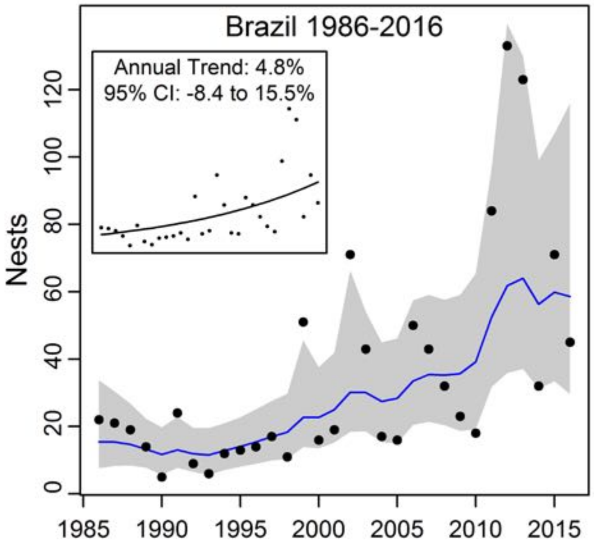
DPS. Though this number represents an index of nesting female abundance, limited by the extent of monitoring, it is likely a close approximation of total nesting female abundance, capturing the vast majority of nesting activity for the DPS in recent years.

The total index of nesting female abundance (approximately 27 females at the index beach) places the DPS at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017a). These processes, working alone or in concert, place small populations at a greater extinction risk than large populations, which are better able to absorb the loss of individuals. Due to its very small size, the DPS has limited capacity to buffer such losses. We do not have historical abundance estimates with which to provide context on the extent of past nesting prior to the mid-late 1980's. It appears that nesting habitat quality outside of the core nesting area is relatively low (Thomé *et al.* 2007), so the extent to which this DPS could increase its nesting range or abundance is unknown. Given the intrinsic problems of small population size, we conclude that nesting female abundance is a major factor in the extinction risk of the SW Atlantic DPS.

6.1.2 Productivity

The SW Atlantic DPS exhibits an increasing, although variable, nest trend (Figure 20). Long-term monitoring data for this small DPS are limited to the stretch of index beach in Brazil, Comboios and Povoação, which are combined together into a single time series that meets our standards for conducting a trend analysis. Over the 31-year data collection period (1986/1987 through 2016/2017), the mean annual number of nests for these beaches was 35. While this is below our criterion of 50 annual nests for conducting a trend analysis, we made an exception for this site due to the high quality and consistency of the data, and the fact that these are the only data available for the DPS. As with all DPSs, we report the BSSM trend analysis results as the median and CI, which reflects that there is a 95 percent chance that the trend falls between the low and high CI values. The wider the CI, the less confident we are in the estimated median trend. The higher the "f statistic" the more confident we are in the sign (positive or negative) of the estimated median trend. 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 these two beaches represent the majority of known nesting activity for this DPS, we conclude that the increasing trend is representative of the entire DPS.

Figure 20. Nest trend at index beaches in Brazil. The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (nests). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the inter-annual variability.



We base our conclusion on the available data from consistent, complete surveys done by Projeto TAMAR along the index nesting beach (Povoação and Comboios, Espírito Santo, Brazil) since 1986. These data are representative of the entire DPS, as nesting outside of the index beach is limited (approximately 20 percent; Colman *et al.* 2019). Our conclusion is in agreement with that of the IUCN, which characterizes the population as increasing (Tiwari *et al.* 2013c). 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 shows an increase in nesting, the most recent 3 years of data (30, 64, and 38 nests from 2014 to 2016) show a marked reduction in nests compared to the previous 3 years (78, 124, and 102 nests from 2011 to 2013). The reason for this reduction is unknown and not necessarily reflective of declining nesting female abundance. It could reflect changes in productivity metrics (a longer remigration interval or reduced clutch frequency) related to environmental shifts or prey availability.

The productivity parameters for this DPS (Table 8) 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 (60.4 nests/year x 87.7 eggs/nest x 66.0 percent hatching success; Colman *et al.* in press). This is an index as it does not include the limited nesting outside of the index area, as detailed in section 6.1.1 above. Additionally, the productivity parameters were estimated more than 10 years ago, and they may have changed as a result of possible environmental or productivity changes since that time. For example, 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 ± 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). Therefore, we have a moderate degree of confidence for these productivity metrics.

Table 8. Productivity parameters for the SW Atlantic DPS.

Productivity metric	**Variable by Nation**	**Reference**
Size of nesting female (CCL cm)	152.8 (mean) ± 10.0 SD; 124.7 to 182.0 (range)	Colman *et al.* 2019
Remigration interval (years)	2 to 3	Thomé *et al.* 2007
Clutch size (eggs)	87.7 mean; range 5–131; Total eggs and shelled albumen gobs (mean): 109.8.	Thomé *et al.* 2007
Clutch frequency	Specific data for this assemblage not available. We use a generic value of 5 nests/season consistent with Thomé *et al.* (2007) and Tiwari *et al.* (2013).	
Incubation period days)	67.8 (mean); 61.5–78.0 range	Thomé *et al.* 2007
Hatching success (percent)	66.0 (mean); 38.8 to 82.4 (range)	Colman *et al.* 2019

While the nest trend for this DPS is increasing, 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 the currently positive nest trend indicates a reduced extinction risk, tempered by the small population size and potentially limited capacity for expansion as detailed in Section 6.1.1 Abundance.

6.1.3 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 internesting 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 and 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 buffers the DPS against local catastrophes or environmental changes that could limit prey availability. However, the highly 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.

6.1.4 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.* 2013b; Vargas *et al.* 2017). Nevertheless, overall haplotype diversity for leatherback turtles is low when compared to other species, so in this context, genetic diversity is only moderate. 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 some added temporal resilience if environmental conditions result in poor 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 (Miazan *et al.* 2001; Lopez-Mendilaharsu *et al.* 2009; Prosdocimi *et al.* 2014). With its extremely limited nesting range, small population size, and moderate genetic diversity, we conclude that there is little overall diversity and resilience for this DPS, increasing its extinction risk.

6.2 ESA Section 4(a)(1) Factors

The following sections describe and assess the 4(a)(1) factors or threats. For each, we evaluate the best available information on the threat. We also describe, if possible, exposure and impact of each threat.

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

Within the limited nesting range of this DPS, the current or threatened destruction, modification, or curtailment of habitat is a threat. A significant portion of the nesting beach range is protected as a federal reserve under 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. Although coastal light pollution has been documented to be increasing in Brazil, nesting has not appeared to be 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. All light sources with a light intensity greater than 0 lux (lux = lumen per m^2) 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 in 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). The 2015 collapse of a tailings dam at an ore mine upstream of the Doce River had an undetermined, but potentially long-term, impact on the SW Atlantic leatherback turtles. Tens of millions of cubic meters of heavy metal-laden mining waste entered the river and ultimately passed through the mouth of the Doce River, in the middle of the primary stretch of nesting beach for this DPS. 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 not be evident for decades following the spill. Projeto TAMAR has added sampling for heavy metals in eggs and nesting females to their annual monitoring, and are 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 (source: ICMBio, MMA, Brazil).

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).

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

Though specific information on leatherback turtles is not available, there was 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, providing them with a feeling of connection and responsibility to the turtles (Almeida and Mendes 2007; Marcovaldi *et al*. 2008). 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 to low levels, and there is no 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, because of the small size of the population even low levels of egg poaching have the potential to impact the population. Therefore, we conclude that overutilization poses some threat to the SW Atlantic DPS.

6.2.3 Disease and Predation

While there is little specific information on disease for the SW Atlantic DPS, hatchlings are likely preyed upon by land, air, and marine predators, and there is likely a small degree of egg predation by local invertebrates, reptiles, and mammals. According to Thomé *et al.* (2007), occasional relocation of nests and nest protection occur when inundation (primarily) or predation risk (secondarily) 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).

All eggs and hatchlings have some potential for exposure to predation. The impact of predation on eggs and hatchlings is likely low. Some predation on large juveniles and adults in the marine environment, especially by sharks, occurs (Bornatowski *et al.* 2012), but the frequency and impact on those populations is not well understood.

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 (reduced egg and hatching success). We conclude that predation is a threat to the SW Atlantic DPS.

6.2.4 Inadequacy of Regulatory Mechanisms

The SW Atlantic 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.

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^2) on these beaches are prohibited by a federal ordinance (Portaria IBAMA 11/1995).

The take of leatherback turtles is illegal throughout the SW Atlantic region. Regional regulations include: Brazil Portaria, Manter proibida a captura de tartarugas marinhas das espécies *Caretta caretta*, *Dermochelys coriacea*, *Eretmochelys imbricata* e *Lepidochelys olivacea* 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 Nº 9605/1998). While these protections are mostly effective, very low levels of egg poaching still exist (Thomé *et al.* 2007).

As detailed in Section 6.2.5.1 Fisheries Bycatch, fisheries bycatch is the primary threat to SW Atlantic DPS. 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 turtle excluder devices in trawl fisheries is mandatory (INSTRUÇÃO NORMATIVA MMA Nº 31, DE 13 DE DEZEMBRO DE 2004), but most fishermen do not use it, and there is little or no enforcement by authorities (IAC Brazil Annual Report 2018). Despite 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 al.* 2012).

In summary, numerous regulatory mechanisms protect leatherback turtles, their eggs, and nesting habitat throughout the range of this DPS. Though the regulatory mechanisms provide some protection to the DPS, many are inadequate to sufficiently 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 consider the magnitude of this threat in the following section. In summary, we consider the inadequacy of the regulatory mechanisms to be a threat to the DPS.

6.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

Other threats affect the SW Atlantic DPS. Fisheries bycatch is the primary threat, while additional threats include: vessel strikes; climate change; pollution; and channel dredging.

6.2.5.1 Fisheries Bycatch

Fisheries bycatch is the primary threat to the SW Atlantic DPS. Threats come from both commercial and artisanal fisheries along coastal foraging and breeding areas as well as on the high seas. The extensive foraging range of this DPS makes it vulnerable to interactions with fisheries off Brazil, Uruguay, and Argentina, in the pelagic waters of the South Atlantic, 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 given the lack of differentiation from SE Atlantic individuals that forage within the same range. Because the SE Atlantic DPS outnumbers the SW Atlantic DPS by several orders of magnitude (see section 7.1.1 Abundance of SE Atlantic DPS) most fishery interactions likely involve SE Atlantic individuals. However, the proportional impact is expected to be similar for both DPSs interacting with those fisheries. Incidental captures in gillnets and surface, deep-water long-line 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).

Stranding information often provides an insight into potential fishery interactions. Although typically viewed as an oceanic species (Eckert *et al.* 2012), foraging is common in the coastal waters of the SW Atlantic Ocean (López-Mendilaharsu *et al.* 2009; Almeida *et al.* 2011). Waters off the coast of the southernmost state of Brazil, Rio Grande do Sul, represent an important part of that large foraging and developmental area; it is also extensively used for fishing activities (Scherer *et al.* 2014; Monteiro *et al.* 2016). The coast along southern Brazil has been shown to have the highest number of leatherback strandings within the nation (Barata *et al.* 2004). Surveys from 1995 to 2014 in Rio Grande do Sul show that the highest stranding rates for sea turtles occurred from October to March, which coincides with the greatest fishery effort for bottom pair trawl and gillnets along the coast (Monteiro *et al.* 2016). In addition, coastal bottom trawl and artisanal gillnet fisheries were the main causes of death of leatherbacks found stranded in Uruguay (Velez-Rubio *et al.* 2013). Because fishing effort has increased over time (Secchi *et al.* 2004; Ramos and Vasconcellos 2013), we would expect strandings to increase over time. However, leatherback strandings have remained relatively consistent across years (Monteiro *et al.* 2016), possibly suggesting declines of the in-water population. The in-water population, however, includes individuals of the SW and SE Atlantic DPSs, with the latter being far more prevalent. Therefore, most leatherback turtles stranding in southern Brazil, and stranding or incidentally captured off Uruguay and Argentina, are thought to be part of the SE Atlantic DPS (Vargas *et al.* 2008; Prosdocimi *et al.* 2014; Vargas *et al.* 2019).

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 gillnets, including coastal set, fixed, encircling, and pelagic drift gillnets. The actual number of takes is likely substantially higher as many takes 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 alternative fishing techniques that are more sustainable and cause fewer impacts to sea turtles. However, such fisheries occur throughout important coastal foraging areas off South America. Additionally, coastal artisanal gillnet fishery interactions with leatherback turtles are known to occur off multiple nations along the western coast of Africa, where some individuals from the SW Atlantic DPS forage (Riskas and Tiwari 2013). The Río de la Plata estuary, an important foraging area off Uruguay, has numerous documented instances of leatherback turtle entanglements, including mortalities from the coastal bottom-set gillnet fisheries (Fallabrino *et al.* 2006; Lopez-Mendilaharsu *et al.* 2009; Velez Rubio *et al.* 2013).

Larger-scale commercial coastal and ocean gillnet fisheries are also a significant threat for the SW Atlantic DPS, with high bycatch rates reported off Brazil in drift and set gill nets (Fiedler *et al.* 2012; Ramos and Vasconcellos 2013). Drift gillnet fishing off Brazil started in 1986, targeting hammerhead sharks (Domingo *et al.* 2006). In 1998, a Brazilian federal ordinance limited the use and transport of bottom and drift gill nets over 2.5 km long. The challenge of enforcing such limiting regulations, however, was evident as vessels from the ports of Itajaí, Navegantes and Porto Belo, in Santa Catarina state, south Brazil, deployed nets up to 7,846 m long between 2005 and 2006 (Kotas *et al.* 2008). Then in 2010 the ordinance was suspended, permitting unrestricted fishing with driftnets (Fiedler 2012). 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 recorded as bycatch 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). 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. However, despite the reduction in the fleet and the possible collapse of the fishery, there is no current regulation to prevent the regrowth of the driftnet fishery, which would have severe impacts on sea turtles (Fiedler 2012). Various other gillnet fisheries such as bottom gillnets for sharks and molluscs 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 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 and southeastern 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, 8 leatherback turtles were captured, with a mean capture rate of 0.03 turtles/1000 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 the SW Atlantic is fished by longliners from Uruguay, Chile, Japan, Taiwan, and Spain in addition to Brazil (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 26º and 37º S between April 1998 and November 2000 observed 27 bycaught leatherback turtles (Domingo Balestra 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 (CEPERG/ IBAMA 2002). Lopez-Mendilaharsu *et al.* (2019) analyzed 28 records of juvenile (≤100 cm CCL) leatherback turtles incidentally captured in Brazilian longline fisheries, indicating that individuals may be exposed to this threat one year after nest emergence. 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 (Riskas and Tiwari 2013). 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 1o- 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). Stranding rates of leatherback turtles along the Brazilian coast coincide strongly with the height of bottom trawling effort off Rio Grande Do Sul (Monteiro *et al.* 2006). 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). 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 other sea turtle species. Giffoni *et al.* (2013) found that from 1990 to 2012 only 2 of the 8,367 total sea turtles documented to be taken in those fisheries were leatherback turtles, and both were alive.

While specific information to estimate overall bycatch and mortality rates of SW Atlantic leatherback turtles is not available, it is clear that fisheries, especially gillnets and longlines, are a major threat to the DPS across its range. Immature and adult individuals are exposed to high fishing effort throughout their foraging range and in coastal waters near nesting beaches. Mortality is also high, with reported mortality rates of up to 31 percent. Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when nesting females are bycaught 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 to levels that may not be sustainable. Therefore, we conclude that fisheries bycatch is a major, and the primary, threat to the SW Atlantic DPS.

6.2.5.2 Vessel Strikes

There is little information regarding boat strikes for the SW Atlantic DPS. Many of the primary foraging areas for this DPS off 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). Affected individuals may include immature and mature turtles. Impacts range from injury to mortality. We conclude that vessel strikes are likely a threat to the DPS.

6.2.5.3 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. It is unclear how increased sea surface temperatures, more extreme storm events, coastal erosion, and altered ocean currents would impact the DPS. Therefore, we focus on the potential impact of increases in sea level and sand temperatures.

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 of the area currently being used, there is little opportunity for a spatial shift in nesting in response to changing temperatures.

This DPS has some nesting year round, 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. We also do not have insight into the impacts on productivity and survivorship for such shifts in nesting.

The threat of climate change is likely to modify the nesting conditions for the DPS, and the impact would be inescapable as 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.

6.2.5.4 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 with the injury or death of turtles found in waters occupied by the SW Atlantic DPS (Bugoni *et al.* 2001; Eckert *et al.* 2009; Schuyler *et al.* 2013; Scherer *et al.* 2014). However, individuals found to have been injured or killed by marine debris and plastics have not been assigned to a particular DPS 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 from maritime transport, to tourism, to domestic and industrial waste discharges 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 are also a concern, though no specific impacts to leatherback turtles were noted from that event (Garcia *et al.* 2016). 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)

Attributing impacts of pollution specifically to the SW Atlantic DPS is difficult, and we cannot quantify such impacts. However, given the prevalence of such pollutants, we conclude that pollution is a threat to the DPS.

6.2.5.5 Channel Dredging

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

6.3 Extinction Risk Analysis

After reviewing the best available information, we conclude that the SW Atlantic DPS has a high extinction risk. The index of nesting female abundance is 27 females. Such a nesting population size places this DPS at risk of stochastic or catastrophic events that increase its extinction risk. There is little information to determine if the very small size of the DPS is the result of past and current threats, or if it is a fringe population with limited suitable nesting beach availability.

Although there has been substantial variability in nesting at the index nesting beach since 1986, the nest trend shows a strong, nearly five percent average annual increase, with the largest increase occurring in the past decade. There is only one nesting aggregation, limited to a relatively small stretch of beach along a single coast. The index nesting beach on which the majority of the nesting occurs spans only 47 km. The total length of beach along Povoação and Comboios, within which the index survey area occurs, is less than 100km. Some nesting also occurs outside of 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. As a result of the single nesting site, there is no metapopulation structure within this DPS; however, there is ample genetic diversity. This DPS uses multiple, distant, and diverse foraging areas, providing some resilience against reduced prey availability. Based on these demographic factors, we find the DPS to be at an increased risk of extinction as a result its limited abundance, spatial structure, and resilience.

In addition to the demographic factors, the current threats faced by this DPS also place it at a high risk of extinction, as summarized in Table 9. 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 and impact of this threat are high. Additional threats include: habitat modification, overutilization, predation, inadequate regulatory mechanisms, pollution, 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 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 presents a potentially increasing threat to the DPS. Ingestion of plastics and entanglement in marine debris result in injury and reduced health, and sometimes mortality. 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. Vessel strikes pose a threat that is likely to increase over time as recreational and commercial vessel activity increases, resulting in more opportunity for interactions.

Table 9. Threats to the SW Atlantic DPS. Exposure refers to the individuals affected by the threat. Impact refers to how the threat affects the demographic factor(s). The primary threat is identified with an asterisk.

Threat	**Exposure**	**Impact**
Destruction or modification of habitat	Nesting females, eggs, hatchlings	Reduction of nesting and hatching success (productivity)
Overutilization	Few eggs and nesting females; few turtles at sea	Loss of nesting females (abundance) and reproductive potential (productivity)

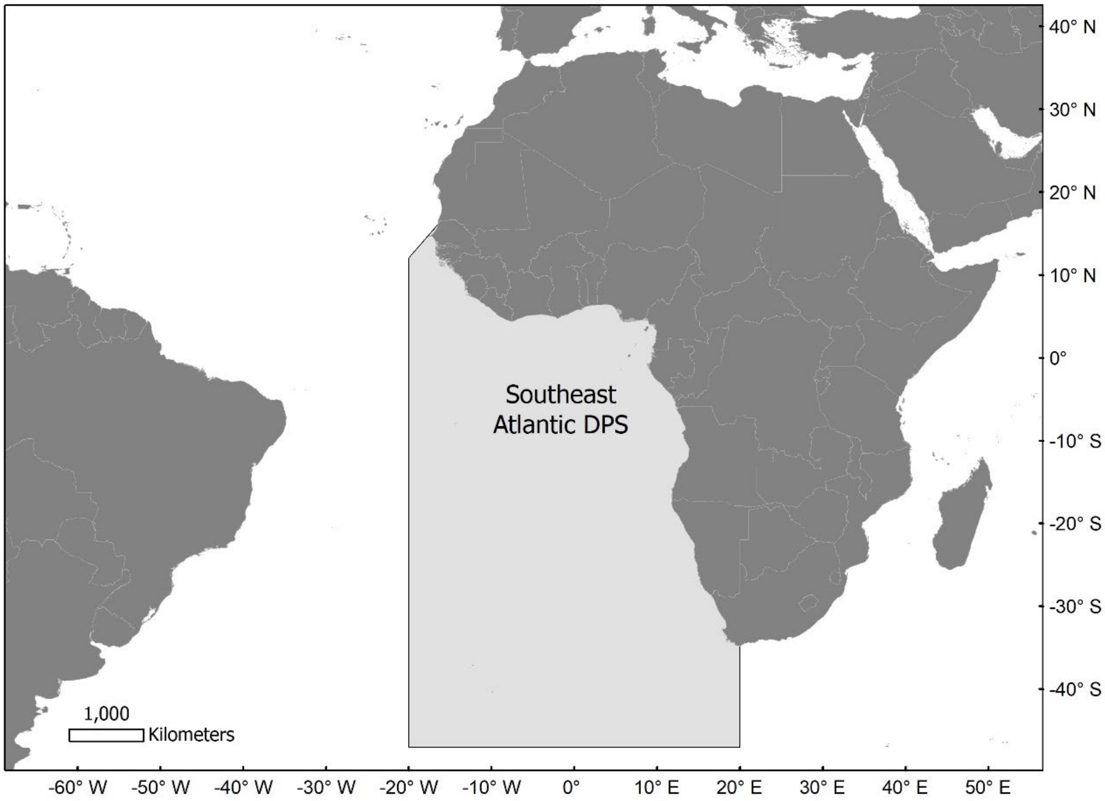
Threat	**Exposure**	**Impact**
Predation	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Inadequate regulatory mechanisms	Eggs and turtles of all life stages	Some laws are poorly enforced
Fisheries bycatch*	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) including loss of nesting females (productivity)
Vessel strikes	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) including loss of nesting females (productivity)
Pollution	Eggs and turtles of all life stages	Lethal (abundance) and sublethal (productivity) effects
Climate change	Nesting females, eggs, hatchlings	Reduction of nesting and hatching success (productivity)
Channel dredging	Forging juveniles and adults	Loss of individuals (abundance)

Thus, we find that the SW Atlantic DPS is at a high level of extinction risk. Its low abundance places its continued persistence in question despite the apparent increasing nest 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, 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. We conclude that the SW Atlantic DPS meets the definition for high risk of extinction (see Extinction Risk Assessment). The moderate risk definition does not apply because it is at a high risk of extinction now (at present), rather than on a trajectory to become so in the foreseeable future. We have high confidence in this conclusion because of the extremely low abundance combined with the major threat of fisheries bycatch.

7.0 Southeast Atlantic DPS

We define 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 (Figure 21). 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 (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.

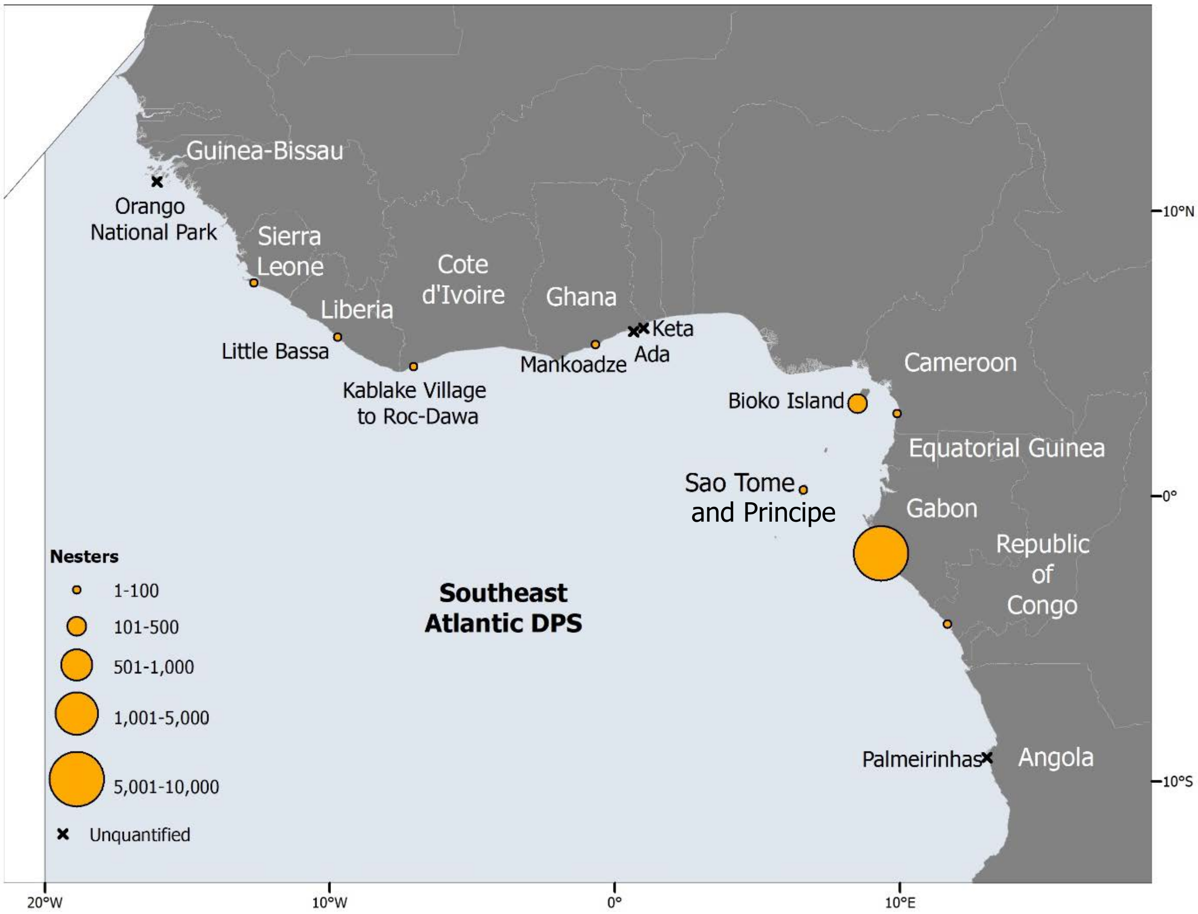
Figure 21. SE Atlantic DPS boundary map.



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 (Figure 22). However, the vast majority of nesting occurs in Equatorial Guinea (including Bioko Island), Gabon, and the Republic of Congo (TEWG 2007; Fretey *et al.* 2007, Witt *et al.* 2009; Tiwari *et al.* 2013b). Gabon may have 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 (Gabon Sea Turtle Partnership, unpublished data, 2019; Formia *et al.* in prep) compared to those described in Witt *et al.* (2009).

Figure 22. Nesting sites of the SE Atlantic DPS. Size of circle represents the index of nesting female abundance. An "X" indicates that nesting was documented but not quantified.



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 (Figure 23, 24). In fact, because of the much larger size of this DPS when compared to SW Atlantic DPS, the vast majority of individual leatherback turtles found in the western South Atlantic along the coast of South America, and on the high seas, are expected to be from the SE Atlantic DPS. As described in the section on the SW 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) originate from western African beaches.

Figure 23. Satellite tracks of post-nesting females tagged at nesting beaches in western Africa.
Image: Figure 1 from Witt *et al.* (2011), Proceedings of the Royal Society B, *https://royalsocietypublishing.org/doi/full/10.1098/rspb.2010.2467*

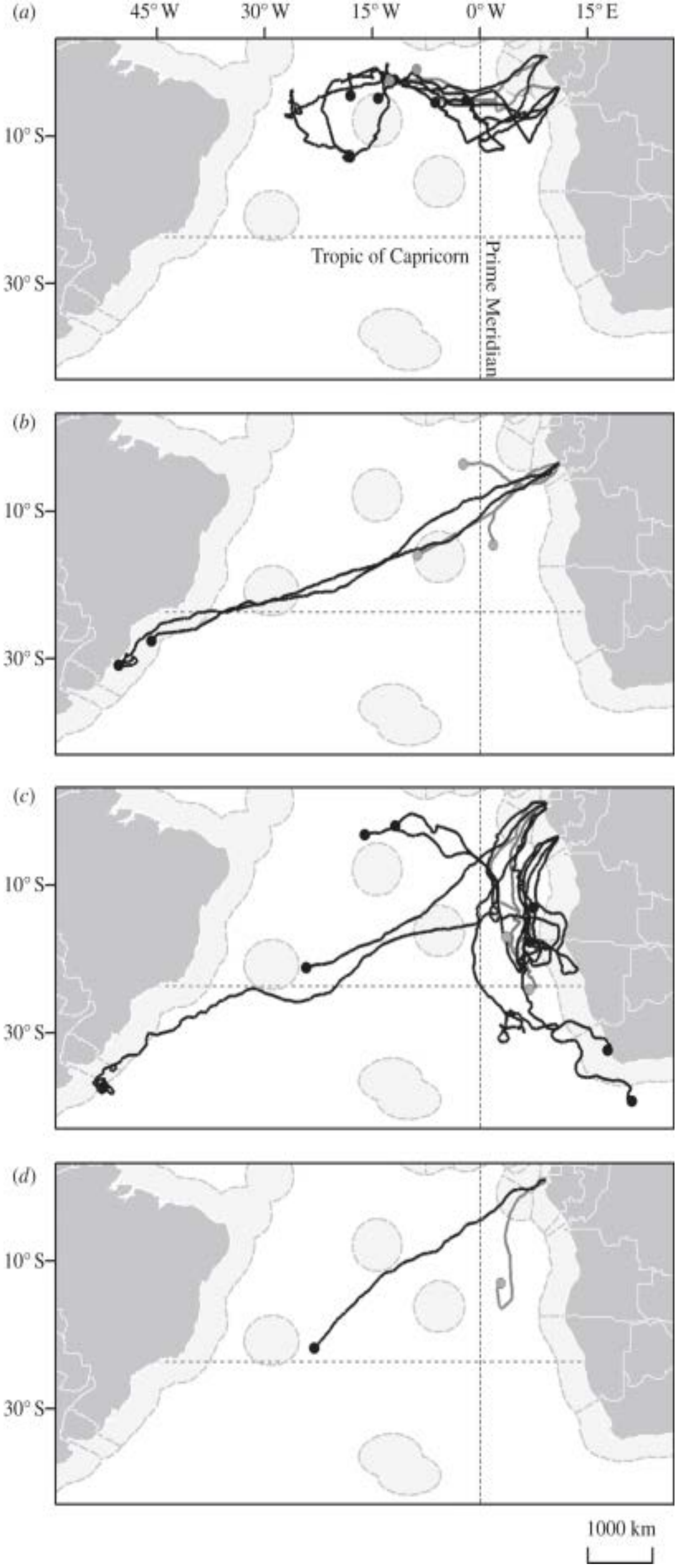
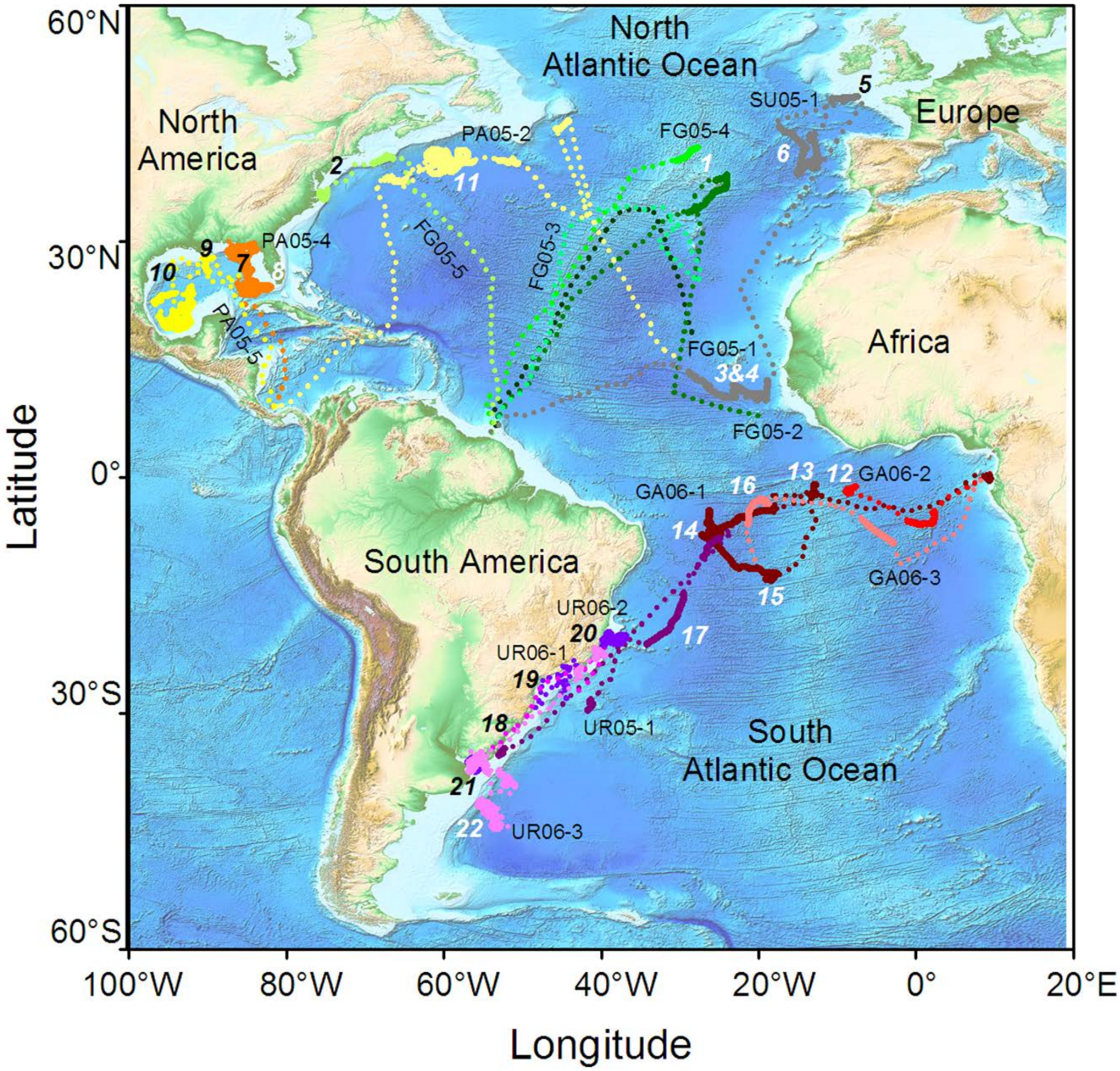


Figure 24. Satellite tracks of leatherback turtles throughout the Atlantic Ocean. Tracks #12-22 show individuals from the SW and SE Atlantic DPSs and their overlapping range. Image: Figure 1 from Fossette *et al.* (2010a), PLOS One, *https://doi.org/10.1371/journal.pone.0013908*.



7.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the SE Atlantic DPS.

7.1.1 Abundance

We estimated the total index of nesting female abundance for the SE Atlantic DPS to be 9,198 females. We based this total index of nesting female abundance on the data summarized in Table 10 and 12 and explained in detail below. We used the best available data to calculate the total index of nesting female abundance for the DPS. In Gabon, we used data from annual aerial surveys of the nesting beaches along with an estimate of 95 percent (Formia *et al.* in prep) of emergences resulting in nesting in order to determine an estimated nest count. To estimate the index of nesting female abundance in Gabon, we used a remigration interval of three years and a clutch frequency of 7.8 clutches per season per female based on nesting beach surveys (Casale *et al.* in prep; Formia *et al.* in prep.). We estimated the index of nesting female abundance in Gabon to be 8,495 females. For the remainder of the DPS, where site-specific data were not available, we used a remigration interval of 3 years and a clutch frequency estimate of 5 clutches per year. Totaling all nesting females, we found the total index of nesting female abundance to be 9,198 females for the SE Atlantic DPS. Despite covering a vast area, overall sea turtle research and conservation projects in this region are relatively recent. Nesting beach monitoring varies greatly, and in most cases is limited and incomplete. The existence of unknown or unmonitored nesting beaches is also likely.

Table 10. Available nesting data for the SE Atlantic DPS. Number of nests (or other units, as identified) recorded for the first and last monitored at surveyed nesting beaches. We also include the highest and lowest number of recorded nests (or other units, as identified). We calculated the index of nesting female abundance by summing the number of nests over the most recent remigration interval (i.e., 3 years; see Table 12) divided by the clutch frequency (5 clutches per season for all except Gabon, where data indicates 7.8 per season; see Table 12). We also provide the index of nesting female abundance for each nation when available data met our criteria.

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Senegal				Unquantified
Sporadic nesting reported in the past, but none observed in recent years.	N/A	N/A	Maigret 1978; Dupuy 1986; Tomas Diagne, pers. comm.	
The Gambia				Unquantified
Nesting reported but very rare.	N/A	N/A	Barnett *et al.* 2004; Hawkes *et al.* 2006	

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Guinea				Unquantified
Nesting has yet to be documented but is likely given nesting in neighboring areas.	N/A	N/A	Fretey 2001	
Guinea-Bissau				Unquantified
Orango National Park, Bijagos Archipelago, 52 km	Only track data, not nests. Extremely limited nesting activity. 5 (2012) 0 (2014)	High: 5 (2012) Low: 0 (2014)	IBAP – Instituto da Biodiversidade e das Áreas Protegidas	
Sierra Leone				39
Varied beaches	20 (2008/2009) 70 (2017/2018)	High: 80 (2012/2013) Low: 12 (2010/2011)	Aruna and Tiwari in prep.	39
Liberia				45
Little Bassa, 22 km Rivercess County, 18 km	6 (2011/2012) (note: only one of two beaches monitored in 2011/2012) 82 (2017/2018)	High: 95 (2014/2015) Low: 6 (2011/2012- with only one beach monitored) Low: 60 (2016/2017- with both beaches monitored)	Saykpa *et al.* in prep	45

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Ivory Coast (Cote d' Ivoire)				40
Kablake Village to Roc-Dawa, 30km	70 (2011/2012) 90 (2017/2018)	High: 102 (2013/2014) Low: 43 (2016/2017)	(Dah and Gomez unpublished data)	40
Ghana				4
Mankoadze, 3.4 km	45 (2012/2013) 6 (2016/2017)	High: 45 (2012/2013) Low: 6 (both 2015/2016 and 2016/2017)	Agyekumhene, unpublished data	4
Ada, 5 km	2000-2017	Average: 34 nests/year (no other data available)	Agyeman, unpublished data	Unquantified
Keta Beach, 20 km	N/A	Average: 80 nests/year (no other data available)	Fuseini, unpublished data	Unquantified
Togo and Benin				Unquantified
Total of 177 km of coastline	27 (2002/2003)	Only one recorded year.	Segniagbeto 2004	

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Nigeria				Unquantified
Nesting has been noted.	N/A	N/A	Fretey 2001; Mojisola *et al.* 2015	
Cameroon				3
Varied surveys across: Ipeyendje, Mbendji, Eboundja, Bekolobe, Mpalla, Likodo, Lolabe, Ebodjé, Mbendji	9 (2013/2014) 10 (2017/2018)	High: 10 (2017/2018) Low: 1 (2015/2016)	Nesting data provided by J. Fretey and A.H. Nibam.	3
Sao Tome and Principe				46
Sao Tome- Nesting around island on various small named beaches: Tamarindos/Micolo, Santana, Inhame, Jale, Praia Grande (ST), *Io Grande, Celeste, Muteca, Micondo, Pomba, Crija, Xixi, Sete Ondas, Plano de Aqua Ize* (italicized names only have data from 2016/2017 and 2017/2018)	16 (2015/2016) 118 (2017/2018)	High: 118 (2017/2018) Low: 16 (2015/2016- no data from multiple beaches) Low: 50 (2016/2017- lowest total year when all beaches have available data)	São Tomé: Programa Tatô	37
Principe- Nesting around island on various small named and unnamed beaches totaling 7.5 km: Grande, Boi, Macaco, uba, Ribeira Ize, Sundy, Micoto, Montanha, Marmita, Bumbo, Infante, Seca, Cabinda, and unnamed beaches.	22 (2015/2016) 8 (2017/2018)	High: 22 (2015/2016) Low: 8 (2017/2018)	ProTeTuga 2018, unpublished data	9

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
	Note: Sporadic data is available for a few beaches as far back as 1998/19199, but consistent data is not available for all beaches until the 2015/2016 season.			
Equatorial Guinea				457
Bioko Island: Gran Caldera Scientific Reserve (Beaches A-E)	4960 (2000/2001) 1500 (2017/2018) (Note: only body pits, not actual nests were available for Bioko, so this is likely a small overestimate)	High: 5662 (2001/2002) Low: 230 (2008/2009)	2000 – 2014: Honarvar *et al.* 2016 2015 – 2018: Shaya Honovar and Elizabeth Sinclair; Bioko Marine Turtle Program and Purdue University; Dana Venditti, Bryan Featherstone and Mary Katherine Gonder	457

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Gabon				8,495
Gabon coastline, 600 km. Note: Aerial and ground survey data provided was for emergences, not nests. Used 95% of emergences result in nesting, per A. Formia, to convert to nests.	108,588 (2002/2003) 24,093 (2015/2016) Note: No available data for 2003/2004 and 2004/2005.	High: 108,588 (2002/2003) Low: 4,275 (2009/2010)	Gabon Sea Turtle Partnership; Formia *et al.* in prep.	8,495
Republic of Congo				69
Numerous beaches along the coast.	70 (2003/2004) 149 (2016/2017)	High: 360 (2012/2013) Low: 45 (2008/2009)	Breheret, SWOT data sheet.	69
Democratic Republic of Congo				Unquantified
Nesting has been noted.	N/A	N/A	OCPE-ONG 2006	
Angola				Unquantified
Palmeirinhas	27 (2003/2004) 4 (2005/2006)	High: 27 (2003/2004) Low: 4 (2005/2006)	Weir *et al.* 2007	Unquantified

Table 11. The number of nesting sites by index of nesting female abundance. For this DPS we used each nation as a “nesting site” because of the lack of finer scale divisions of nesting beaches. We estimated the total index of nesting female abundance (for the DPS) by summing the indices of nesting female abundance from Table 10. We calculated the percentage at the largest nesting site by dividing that site’s index of nesting female abundance (Table 10) by the total index of nesting female abundance for the DPS.

Index of nesting female abundance	**Number of nesting sites**
Unquantified	9
1 - 10	2
11 - 50	4
51 - 100	1
101 - 500	1
501 - 1,000	
1,001 - 5,000	
5,001 - 10,000	1
10,001 - 20,000	
>20,000	
Total number of sites	18
Total index of nesting female abundance (DPS)	9,198
Confidence in total index of nesting female abundance	Moderate (much of coastline is not regularly monitored, but Gabon captures the vast majority of nesting and Gabon nesting numbers are uncertain)
Largest nesting site, percentage of total index	Gabon, 90+ percent

Our index of 9,198 nesting females may be an underestimate 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.* 2013b), which are represented in our index. Below, we discuss the best available data on abundance from each nation.

According to Witt *et al.* (2009), Gabon once hosted the largest single leatherback turtle nesting aggregation in the world, with an estimated 36,185 to 126,480 clutches per year, and a total estimate of 15,730 to 41,373 nesting females, analyzing aerial survey data between 2002 and

2007. These estimates were based on a combination of aerial surveys, along the entire 600 km stretch of available nesting beach, and ground-truthing surveys, conducted during the 2002/2003, 2005/2006, and 2006/2007 nesting seasons. More recent aerial surveys of the Gabon nesting beaches (Formia *et al.* in prep) indicate a likely overall steep decline in nesting from the levels in the early 2000's, with a high of 108,588 estimated nests in 2002/2003, and a low of 4,275 estimated nests in 2009/2010, and fewer than 25,000 nests in the final year of available data (2015/2016). Based on 2013/2014–2015/2016 (the last 3 years of data available), Gabon has an estimated 8,495 nesting females. Bioko Island, off Equatorial Guinea, represents an important nesting site for the SE Atlantic DPS. There is only scattered nesting on continental Equatorial Guinea (Fretey 2001). While nesting occurs in other parts of Bioko Island, the only recent available data are from the Gran Caldera Scientific Reserve on Bioko Island. Monitoring at the reserve is split into 5 segments, "Beaches A–E," but we present the combined data for the Reserve. Turtles likely nest across the beaches during one season. There is some variation in the months monitored by year and beach, but monitoring is primarily October or November through March or April. Published nesting data were available from 2000/2001 through 2014/2015 (Honarvar *et al.* 2016). Nesting data for southern beaches from 2015/2016 through 2017/2018 were provided by Shaya Honovar and Elizabeth Sinclair (Bioko Marine Turtle Program and Purdue University) and Dana Venditti, Bryan Featherstone, and Mary Katherine Gonder (Drexel University). The data provided body pits, not actual nests. While our estimate may be slightly high for the monitored beaches, nesting occurs in other areas of Equatorial Guinea for which we have no data. Thus, our overall estimate is likely an underestimate. We estimate 457 females nesting on the Bioko Island Reserve. Older data, from 2000/2001 to 2004/2005, found an average of 3,896 nests annually (Rader *et al.* 2006), which would result in an estimate of 2,338 nesting females (i.e., [3,896 nests/year x 3 year remigration interval]/5 nests/year). The likely decline in this population is further discussed in Section 7.1.2 Productivity.

For the Republic of Congo, we have data from 2003/2004 to 2016/2017. The monitoring activities covered numerous beaches along the coast, with varying duration, from late September to early November and typically ending in early April. The data provided (SWOT datasheet provided by N. Breheret) do not indicate when beaches were surveyed; they only showed when a turtle nest was found (i.e., only positive results were reported), so it is unclear whether the variation in beaches recorded for each year reflects scattered nesting or differential effort and inconsistent surveying. Based on the data available, we estimated 69 females nesting in the Republic of Congo. In an analysis of older data (1999 to 2008), Girard *et al.* (2016) estimated 933 nests per year on the monitored beaches, which would result in an estimate of 560 nesting females in the population ([933 nests/year x 3 year remigration interval]/5 nests/year). It is unclear if the large difference is related to a decline in nesting, differences in the surveys, or both.

Leatherback turtles nest in small numbers on mainland Guinea-Bissau with only a handful of nests reported in a season (Agardy 1993; Barbosa *et al.* 1998; Fretey *et al.* 2007). Only one beach is monitored regularly, in Orango National Park in the Bijagos Archipelago. Nesting activity is minimal in that area, with only occasional leatherback nesting tracks recorded. Nesting may occur on other beaches in the archipelago but is expected to be similarly rare.

Further south in Sierra Leone, the number of beaches monitored varied over time from 11 to 28. Surveys are primarily conducted from October to April. The number of monitoring days in Sierra

Leone was consistent except for the period 2010/2011 when additional funds were obtained to monitor during the rainy season (from May to August), resulting in 4 additional nests recorded (Aruna and Tiwari, in prep.). We estimate about 39 nesting females in Sierra Leone.

In Liberia, leatherback nesting is recorded along 22 km of beach at Little Bassa, and 18 km of beach in Rivercess County. The data provided does not separate out the counts for the two areas. Data for 2011/2012 came from one beach, and had only 6 nests. Nesting counts from 2012/2013 to 2017/2018 covered two beaches and ranged from 60 to 95 nests (Saykpa *et al.* in prep). We estimate about 45 nesting females in Liberia.

For the Ivory Coast, available data are from 30 km of beach between Kablake Village and Roc-Dawa. Data indicate that the leatherback turtles prefer the 10 km portion of the beach between the rivers Dodo and Gnegbagbo, where about 70 percent of nests are found (Dah and Gomez unpublished data). Additional studies involving single years of nesting counted 218 nests over 41 km of beach in February 2001 (Gomez 2005) and 189 nests reported from non-exhaustive surveys of 27 km of coastline during the 2001 to 2002 nesting season (Penate *et al.* 2007). We estimate about 40 nesting females in the Ivory Coast.

In Ghana, leatherback turtles appear to be the second most common nesting sea turtle species (Amiteye 2000; Agyekumhene 2017). The primary nesting beach monitoring is divided among 3 beaches: Mankoadze, Ada, and Keta Beach. We obtained information on nest averages but did not receive individual year data to estimate abundance for Ghana outside of Mankoadze, which has the lowest average of the three monitoring areas. Mankoadze averaged 17 nests annually over 3.4 km of beach from 2012 to 2017 (Senior Manager: Andy Agyekumhene). Ada averaged 34 nests annually from 2000 to 2017 over 5 km of beach (Senior Manager: Dickson Agyeman). Keta Beach averaged 80 nests over an unspecified time frame across its 20km stretch (Senior Manager: Abdul-Kareem Fuseini). A separate time series was provided for Mankoadze from the 2012/2013 to 2016/2017 nesting season. The data for Mankoadze is too short to determine a trend but showed a steep drop from 41 nests (2012/2013) to 22 (2013/2014), then down to 5 (2014/2015), 5 (2015/2016) and 6 (2016/2017). In a 1-year survey between August 2006 and March 2007, 481 leatherback nests were counted along a 7 km stretch near Ada (Agyekumhene *et al*. 2010), which is far greater than the average from 2012 to 2017, possibly indicating a decline or shift in nesting. We estimated 4 nesting females for the Mankoadze stretch of beach but did not have sufficient data to estimate for the rest of Ghana.

In Cameroon, nesting is minimal, and scattered nesting across various beaches such as Ipeyendje, Mbendji, Eboundja, Bekolobe, Mpalla, Likodo, Lolabe, Ebodjé, and Mbendji. Monitoring at those beaches had differing effort and came from different projects over the years (unpublished data provided by J. Fretey and A.H. Nibam). From 2014/2015 to 2016/2017 only one or two nests were laid per year, while in 2017/2018 there were 10 nests. This may be a result of varied survey effort, or turtles nesting in Cameroon may also nest in neighboring nations. Girard *et al.* (2016) estimated on average 43 leatherback nests annually (95 percent CI: 15 to 177), which would calculate to 26 nesting females (43 per year x estimated 3 year remigration interval, divided by estimated 5 nests/season). However, Girard *et al.*'s (2016) dataset included surveys from 1999 to 2008, and is thus less recent than the data we obtained from J. Fretey and A.H. Nibam (2018 unpublished data), therefore the nesting may have declined or nesting shifted to other locations. We estimated 3 nesting females in Cameroon based on the dataset we were provided.

The nation of São Tomé and Principe has small amounts of nesting across numerous small beaches on the islands: On São Tomé nesting is monitored across numerous small beaches, and we received data from 2015/2016 through 2017/2018 for Tamarindos/Micolo, Santana, Inhame, Jale, and Praia Grande, and 2016/2017 through 2017/2018 for Io Grande, Celeste, Muteca, Micondo, Pomba, Crija, Xixi, Sete Ondas, and Plano de Aqua Ize. We estimate 37 nesting females in São Tomé but, given the small size of the island and the proximity to other mainland nesting beaches, it is likely that some individuals nest on São Tomé as well as the mainland. Girard *et al.* (2016) had estimated 78 nests annually on São Tomé based on a more complete, but older, dataset (1999 to 2008). On Principe, monitoring has occurred sporadically across numerous beaches (Grande, Boi, Macaco, uba, Ribeira Ize, Sundy, Micoto, Montanha, Marmita, Bumbo, Infante, Seca, Cabinda, and unnamed beaches). Data is inconsistent, with no data for any of the beaches from 2000/2001 through 2002/2003, and 2007/2008 through 2013/2014. Most beaches have no data until the 2014/2015 nesting season, with Infante and Seca still unmonitored that season. Data are available for all beaches starting in 2015/2016 through 2017/2018 (ProTeTuga 2018, unpublished data). Based on the last 3 years of data, we estimate Principe to have 9 nesting females. Therefore the country total for Sao Tome and Principe is estimated at 46 nesting females.

Angola represents the likely southern limit of nesting for the DPS. Surveys of the Angolan coast indicate that leatherback nesting occurs on the northern and central beaches (Carr and Carr 1991; Weir *et al.* 2007). There is very limited data available, and what is available is not recent. For one beach in Angola, Palmeirinhas, we used data presented in Weir *et al.* (2007), covering the 2003/2004, 2004/2005, and 2005/2006 nesting seasons. Based on that data, we estimated eight nesting females; however, it is important to note that nesting declined from a high of 27 nests in 2003/2004 to nine nests the following season, and only four in the final year of available data. If the number of nests stayed the same or continued to decline compared to the 2005/2006 season, the estimate for this beach would decline to one or two turtles.

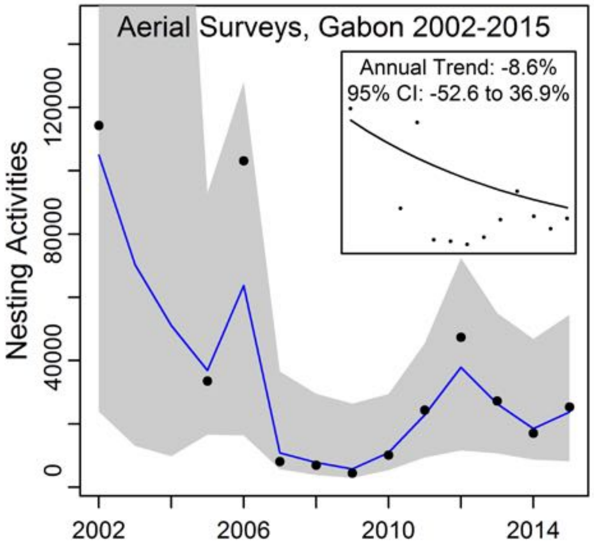
A handful of other nations along the Atlantic coast of Africa are also known, anecdotally or through occasional reporting, to have sporadic leatherback turtle nesting. Sporadic leatherback nesting has been reported from Senegal in the past (Maigret 1978; Dupuy 1986), however no nesting has been observed in recent years (Tomas Diagne, African Aquatic Conservation Fund, pers. comm., 2019). In the Republic of The Gambia, leatherback nesting appears to be a rare occurrence (Barnett *et al.* 2004; Hawkes *et al.* 2006). Nesting has yet to be documented in Guinea (Fretey 2001). Along the coastline of both Togo and Benin, which extends over 177 km, nesting activity is low and sporadic, with 27 nests observed during the 2002/2003 season on the beaches of Togo (Segniagbeto 2004). Leatherback turtles nest in Nigeria, but nesting numbers are not quantified (Fretey 2001; Mojisola *et al.* 2015). On the short coastline of the Democratic Republic of Congo nesting by leatherback turtles has been indicated (OCPE-ONG 2006).

The total index of nesting female abundance is not at a level that reduces risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017a). We do not have historical abundance estimates with which to provide context on the extent nesting prior to the 2000s. We conclude that the total index of nesting female abundance of 9,198 does not reduce the extinction risk of this DPS.

7.1.2 Productivity

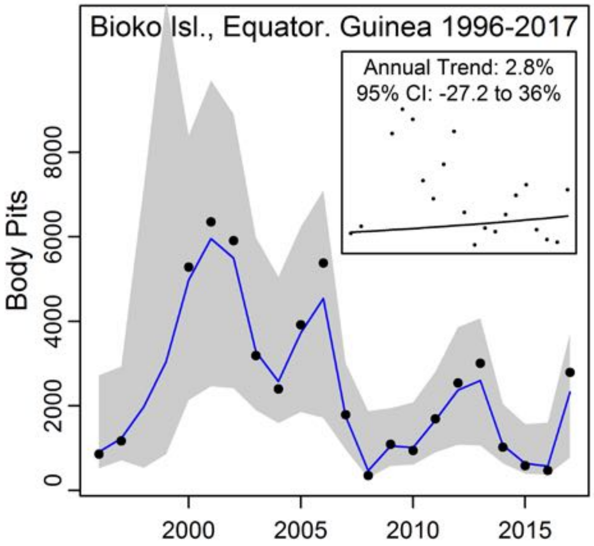
The SE Atlantic DPS demonstrates a declining nest trend at the largest nesting aggregation. We performed a trend analysis on the 12 years of aerial survey data covering the nesting beaches along the entire coast of Gabon (Figure 27). As with all DPSs, we report the BSSM trend analysis results as the median and CI, which reflects that there is a 95 percent chance that the trend falls between the low and high CI values. The wider the CI, the less confident we are in the estimated median trend. The higher the "f statistic" the more confident we are in the sign (positive or negative) of the estimated median trend. Data collected from 2002/2003 through 2015/2016 (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).

Figure 25. Nesting activity trend from aerial surveys of nesting beaches in Gabon. The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (nesting activities). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the inter-annual variability.



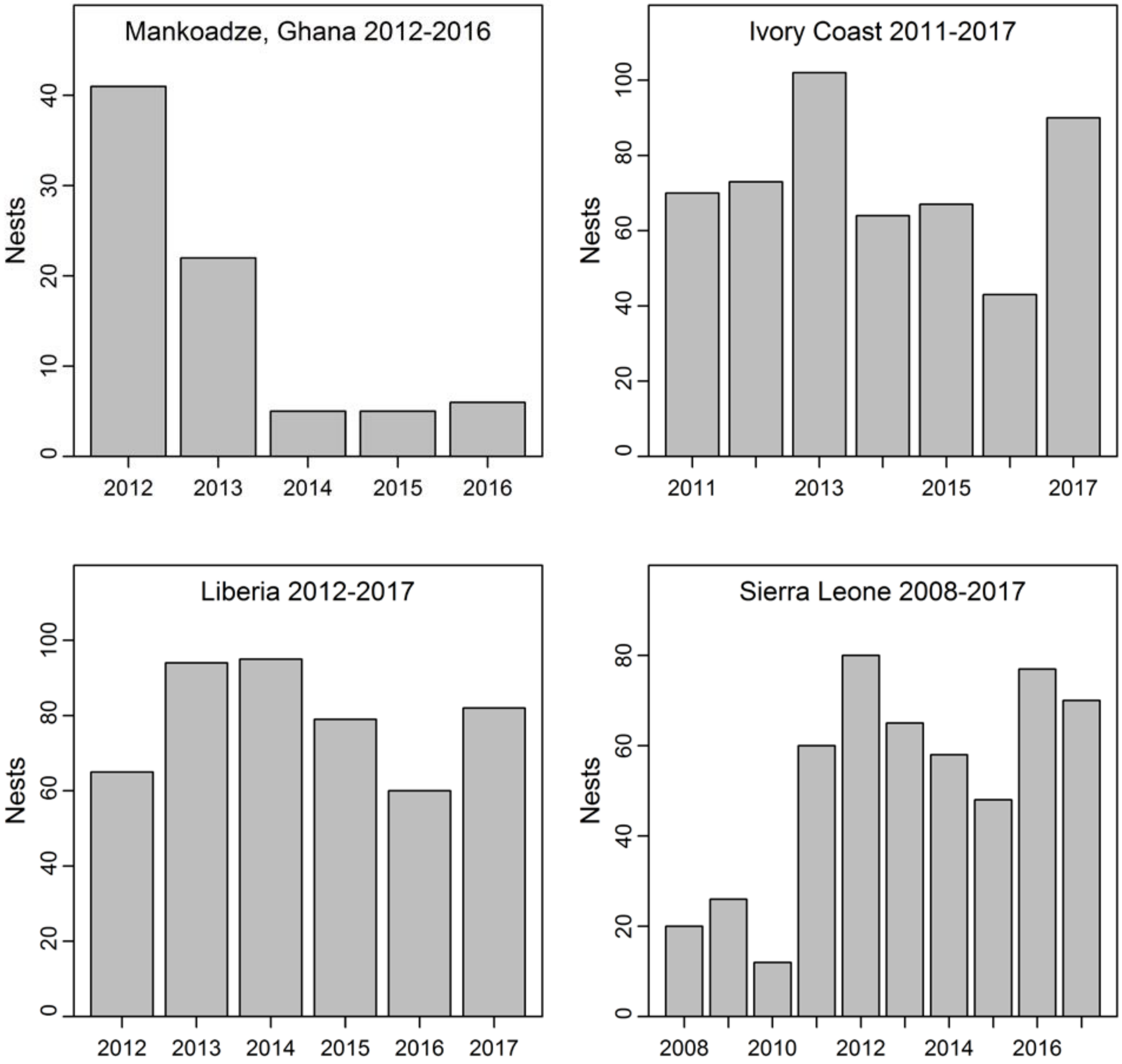
The best dataset available from the smaller nesting aggregations outside of Gabon are from Equatorial Guinea (Figure 26); though it hosts thousands of nests, this nesting aggregation is a fraction of the size of the one in Gabon. On Bioko Island, the number of body pits were monitored in the Gran Caldera Scientific Reserve (Beaches A−E) from 1996/1997 through 2017/2018. The number of body pits increased 2.8 percent annually (sd = 15.6 percent; 95 percent CI = −27.2 to 36.0 percent; f = 0.576; mean annual body pits = 2,538).

Figure 26. Body pit trend at Gran Caldera Scientific Reserve on Bioko, Equatorial Guinea. The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (body pits). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the inter-annual variability.



Because we did not have sufficient data for trend analyses, we provide bar graphs of the data from Ghana, Ivory Coast, and Liberia (Figure 27). For Ghana, the nest counts from Mankoadze beach (2012/2013–2016/2017) appear to have decreased over time, but the nest counts are low and could simply reflect natural interannual variability. For the Ivory Coast, there is no apparent pattern in annual nest counts. For Liberia, there is also no apparent trend in nest counts monitored on beaches in Little Bassa and Rivercress County. For Sierra Leone, it appears there could possibly be an increase in the number of nests observed from 2008 to 2017, but the overall numbers are low (12–80 nests/yr) so the variability observed could be natural or within the limits of the data observation errors.

Figure 27. Bar graphs of nests at beaches of Ghana, Ivory Coast, Liberia, and Sierra Leone.



Because the Gabon aerial data were not available at the time, the IUCN Red List assessment concluded that a trend could not be evaluated for the SE Atlantic Ocean subpopulation (Tiwari *et al.* 2013b). Girard *et al.* (2016) reviewed the status of leatherback turtles between 1998 and 2008 for Central Africa and concluded that the Gabon had a decreasing trend, whereas Congo showed an increasing trend with an overall stable nest trend in the Central African region over 8 years.

There is also a high degree of uncertainty regarding the productivity parameters for this DPS (Table 12). Where information is available, it is often from a limited area that may not be representative of the entire DPS. Data for size of nesting females, clutch size, hatching success, and incubation period appear to be similar to that found in other DPSs.

Table 12. Productivity parameters for the SE Atlantic DPS.

Productivity metric	**Variable by nation**	**Reference**
Size of nesting female (CCL cm)	Gabon (Kingere, Gamba, and Mayumba): Average between 150.4–150.9 in all three places sites	Billes 2000; Verhage *et al.* 2006; Ikaran 2010
	Angola (Palmeirinhas): Average 149.7	Weir *et al.* 2007
Remigration interval (years)	Gabon: 2.3	Casale *et al.* in prep
Clutch size (eggs)	Gabon (Kingere): 78.2 (26–123)	Ikaran 2010
	Gabon (Gamba): 65.9 ± 17.2	Verhage *et al.* 2006
	Gabon (Mayumba): 77.8 ± 20.4	Billes 2000
	Angola (Palmeirinhas): 67.5	Weir *et al.* 2007
Clutch frequency (nests per season)	Gabon: 7.8	Casale *et al.* in prep.
Incubation period (days)	Gabon (Kingere): 64.6 (57−72)	Ikaran 2010
Hatching success (percent)	Gabon (Kingere): 16 (including failed, inundated nests; 62 percent of nests did not produce any live hatchlings over 3 years; 46.3 percent success rate among surviving nests)	Ikaran 2010
	Gabon (Mayumba): 67.3 (1999/2000) and 71.0 (2008/2009) of hatching success from surviving nests (no data provided on percent of nests surviving).	Billes 2000; Ikaran 2010 (appendix I)

Productivity metric	**Variable by nation**	**Reference**
	Gabon (Gamba): 67 (of surviving nests) (no data on percent of nests surviving)	Livingstone 2007
	Equatorial Guinea (Moraca, South Bioko Island): 57.85 ± 25.06 (with 100 percent nest survival)	Tomas 2010
Sex ratio	Gabon (Kingere): Based on incubation temperatures, hatchling sex ratios thought to be balanced overall with almost exclusively males early in the season, and mostly females late in the season.	Ikaran 2010

We conclude that the declining nest trend and productivity parameters contribute to the extinction risk of this DPS.

7.1.3 Spatial Distribution

The SE Atlantic DPS has a broad spatial distribution. The nesting range is centered around Gabon, but with additional nesting aggregations stretching along the coast of Africa from Senegal to Angola. Genetic data available for Gabon and Ghana indicate significant genetic differentiation based on mitochondrial DNA data, but weak differentiation based on microsatellite analyses, likely indicating demographically independent subpopulations connected by limited gene flow (Dutton *et al.* 2013b). There is little available information on the inter-nesting movements of females, so we are unsure if females move between nesting aggregations within or between nesting seasons. 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 have shown that leatherback turtles from several important nesting populations in the Atlantic use the waters further south 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). Drifter data suggest that currents may carry small leatherbacks (under 80 cm CCL) from West Africa nesting beaches to the Equatorial central Atlantic (Lopez-Mendilaharsu *et al.* 2019).

The wide distribution of foraging areas likely buffers the DPS against local catastrophes or environmental changes that could limit prey availability. With the expansive nesting range, the DPS is also somewhat insulated 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 reduce the extinction risk of the SE Atlantic DPS.

7.1.4 Diversity

The SE Atlantic DPS demonstrates some diversity. Genetic analyses for this DPS are limited, but Dutton *et al.* (2013b) found moderate genetic diversity in samples from Gabon and Ghana, including four new haplotypes unique to females nesting in western Africa. Nesting occurs on continental and insular beaches. There are multiple foraging strategies, including pelagic and coastal, along either side of the Atlantic. The genetic diversity, along with multiple and diverse foraging sites (coastal and pelagic), and combination of insular and mainland nesting provide diversity and resilience that may reduce the extinction risk of this DPS.

7.2 ESA Section 4(a)(1) Factors

The following sections describe and assess the 4(a)(1) factors or threats. For each, we evaluate the best available information on the threat. We also describe, if possible, the exposure and impact of each threat.

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

For the SE Atlantic DPS, the present, or threatened, destruction, modification, or curtailment of the habitat or range is a major threat. Between current threats of nesting beach obstructions, human-induced erosion at the nesting beaches, and light pollution, along with the likely future expansion of threats from coastal construction and development in the region, there are significant concerns about the ability of the DPS to remain viable in the coming decades without major efforts to curtail this threat.

Physical changes impacting the suitability or accessibility of nesting beaches for leatherback turtles is an ongoing problem for the DPS. Logs that have broken loose from timber rafts of industrial logging operations often wash up on the beaches sometimes at densities of 247 logs/km in Gabon and have been demonstrated to block 30.5 percent of the beach in Pongara, Gabon, disrupting or aborting an estimated 2,111 nesting attempts (Laurance *et al.* 2008). Given Gabon's importance as the largest nesting ground in the DPS, by far, and the obstruction of such a significant percent of the nesting beaches in that nation, this threat is likely to have a large detrimental impact on the entire DPS. 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). We are not aware of any substantive efforts to curtail the problem since the publication of that study. In addition to blocking nesting attempts, several leatherback turtles have died as result of being trapped or wedged 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. In some cases, nesting females have been found dead after being trapped on the beach for an extended period of time. Logs appear to be a problem in Gabon, Equatorial Guinea and Cameroon (Formia *et al.* 2003).

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).

Along with coastal development, light pollution modifies nesting habitat, deterring nesting females and disorienting both hatchlings and nesting females. Bourgeois (2009) found that artificial lights disoriented leatherback hatchlings in Pongara National Park in Gabon: 27 of the 41 nests (66 percent) studied had hatchlings crawl 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) likewise concluded that light pollution from Libreville and the village of Pointe Denis is a major threat to nesting females and hatchlings that get disoriented and lost in the surrounding savannah.

More generally, urbanization and coastal development are rapidly growing threats at some nesting beaches (Girard and Honarvar 2017). Until now, coastal construction has been limited in Gabon. Some of the nesting beaches are protected as parks and others due to their remoteness. However, according to Ikaran (2010) there is a high potential for coastal development in Gabon. Though for now, there are uninterrupted wild beaches a few km south of Pointe Denis, an important and growing tourist and development area. Along with direct habitat loss from coastal development and urbanization, impacts from pollution and litter would be expected to increase, as discussed in 7.2.5.2 below.

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 the most important nesting beach in Gabon (Mayumba National Park) that stretches to the outer limits of the EEZ.

We conclude that 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 important individuals. Based on the information presented above, we conclude that this is a major and increasing threat to the DPS.

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

Throughout much of its nesting range, sea turtles are under intense pressure from the poaching of eggs and turtles for various uses. As a result, poaching is a threat to the SE Atlantic DPS. In addition to consumption for food, leatherback turtles are an important component in some traditional medicine, and religious practices.

Along the primary nesting beaches of Gabon, poaching is not considered to be a significant threat. Poaching only occurs on a small scale because 78 percent of nests occur within national parks, and human population density along the coast is extremely low compared to many nations worldwide (A. Formia pers. comm., 2018). However, elsewhere in the region, the level of poaching is high, or would return to high levels, if not limited by activities funded through the

USFWS' Marine Turtle Conservation Fund enacted under the Marine Turtle Conservation Act. These activities reduce poaching through increased project presence on the beaches, beach monitoring, hiring of locals for participation in the projects, and raising awareness and providing education to local communities (M. Tiwari, pers. comm., 2018).

Conflicting beliefs about sea turtles exist throughout the region. In some communities sea turtles are considered food provided by the gods, while in others they have been historically protected by indigenous taboos, often based on stories passed down by ancestors (Topka and Abule 2014; 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 Côte d'Ivoire 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; it is believed 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, and over a long duration, it also reduces recruitment and thus abundance. Given the moderate exposure but high impact, we conclude that the poaching of turtles and eggs poses a threat to the DPS.

7.2.3 Disease and Predation

Given the widespread nesting of SE Atlantic leatherback turtles along the western African coast, and the limited information on many of those nesting beaches, much remains unknown about the threat of disease and predation on this DPS. In some specific areas there appears to be high predation pressure, but it may not be widespread, as other areas are known to have little issue with predation.

Predation of leatherback eggs and/or hatchlings has been documented for a variety of predators, including: various ants (*Dorylus spininodis* and unidentified species*)*, ghost crabs (*Ocypode spp.*), monitor lizards (*Varanus niloticius*), crows (*Corvus albus*), mongoose *(Atilax paludinosus)*, porcupine (*Atherurus africanus*), domestic dogs (*Canis familiaris*), African civet cat (*Civettictis civetta and Viverra civetta*), and drills (*Mandrillus leucophaeus*) (summarized from Eckert *et al.* 2012). In Kingere, Gabon, high predation rates of eggs by crabs, lizards, mongooses, small cat species, and ants were noted by Ikaran (2010). The predation was found to contribute to a low hatching success rate of only 16 percent of eggs, and 62 percent of the nests did not produce any live hatchlings. However, it is important to note that the study attributed the low hatching success to a combination of high predation rates and beach erosion and flooding

(Ikaran 2010). The impact of predation alone could not be teased out from the information available. Along with crabs, ants (*Dorylus spininodis*) were the primary predator among the studied nests, but the magnitude of predation did not appear to be as high at other beaches in Gabon (Ikaran 2010).

As is common for all sea turtle species, 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 Zleeney 2011).

Information on diseases among leatherback turtles originating in the SE Atlantic is minimal, but an analysis of samples from nesting females from Gabon indicated normal blood chemistry parameters (Deem *et al.* 2006).

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 (reduced egg and hatching success). We conclude that predation poses a threat to the SE Atlantic DPS.

7.2.4 Inadequacy of Regulatory Mechanisms

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

While much of the most important nesting beach habitat (and turtles utilizing that habitat) in Gabon is protected because of inclusion in parks as well as remoteness from civilization, encroaching development and associated impacts, as discussed previously, may be an increasing threat. Many nesting beaches in other nations throughout the region are not as well protected.

The harvest of turtles and eggs is illegal throughout most of the nations from which the SE Atlantic DPS originates. For example, in Gabon, all sea turtle species are integrally protected by law (2011 decree 0164/PR/MEF). In some cases, these protective mechanisms are inadequate. While Congo does not have laws specifically protecting sea turtles, they are protected by wildlife laws that prohibit the hunting and collection of wildlife and their products, including eggs, between November 1 and April 31 annually. They are also protected in the Conkouati-Douli National Park; however, in areas without permanent 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 legally protected all sea turtles under Law 8/1988 and Decree 183/87 on fishing (Tomás *et al.* 2010). However, the poaching of eggs and sometimes nesting 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 it is not unusual for people to poach sea turtles or eggs on the beach (Tanner 2013). Throughout the region, as elsewhere in the world, enforcement is variable, but 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, including specific fishery laws such as Brazilian gear restrictions (see Section 6.2.4) or Nigeria's legal requirement to use turtle excluder devices in shrimp bottom trawls, many leatherback turtles die from incidental capture in fisheries throughout the range of the DPS.

Numerous regulatory mechanisms protect leatherback turtles, their eggs, and nesting habitat throughout the range of this DPS. Though the regulatory mechanisms provide some protection to the turtles, many are inadequate to sufficiently 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.

7.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

The SE Atlantic DPS faces other factors that affect its continuing existence, including fisheries bycatch, which is the primary threat to the DPS. Additional threats include: vessel strikes; pollution; and climate change.

7.2.5.1 Fisheries Bycatch

Fisheries bycatch is the primary threat to the SE Atlantic DPS. Threats come from both 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 Brazil, Uruguay, and Argentina, in the pelagic waters of the South Atlantic, and along the coastal waters off western Africa. Therefore, the information presented in the fisheries bycatch section for the SW Atlantic (see Section 6.2.5.1 Fisheries Bycatch) is applicable to this DPS and will not be repeated here. Below is information specific to fisheries along the coastal waters of Atlantic Africa.

There is little stranding information available along the coast of Africa. Off Gabon, there is limited observation effort; however, three of six stranded turtles displaying signs of fisheries or vessel-related interactions (Parnell *et al.* 2007). In Senegal, leatherback strandings are generally rare, but five were recorded in a recent survey (Mullié *et al.* 2015), although it is not clear if the strandings were fishery related.

One of the biggest threats to leatherback turtles in Atlantic waters is incidental capture in artisanal and commercial fisheries (Wallace *et al.* 2010b; Riskas and Tiwari 2013; Saykpa 2014). Lewison *et al.* (2004) estimated that 30,000 to 60,000 leatherback turtles were taken as 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^2, 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, and Africa still represents a significant gap in bycatch evaluation studies (Wallace *et al.* 2010b,

Exhibit 8 to Declaration of Adam Keats

2013). However, several studies have surmised that given the degree of fishing activity near nesting and foraging areas, sea turtle bycatch rates in the region are probably high (Lewison *et al.* 2004; Moore *et al.* 2010; Wallace *et al.* 2010b). 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 still largely incomplete (Kelleher 2005; Moore *et al.* 2010; Wallace *et al.* 2010b).

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 turtle estimates mostly include turtles from the SE Atlantic DPS, but tracking studies indicate that the turtles of the much smaller 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 and some purse seine effort and assigns a moderate ecological risk assessment score for the population originating from the SE Atlantic Ocean.

The limited bycatch data for waters of the Atlantic coast of Africa show that other fisheries interact with leatherback turtles, sometimes in large numbers depending on gear type. Between 2005 and 2015, annual captures of leatherback turtles reported as bycatch (live and dead) in artisanal fishing nets in Loango Bay in the Republic of Congo varied between 0 and 774 including recaptures, with the number of individuals varying between 0 and 628 (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). In addition, Gabon requires the use of TEDs in all shrimp trawlers (2015 Arreté 00026/MAEPSA/SG/DGPA). The accidental capture of 4 juvenile leatherback turtles (17 to 21 cm in carapace length) in March 1994 from the waters around São Tomé and Principe suggest that they may be important developmental habitats (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, with reported mortality rates of up to 31 percent. Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when nesting females are bycaught and killed. Therefore, we conclude that fisheries bycatch is a major, and the primary, threat to the SE Atlantic DPS.

7.2.5.2 Pollution

Pollution includes contaminants, marine debris, and ghost fishing gear. As with all leatherback turtles, entanglement in and ingestion of marine debris and plastics is a threat that likely kills several individuals a year. The SE Atlantic DPS faces the threat of pollution across its extensive range throughout the South Atlantic Ocean, from the Atlantic Africa to coastal South America. Much of the discussion in Section 6.2.5.4 Pollution for the SW Atlantic DPS applies to the SE Atlantic DPS as their ranges, and exposure to pollutants, overlap. Throughout Atlantic 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 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 (Fossette *et al.* 2014; López-Mendilaharsu *et al.* 2009).

The Gulf of Guinea has increasingly been the focus of extensive oil exploitation activities after the discovery of large oil reserves. Drilling activities by large oil corporations, with associated pollution and habitat destruction, are serious threats to major 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, main sources of pollution include industrial waste, raw/untreated sewage and pesticides. Hydrocarbon production contributes about 95 percent of the nation’s gross national product. Oil exploration, exploitation, and transportation have a significant effect on the environment. Crude and refined oil spills incidents are very 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. The area where frequent spillages occur is categorized as an ecologically sensitive or critical area, comprised of mangrove ecosystems (Adegbile Oyeronke 2013 presentation "National Report; Sea Turtles in Nigeria.).

The discussion in Section 6.2.5.4 Pollution for the SW Atlantic DPS includes specific information on industrial activity and pollutants along the South American coast, as well as evidence of plastics and other marine debris ingestion. That information applies to the SE Atlantic DPS as individuals from this DPS use those waters extensively and encounter all of the threats detailed in that section.

While it is clear that individuals from the SE Atlantic DPS have a high probability of encountering pollution across their range and throughout their lifecycle, we cannot quantify such impacts. However, given the prevalence of such pollutants we conclude that pollution is a threat to the DPS.

7.2.5.3 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 has the potential to negatively impact sea turtle nesting habitat. While little specific information is available on the effects of sea level rise on increasing erosion of leatherback turtle nesting beaches, there is evidence that it may be impacting SE Atlantic DPS nesting beaches. Erosion of important nesting beaches in Gabon may be at least partially attributable to sea level rise. From 1983 through the 2000's some areas have seen as much as 100 m of beach width lost, impacting availability of suitable nesting beach (Gabon Sea Turtle Partnership website: *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) compared to other sea turtle species.

In addition to the loss of nesting habitat, 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, she 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 the DPS, and the ability to nest in different locations along existing beaches, or on new beaches, is unclear. 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.

7.2.5.4 Vessel Strikes

There is little information regarding boat strikes for the SE Atlantic DPS. However, there is information to suggest that it is a potential, and possibly increasing, threat across at least a portion of its 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.

7.3 Extinction Risk Analysis

After reviewing the best available information, we conclude that the SE Atlantic DPS has a high extinction risk, as summarized in the following paragraphs. The index of nesting female abundance for the DPS 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. The DPS has a large spatial distribution. Nesting occurs along a very long stretch of coastline in Atlantic Africa, thus, the population is buffered from stochastic events that could otherwise have catastrophic effects on the entire DPS. There is metapopulation structure within this DPS, with fine-scale genetic differentiation between Gabon and Ghana. Genetic diversity also appears to be moderate. This DPS uses multiple, distant, and diverse foraging areas, providing some resilience against reduced prey availability or marine threats.

The SE Atlantic DPS faces several threats, as summarized in Table 16. 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 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 the densest nesting beaches in Gabon are protected because they occur in parks or are far from any towns. Many of the beaches outside of Gabon (e.g., Guinea-Bissau) have limited or no protection. Overutilization is highly varied, but quite extensive in some areas. Funding from the Marine Turtle Conservation Act has resulted in some reduction of the overutilization 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. While many laws are in place to protect sea turtles, the continued impacts of logging, poaching, and fisheries bycatch indicate that regulatory mechanisms are inadequate to completely address these threats. Additionally, many areas in the region have little or no enforcement of laws protecting turtles or nests on the beach.

Table 13. Threats to the SE Atlantic DPS. Exposure refers to the individuals affected by the threat. Impact refers to how the threat affects the demographic factor(s). The primary threat is identified with an asterisk.

Threat	**Exposure**	**Impact**
Destruction or modification of habitat	Nesting females, eggs, hatchlings	Loss of nesting females (abundance) and reduction of nesting and hatching success (productivity)
Overutilization	Eggs and nesting females	Loss of nesting females (abundance) and reproductive potential (productivity)
Predation and disease	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Inadequate regulatory mechanisms	Eggs and turtles of all life stages	Some laws are poorly enforced

Threat	**Exposure**	**Impact**
Fisheries bycatch*	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) and loss of nesting females (productivity)
Pollution	Eggs and turtles of all life stages	Lethal (abundance) and sublethal (productivity) effects
Climate change	Nesting females, eggs, hatchlings	Reduction of nesting and hatching success (productivity)
Vessel strikes	Females approaching nesting beaches and individuals in nearshore foraging areas	Loss of individuals (abundance) including loss of nesting females (productivity)

The DPS is relatively data-poor, reducing our ability to quantify threats for more than a small portion of the population. Based on the limited data available, the Team struggled with this decision, individually and as a Team. The Team voted as follows:

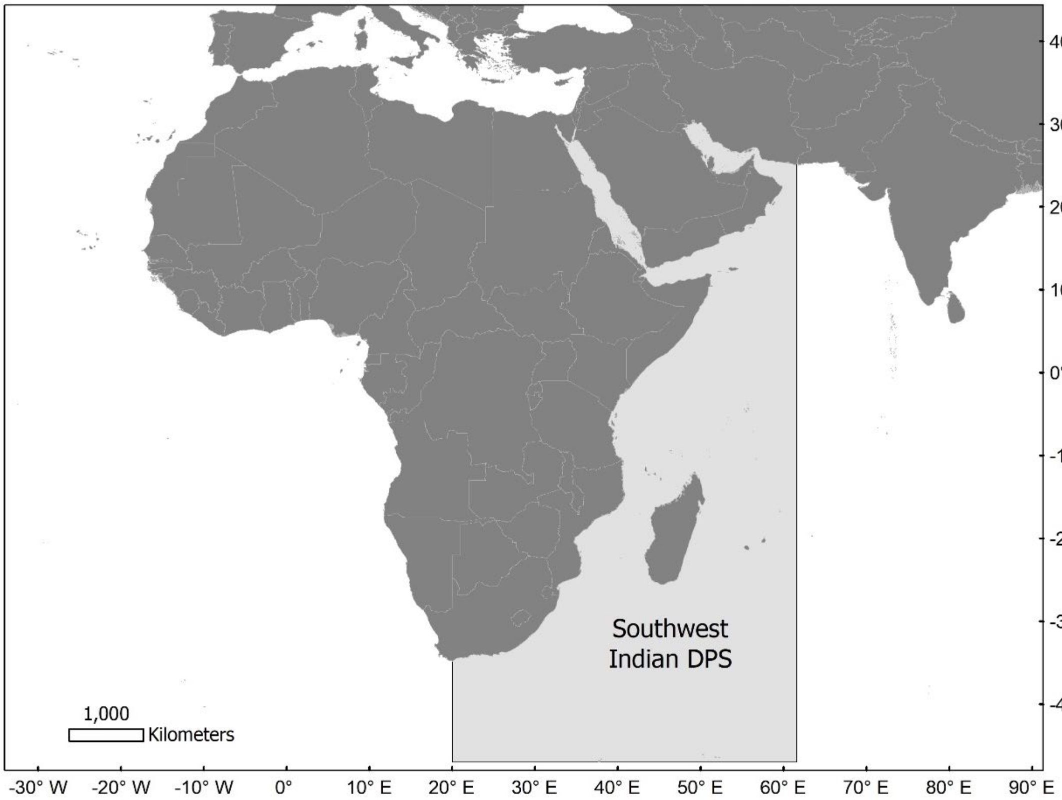
- 9 voted high extinction risk/moderate confidence due to threats and loss of abundance (not high confidence due to the lack of data on this DPS)
- 2 voted moderate extinction risk/low confidence due to the lack of data on this DPS

Therefore, the Team concludes that the SE Atlantic DPS meets the definition for high risk of extinction (see Extinction Risk Assessment) because the decreasing nest trend (8.6 percent annually since 2002) is at or near a level that places its continued persistence in question. Its index of 9,198 nesting females also reduces our confidence in its continued persistence. It faces clear and present threats that are likely to create imminent and substantial demographic risks (declining trends and reduced abundance). The moderate risk definition does not apply because it is at a high risk of extinction now (at present), rather than on a trajectory to become so in the foreseeable future. We have moderate confidence in this conclusion because though available data indicate reduced abundance, a declining trend, and severe threats, the DPS is relatively data-poor.

8.0 Southwest Indian DPS

We define 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 (Figure 28). The western boundary occurs at 20° E, the southern tip of Africa, and approximately where the Agulhas and Benguela Currents meet. The eastern boundary occurs at the border between Iran and Pakistan, where the Somali Current begins. The Antarctic Circumpolar Current defines the southern boundary of this DPS.

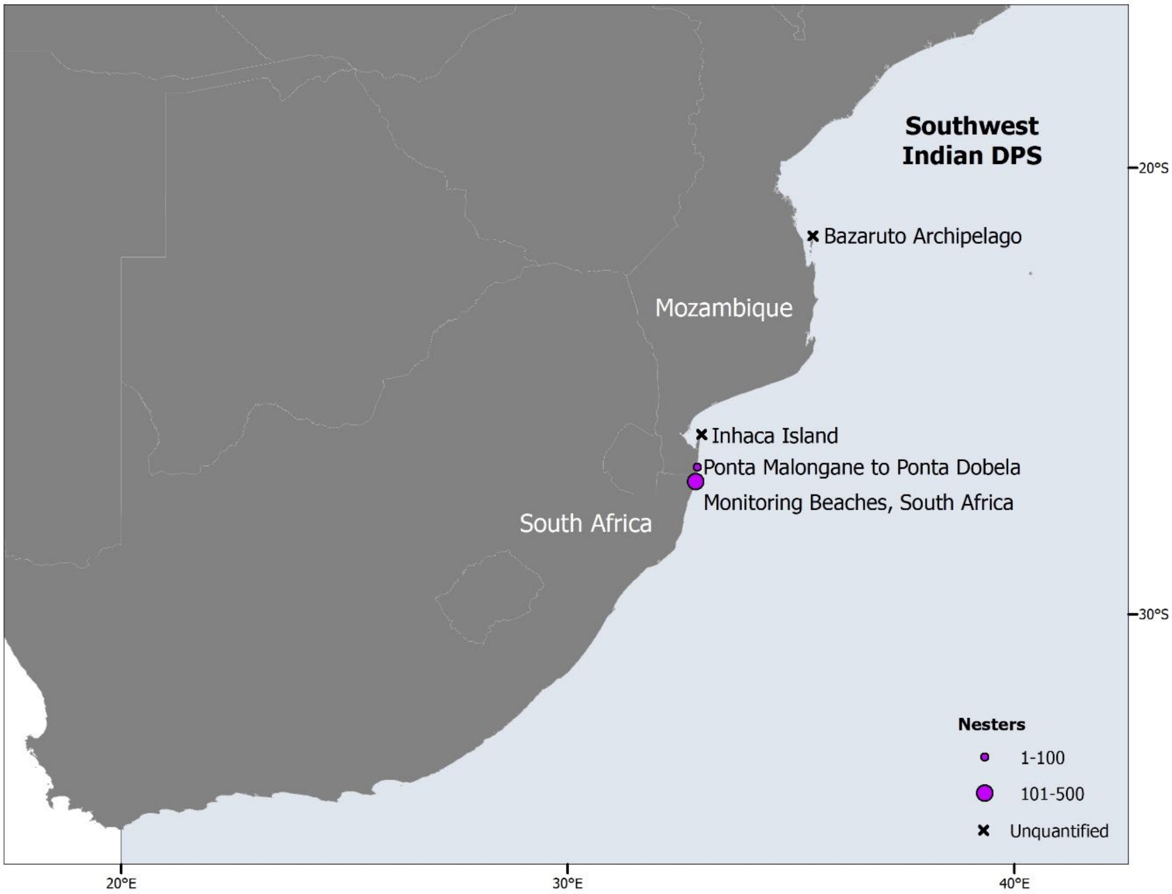
Figure 28. SW Indian DPS boundary map.



The range of the DPS (i.e., all 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 from this DPS 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 on beaches in South Africa and Mozambique (Figure 29). Nesting occurs 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). Sporadic nesting has been observed in other locations, including a single nesting event in Watamu, Kenya in January 2014 (unpublished data, Watamu Turtle Watch, Local Ocean Conservation 2019). In addition, there is occasional nesting on the southern coast of South Africa, outside of the monitoring area; however, these nests rarely hatch successfully (R. Nel, Nelson Mandela University, pers. comm., 2019).

Figure 29. Nesting sites of the SW Indian DPS. Size of circle represents the index of nesting female abundance. An "X" indicates that nesting was documented but not quantified.



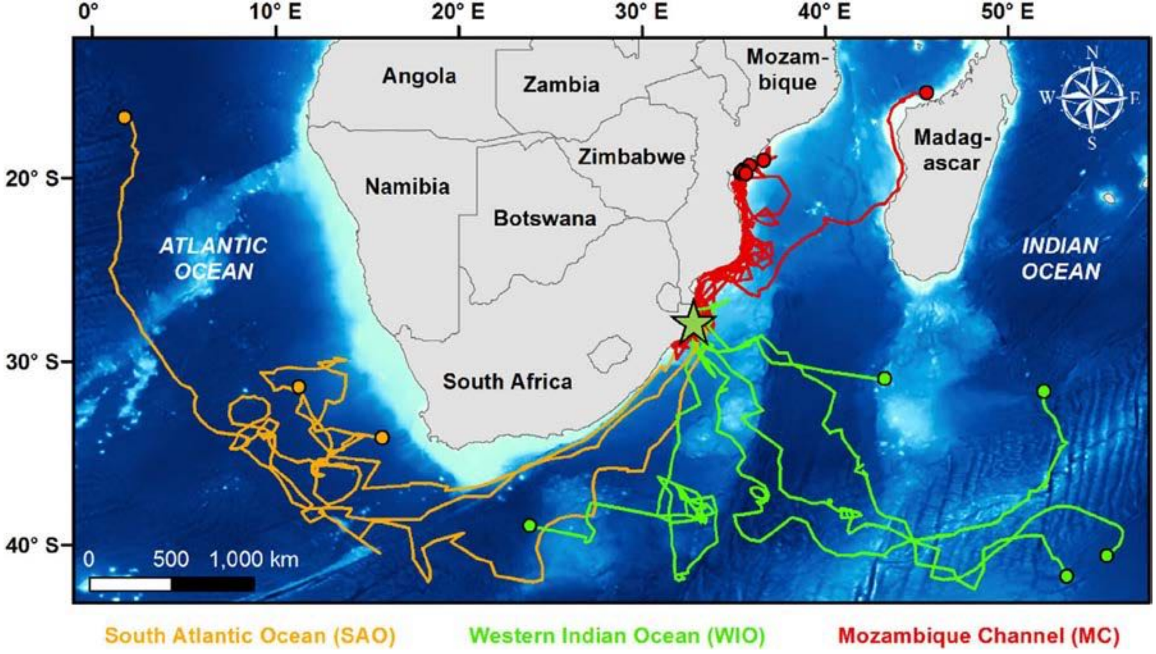
Nesting occurs on long (5 to 15 km), broad (50 to 100 m), silica sand beaches with little vegetation (Botha 2010; Nel *et al.* 2015; Robinson *et al.* 2017). The beaches are characterized by pristine, intact dunes that rise up to 100 m above sea level, interspersed with a few dynamic dunes and small, 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 and Ansorge 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 = 41) 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 tracking data of 11 post-nesting females tagged between 1996 and 2003 (Hughes *et al.* 1998; Luschi *et al.* 2003a; Sale *et al.* 2006). Robinson *et al.* (2016) satellite tracked 16 post-nesting females tagged between 2011 and 2013 (Figure 30). 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 (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.* (2003b 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 lead to highly productive foraging opportunities, with minimal energetic cost required to return to waters off nesting beaches.

Figure 30. Satellite tracks of 16 post-nesting females tagged at South African nesting beaches. Figure 1 from Robinson *et al.* (2016), Image: Figure (1a) from Robinson *et al.* (2016), Scientific Reports, *https://www.nature.com/articles/srep37851*.



8.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the SW Indian DPS.

8.1.1 Abundance

We estimated the total index of nesting female abundance for the SW Indian DPS to be 149 females. We based this total index on the following data, summarized in Table 14 and Table 15 and explained in detail below: nest monitoring data from Ezemvelo KwaZulu-Natal Wildlife (Ezemvelo 2018); and nest monitoring data from Centro Terra Viva Estudos e Advocacia Ambiental (CTV 2018). This total index 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 only occurs 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. Throughout this section, we present abundance indices for South Africa and Mozambique separately, because we received these datasets separately; however, as described under Section 8.1.3 Spatial Distribution, nesting females move freely between South African and Mozambican beaches, which comprise a single breeding aggregation for the entire DPS.

Table 14. Available nesting data for the SW Indian DPS. Number of nests (or other units, as identified) recorded for the first and last years monitored at surveyed nesting beaches. We also include the highest and lowest number of recorded nests (or other units, as identified). We calculated the index of nesting female abundance by summing the number of nests over the most recent remigration interval (i.e., 3 years; see Table 16) divided by the clutch frequency (7 clutches per season for South Africa and 2.25 and 7 clutches per season for Mozambique; see Table 16). We also provide the index of nesting female abundance for each nation.

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (years)**	**Reference**	**Index of nesting female abundance**
South Africa				134
"Monitoring area," 56 km	321 (1973) 315 (2016)	High: 578 (1994) Low: 88 (2001)	Ezemvelo 2018	(298+325+315)/7 = 134
Mozambique				15
Ponta Malongane to Ponta Dobela, 32 km	19 (1994) 36 (2017)	High: 49 (2009) Low: 6 (1997)	CTV 2018	(31+34+36)/2.25 = 45 (31+34+36)/7 = 15*
Inhaca Island, 12 km	3 (1988) 10 (2009)	High: 29 (1992) Low: 0 (2005)	Louro 2014	Unquantified
Bazaruto Archipelago, 50 km	132 eggs (1997) 115 eggs (2004) Additional years: 1999–2001, 2003	High: 690 eggs (2000) Low: 75 eggs (2003)	Louro 2006	Unquantified

*Two estimates are provided for Mozambique because we have an estimated clutch frequency from Mozambique; however, it is much lower than the clutch frequency of South Africa, and turtles move between these nesting beaches. The South African clutch frequency is based on more data and is more conservative to the species; therefore, we use this estimate in the total index.

Table 15. The number of nesting sites by index of nesting female abundance. We estimated the total index of nesting female abundance (for the DPS) by summing the indices of nesting female abundance from Table 14. We calculated the percentage at the largest nesting site by dividing that site's index of nesting female abundance (Table 14) by the total index of total nester abundance for the DPS.

Index of nesting female abundance	**Number of nesting sites**
Unquantified*	2
1–10	
11–50	1
51–100	
101–500	1
501–1,000	
1,001–5,000	
5,001–10,000	
10,001–20,000	
>20,000	
Total number of sites	4
Total index of nesting female abundance (DPS)	149
Confidence in total index of nesting female abundance	Moderate (two-thirds of the coastline is not monitored)
Largest nesting site, percentage of total index	South Africa, 75+ percent

*Sites included in Table 14 but not included in the total estimate for the DPS because recent data are not available over one migration interval. These sites may represent additional nesters in the DPS, but data are outdated or the sites are not consistently monitored.

Our total index of nesting female abundance is 149 females. 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.* 2013c). 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.* 2013c; 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: n = 375 emergences and n = 336 nests in South Africa (Nel 2010);

and n = 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 – 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. In the following paragraphs, we provide details on our indices of nesting female abundance for South Africa and Mozambique and describe their significance to this DPS.

In South Africa, we estimated the index of nesting female abundance to be approximately 134 females. To calculate this estimate, we requested the monitoring data from Ezemvelo (formerly Natal Parks Board). In South Africa, nesting occurs between Cape Vidal and the Mozambique border, on beaches within the iSimangaliso Wetland Park. Since 1965, Ezemvelo has conducted foot patrols of the "index area," a 12.8 km stretch of beaches between Bhanga Nek and the mouth of Kosi Bay, which lies 3.2 km south of the Mozambique border (Hughes 1996; Nel *et al.* 2013; Nel *et al.* 2015). Since 1973, Ezemvelo has conducted foot and vehicular patrols throughout the expanded "monitoring area," a 52.8 km stretch of beaches between Mabibi and the mouth of Kosi Bay (Hughes 1996; Nel *et al.* 2013; Nel *et al.* 2015). Ezemvelo provided nesting data (i.e., number of nests) covering 52 years (1965 to 2016) at the index and monitoring areas. We used the monitoring area data to calculate the nesting female abundance (n = 134) as follows: we divided the total number of nests (298 + 325 + 315 = 938) between 2014 and 2016 nesting seasons (i.e., a 3 year remigration interval) by the clutch frequency (7 clutches/season; see Table 16 for more information on remigration interval and clutch frequency estimates).

Our estimate of nesting female abundance for South Africa is similar to published estimates based on similar monitoring area data, though with some differences due to the time frame of analysis. Nel *et al.* (2013) identified 2,578 nesting females over 45 years (1965 to 2009), a mean of 69.4 ± 38.1 nesting females per season 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 African are minimum estimates because nesting occurs outside of 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 either 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 of 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). Given the Thorson *et al.* (2012) and Harris *et al.* (2015) data, nesting female abundance in South Africa may be greater than our estimate (i.e., 268 to 402 nesting females, Harris *et al.* 2015; 1,000 nesting females, Thorson *et al.* 2012). Thus, the limited range of monitoring is a source of uncertainty for this DPS, and our calculation reflects an estimate of nesting female abundance.

In Mozambique, we estimated the index of nesting female abundance to be 15 females, using the clutch frequency for South Africa. To calculate this estimate, we requested nest monitoring data from CTV. Director Marcos Pereira of CTV provided 24 years of nest count data from 32 km within the Ponta do Ouro Partial Marine Reserve, between Ponta Malongane to Ponta Dobela (CTV 2018). Two datasets were included: data collected by Pierre Lombard from 1994 to present; and data collected by a community monitoring program from 2010 to present. For most years, data were only available for December or December and a portion January; however, since 2012, the community monitoring program collected data over the entire season (from September or October to March). We used the community monitoring program data to calculate the nesting female abundance (n = 45) as follows: we divided the total number of nests (31 + 34 + 36 = 101) between 2015 and 2017 nesting seasons (i.e., a 3 year remigration interval) by the clutch frequency (2.25 clutches/season for Mozambique; see Table 16 for more information on remigration interval and clutch frequency estimates). However, these data are based on a limited sample size of 21 clutches from two locations in Mozambique (Louro 2006), and it is possible that other clutches were not detected (see Thorson *et al.* 2012; Harris *et al.* 2015). Therefore, we also calculated the index of nesting female abundance using the clutch frequency from South Africa, which is based on more data (7 clutches/season). We divided the total number of nests (31 + 34 + 36 = 101) between 2015 and 2017 nesting seasons (i.e., a 3 year remigration interval) by the clutch frequency (7 clutches/season). Therefore, we use this estimate (n = 15) as the index of nesting female abundance for Mozambique.

Published studies provide additional data on nesting in Mozambique. From 2007 to 2014, CTV recorded an average of 28 nests (SD = 11) per season and tagged an average of 8 nesting females (SD = 3.2) per season between Ponta Malongane and Ponta Dobela, where 90 percent of nests in Mozambique occur (Pereira *et al.* 2014). During the 2016 nesting season, CTV surveyed and monitored 288 km of beach (10.4 percent of the Mozambican coastline; Fernandes *et al.* 2018). At the Ponta do Ouro Partial Marine Reserve, they tagged 12 nesting females and resighted seven; they also recorded 98 tracks and 64 nests between Ponta do Ouro and the Bazaruto Archipelago (Fernandes *et al.* 2018). In the 2014 nesting season, they recorded 43 nests over a smaller area (127 to 137 km; Fernandes *et al.* 2015). At Inhaca Island, the Estação de Biologia Maritima da Ilha da Inhaca observed 172 nests between 1988 and 2009, with an annual average of 7.55 ± 7.54 nests (Louro 2014). They also observed leatherback nests in nearby locations: two nests between 1988 and 2009 at Portuguese Island and one nest in 2009 at Macaneta (Louro 2014). In the Bazaruto Archipelago, where monitoring occurs between October and February, Louro (2006) reported the number of eggs per season for six years between 1997 and 2004, with a low of 75 eggs in 2003 and a high of 690 eggs in 2000. We identified Inhaca Island and the Bazaruto Archipelago as unquantified nesting sites in Table 15. Because these areas do not have recent, standardized beach monitoring, we did not include them in our index of nesting female abundance; however, we think that their contribution is very small (i.e., one or two annual nesters).

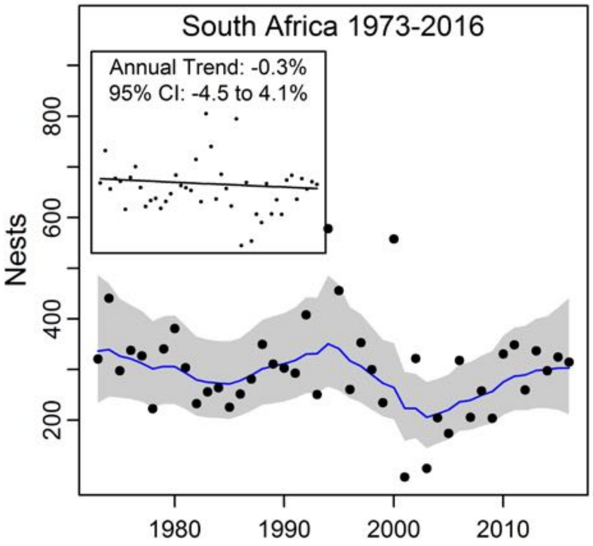
Compared to other published estimates for Mozambique (e.g., n = 10; Hamann *et al.* 2006), our estimate of nesting female abundance (n = 15 to 45) is high. One source of discrepancy is using the clutch frequency for Ponta Malongane (2.25 clutches per season; Louro *et al.* 2006), which is low for the species. 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 (n = 15) is closer to published estimates. Though there is uncertainty regarding the precision of this estimate due to uncertainty in the clutch frequency, based on the best available data, the index of nesting female abundance is approximately 15 females in Mozambique.

The total index of nesting female abundance (n = 149) places the DPS at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017a). 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. We do not have pre-exploitation abundance estimates with which to provide context; however, it is unlikely that the DPS is at or near carrying capacity (Nel *et al.* 2013). In South Africa, where the majority of the DPS nests, Nel *et al.* (2013) found no evidence for low carrying capacity: females forage in multiple areas where jellyfish are prevalent; body condition, clutch sizes, and hatching success indicate that nesting females are healthy; there is ample available nesting space; and there is no evidence of nests being dug up by other sea turtles or significant beach predation. Therefore, the total index of nesting female abundance is likely an indicator 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.

8.1.2 Productivity

The SW Indian DPS exhibits a slightly decreasing nest trend (Figure 31). We base our conclusion on data consistently collected in a standardized approach in the 56 km South African monitoring area (Ezemvelo 2018), where nest counts decreased annually (median = −0.3 percent; 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. As with all DPSs, we report the BSSM trend analysis results as the median and CI, which reflects that there is a 95 percent chance that the trend falls between the low and high CI values. The wider the CI, the less confident we are in the estimated median trend. The higher the “f statistic” the more confident we are in the sign (positive or negative) of the estimated median trend. The slightly decreasing trend in South Africa is representative of the entire DPS, as 80 to 90 percent of nesting is estimated to occur there (Wallace *et al.* 2013c; Nel *et al.* 2015), and the 44–year time series is quite robust.

Figure 31. Nest trend at monitoring beaches in South Africa. The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (nests). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the inter-annual variability.

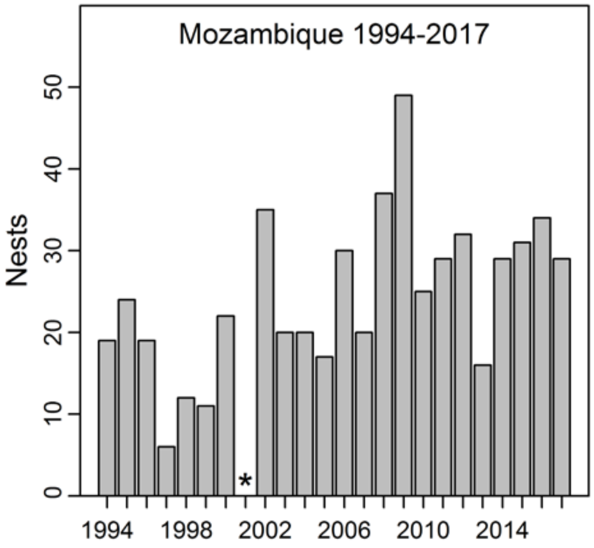


Our trend estimate yields similar results to other published findings. The IUCN Red List concluded that this population has declined slightly, by 5.6 percent over the past three generations, with an annual decline of 0.1 percent in South Africa and 0.7 percent in Mozambique (Wallace *et al.* 2013c). Hamann *et al.* (2006) also identified a declining trend in the nesting population of the SW Indian Ocean. Though the trends were calculated using different methods and different time frames, the consistency of these results (i.e., slightly declining trend in all studies) provides some confidence in the trend for the DPS. However, studies focused on the South African monitoring area (i.e., the source of data for our trend analysis) disagree on the nature of the trend.

Nest trends at the South African monitoring beaches have been variously characterized as: recently declining by Hamann *et al.* (2006) and Nel *et al.* (2013); stable and lacking population growth by Nel *et al.* (2015); and stable by Saba *et al.* (2015). Nel *et al.* (2013), who characterize the trend as declining or recently declining (depending on the dataset used), provide the best available published data because they are based on the most recent, primary data. When using the monitoring area data, which covers a larger area (52.8 km) over a shorter time frame (1973 to 2009), Nel *et al.* (2013) found a declining trend since 1994; when using the index area data (12.8 km from 1965 to 2009), they found an increase in nesting during the first decade, followed by oscillation, and a more recent decline (Nel *et al.* 2013). Nel *et al.* (2015) cites this study, stating: "recent quantitative analysis of long-term data confirms that the number of nesting females increased after initiation of beach conservation, but that population growth was not sustained over time (Nel *et al.* 2013)." In the next sentence, however, Nel *et al.* (2015) concludes "despite being small, the population is stable with consistent conservation efforts continuing." We think that this discrepancy reflects linguistic uncertainty (i.e., imprecise language) because different conclusions (i.e., declining or stable) are drawn from the same 2013 data and analyses. The Saba *et al.* (2015) and Hamann *et al.* (2006) conclusions (stable and declining, respectively) are based on earlier data. Hamann *et al.* (2006) state, "Data from the index beach shows a rise from 10 to 20 nesting females per year in the 1960s, and up to approximately 100 nesting females annually in the 1990s, but in the last four years it has declined to approximately 20 to 40 nesting females visiting the index beach per year." Saba *et al.* (2015) based their population trend conclusions on the Saba *et al.* (2008) "nesting numbers over the past 5 to 10 years." Because neither study provides an explanation of their methods, it is not possible to determine whether the discrepancy is a result of linguistic or epistemic (i.e., different datasets or analyses) uncertainty. A modeling study indicates that the slightly declining nest trend is likely due to declining detectability caused by nesting in unmonitored areas (Thorson *et al.* 2012). Harris *et al.* (2015) suggests that population growth trends may be greater than currently recognized because monitoring only detects approximately 34 percent (on average) of leatherback nesting; however, it is as likely that nest trends are less than currently recognized (i.e., the unknown quantity is as likely to be lower or higher than the estimate). We conclude that the data from the monitoring area are sufficient to provide a representative sample of the overall nest trend. These data have been collected consistently and in a standardized manner for decades. Therefore, based on the best available primary data (Ezemvelo 2018) and supported by the best available published data (Nel *et al.* 2013), the nest trend in South Africa is either stable or slightly declining. 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 conclude that the nest trend in South Africa is slightly declining; however, the nest trend may be stable if nesting in unmonitored areas has increased over time (Thorson *et al.* 2012; Harris *et al.* 2015).

We did not perform a trend analysis on the Mozambican data because we did not have 9 years of data collected in a consistent and standardized manner (i.e., methodology has changed in recent years) and the mean annual nests of 25 fell below our threshold of 50 nests. Instead, we include bar graphs of the data (Figure 32). Nest counts at Ponta do Ouro Partial Marine Reserve from Ponta Malongane to Ponta Dobela, Mozambique, suggest a possible increase from 1994/1995 to 2017/2018; however, the numbers are relatively low and could simply reflect natural interannual variability in nesting activity. It is unlikely that the slight decline in nesting at South African beaches is a result of females relocating to Mozambican beaches (Hamann *et al.* 2006).

Figure 32. Bar graph of nests at Ponta do Ouro Partial Marine Reserve. Asterisk indicates a year when data were not available.



Published information on the Mozambique nest trend reflect epistemic uncertainty (i.e., due to inconsistent sampling), generally concluding that there is no increasing trend, though failing to indicate whether the trend is stable or decreasing. For example, Hamann *et al.* (2006) reported that there does not appear to be an increase in leatherback nesting in Mozambique. At Inhaca Island, where few turtles nest, Louro (2014) reports no significant increase ($F(1,20) = 0.0691$; $p = 0.416$), with the number of nests ranging from 0 to 30 between 1988/1989 to 2009/2010. We conclude that there is a high degree of uncertainty regarding the nest trend in Mozambique.

Despite the recent decline (or lack of continued growth) in nesting, productivity parameters remain high for the DPS (Table 16). The SW Indian DPS achieves 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 ± 38.1 nesting females per season (Nel *et al.* 2013). We have high confidence in the South African productivity metrics, as a result of the long-term and extensive monitoring. We have moderate confidence in the Mozambican productivity metrics because of less extensive monitoring.

Table 16. Productivity parameters for the SW Indian DPS.

Productivity	**Variable by nation**	**Reference**
Size of nesting female (CCL cm)	South Africa: 160	Hughes 1996; Nel *et al.* 2013; Saba *et al.* 2015
	Mozambique: 157.5 (range 145.5 – 175)	Louro 2006
Remigration interval (years)	South Africa: 2–3	Hughes 1996; Lambardi *et al.* 2008; Nel *et al.* 2013; Saba *et al.* 2015
Clutch size (eggs)	South Africa: 104 (range 39 – 154)	Nel *et al.* 2013; Nel *et al.* 2015; Saba *et al.* 2015
	Mozambique: 134.2 total (range 62 – 199), may include eggs and shelled albumen gobs	Louro 2006
Clutch frequency (nests per season)	South Africa: 6 – 8	Nel *et al.* 2013; Saba *et al.* 2015
	Mozambique: Ponta Malongane = 2.25 ± 3.86; Bazaruto Archipelago = 2.00 ± 2.00	Louro 2006
Internesting interval (days)	South Africa: 9.5 (SD 1.4)	Nel *et al.* 2013; Robinson *et al.* 2017
Hatching success (percent)	South Africa: 70–80	Nel *et al.* 2015; Santidrián Tomillo *et al.* 2015
	Mozambique: 86.9 (based on nine nests with 1,005 eggs)	Fernandes *et al.* 2015

While the productivity metrics of the DPS remain high, the overall nest trend appears to be slightly decreasing, driven by the trend in South Africa, where the majority of the DPS nests. Overall, we have moderate to high confidence in productivity and trend for this DPS, due to high confidence in South Africa data but less confidence in data from Mozambique, which also hosts far fewer nesting females. The decline may reflect past and current threats that exceed the population's high productivity metrics. A population growth rate below replacement levels would create a future scenario of increasingly lower abundance, even if the threats remained constant; increasing or additional threats would further worsen this scenario. We conclude that the slightly declining nest trend places the DPS at elevated extinction risk, especially given the limited index of nesting female abundance.

8.1.3 Spatial Distribution

Despite the national boundaries and separate monitoring programs described above, the SW Indian DPS comprises, in essence, a single nesting aggregation. Tagging studies indicate that numerous females nest in both South Africa and Mozambique (Hughes 1996; Nel *et al.* 2015). Females exhibit little nest site fidelity. Within a season, repeat nesting events occur about 10 km apart (Botha 2010; Nel *et al.* 2015); however, internesting distances of over 600 km are possible (Robinson *et al.* 2016). Though genetic studies of fine-scale population structure have not been conducted, several studies have identified the DPS as a single nesting aggregation (Hughes 1996; Luschi *et al.* 2006; Nel *et al.* 2015). As such, the DPS demonstrates no metapopulation dynamics or substructuring.

Nesting occurs along South African and Mozambican coasts, across a total distance of approximately 900 km (Nel *et al.* 2015). For this DPS, nesting has not been documented elsewhere in Africa or Indian Ocean islands (Hamann *et al.* 2006; 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 monitoring beaches are 5 to 15 km long and 50 to 100 m wide (Nel *et al.* 2015; Robinson *et al.* 2016) and do not appear to be at carrying capacity for leatherback nesting (i.e., nesting density is low and nesting success is high; Nel *et al.* 2013).

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.* 2003a, 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 Retroflection 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 against local catastrophes or environmental changes that would limit prey availability (de Vos *et al.* 2019). 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).Without metapopulation structure, in which an extirpated subpopulation is recolonized by individuals from another, the DPS has reduced capacity to withstand other catastrophic events. We conclude that the effects of a widely distributed foraging areas, nesting distribution, and lack of population structure likely result in little effect on the extinction risk of the SW Indian DPS.

8.1.4 Diversity

For 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 ± 0.078 and nucleotide diversity =

0.0004 ± 0.0004; Dutton *et al.* 2013b). 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 deprives the DPS of the raw material necessary for adapting to long-term environmental changes, s1uch as cyclic or directional changes in ocean environments due to natural and human causes (McElhany *et al.* 2000; NMFS 2017a). We conclude that limited overall diversity increases the extinction risk of this DPS by reducing its resilience to threats.

8.2 ESA Section 4(a)(1) Factors

The following sections describe and assess the 4(a)(1) factors or threats. For each, we evaluate the best available information on the threat. We also describe, if possible, the exposure and impact of each threat.

8.2.1 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 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). 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. Nel *et al.* (2006) concludes that coastal erosion is a threat in South Africa, where the high-energy coastline varies seasonally. 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, as described under Section 8.2.5.3 Climate Change.

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. 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 (*https://whc.unesco.org/en/list/914/*; Hughes 2010; Robinson *et al.* 2016). Prior to then (since 1979), 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; Hughes 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 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).

Such protections have minimized vehicular traffic at nesting beaches in South Africa, but beach driving remains a threat on nesting beaches in Mozambique. Erosion is a threat to nesting beaches in both South Africa and Mozambique.

8.2.2 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 bycaught 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.* 2013c; Fernandes *et al.* 2018). Egg poaching still occurs in several locations within Mozambique (Nel 2012; Wallace *et al.* 2013c). 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. Hughes (1995) reported that nearly every nesting female was killed during the civil war (1977 to 1992). Lombard (2005) estimated that 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 (Hughes 1995; 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 (we are not aware of any nests) is also a threat in Madagascar, where leatherback turtles caught in gill nets are taken back to local villages and consumed, as reported 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 caught during a 2007 Malagasy village survey (Humber *et al.* 2010). Though 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) reports that the Malagasy law is not 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 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 poaching during nesting. However, members of the DPS 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 only. We conclude that overutilization, as a result of poaching in Mozambique and Madagascar, poses a threat to the DPS.

8.2.3 Disease and Predation

While we could not find any information on disease, 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., Nelson Mandela University, 2019). Nel (2006) reports current rates of predation as relatively low, and Nel *et al.* (2013) report 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 (deWet 2012).

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 is a much-reduced threat to the SW Indian DPS.

8.2.4 Inadequacy of Regulatory Mechanisms

The SW Indian 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.

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.* 2013c; 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 2005; 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 the nation has instituted an observer program (Petersen *et al.* 2009), few other at-sea conservation measures have been implemented (Honig *et al.* 2007). For Taiwanese fisheries operating within the range of this DPS, Taiwan has regulations to limit the number of vessels in the area and requiring that vessels carry de-hookers; however, bycatch and mortality remain high (Huang and Liu 2010). Similarly, though the extent of bather protection/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 to reduce exposure to and impact from bycatch 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 (*https://whc.unesco.org/en/list/914/*; Nel *et al.* 2015). iSimangaliso includes 280 km of beaches, rocky shores, mangroves, lakes, estuaries, and coastal waters out to three nautical mile (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 of 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^2 iSimangaliso Marine Protected Area (off nesting beaches); and the 6200 km^2 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, regulations to prevent the harvest of turtles and eggs have been adequate in South Africa but inadequate to prevent poaching in Mozambique and Madagascar.

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 are inadequate to sufficiently 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.

8.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

Other threats affect the SW Indian DPS, with fisheries bycatch being the primary threat. Additional threats include: vessel strikes; pollution; and climate change.

8.2.5.1 Fisheries Bycatch

Fisheries bycatch is the primary threat to the SW Indian DPS (Wallace *et al.* 2013a; 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, drift nets, gill nets, 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). Here, we review studies on longline bycatch of this population, organized by location (and described in detail below):

- 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 bycatch over a large portion of the DPS's foraging range
- In the Indian Ocean, Huang and Liu (2010) evaluate the Taiwanese fishery bycatch, which also likely includes turtles of the NE Indian DPS, and Louro (2006) describes illegal longlining in Mozambique waters
- In the SE Atlantic Ocean, Honig *et al.* (2007) and Angel *et al.* (2014) evaluate bycatch, which also likely includes turtles of the SE Atlantic DPS.
- Thorson *et al.* (2012) did not detect a relationship between reduced leatherback nesting and increased longline fishing effort; however, the authors state that a more recent or expansive index of longline fishing effort may provide a different result.

Harris *et al.* (2018) found a positive, significant relationship between the longline fisheries' extent of overlap with leatherback migratory corridors and threat intensity ($F_{1,8} = 184.7$, $P < 0.001$, $R2 = 0.95$), 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, 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) estimated the mean annual bycatch to be 7.8 (± 7.8 S.D.) leatherback turtles, based on 39 leatherback turtle captures reported over 5 years. However, 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 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 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 includes 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) 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 turtles from the much larger SE Atlantic DPS, but tracking 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 this 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 overshadow 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. Stranding records may provide some insight into additional fisheries impacts. For example, in Tanzania, nine leatherback turtles have stranded between 2008 and 2017, and several of those showed evidence of fisheries interactions (L. West, Sea Sense, pers. comm., 2019). In Kenya, Okemwa *et al.* (2004) evaluated 71 reported sea turtle strandings, of which leatherback turtles comprised one percent; fishing activities were thought to be responsible for up to 80 percent of all turtle mortality, with approximately 58 percent killed as a result of entrapment in fishing nets.

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 captures several (i.e., at least two to six but possibly many more) leatherback turtles annually (Louro 2006; Videira *et al.* 2010; SWOT 2018). In 2001, one shrimp trawler captain reported capturing more than six leatherback turtles since the 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 (± 40) leatherback turtles; the overall estimated mortality rate for turtles is 14 percent (Brito 2012). Given the overlap between the fishery and an important foraging area, Pereira (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; de Wet 2012; 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). In a small random sampling exercise for this fleet, 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 Economic Exclusive Zone 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 (Ciccione 2006).

Shark or bather nets, which are gill nets installed off beaches in South Africa to limit human-shark interactions, can result in bycatch of leatherback turtles. According to Nel *et al.* (2015), bather protection nets and boat strikes (addressed under 8.2.5.2 Vessel Strikes) together present the second greatest threat to the DPS, after fisheries. 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).

We conclude that 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 (± 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 bycaught and 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 a major, and the primary, threat to the SW Indian DPS.

8.2.5.2 Vessel Strikes

Vessel strikes are a threat to the SW Indian DPS. According to Nel *et al.* (2015), vessel strikes and bather protection nets (addressed under Section 8.2.5.1 Fisheries Bycatch on shark 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 stranded leatherback turtles was struck by a boat propeller (Nel 2008); however, additional mortalities or injuries may go unnoticed or unreported. This number includes 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 boat strikes pose a threat to the DPS.

8.2.5.3 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 (de Vos *et al.* 2019). It is unclear how increased sea surface temperatures, more extreme storm events, and altered ocean currents would impact the DPS. Therefore, we focus on increases in coastal erosion, sea level, and sand temperatures.

As described under Section 8.2.1 Present or Threatened Destruction, Modification, or Curtailment of Habitat or Range, 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, the rise in sea level (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 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). We add that a shift in the nesting season could have impacts beyond hatching success, such as post-hatchling survival and suboptimal foraging conditions for post-nesting females. We therefore conclude that increased temperatures may be a threat to the DPS, with changes 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. As 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.

8.2.5.4 Pollution

Pollution includes contaminants, marine debris, and ghost fishing gear. As with all leatherback turtles, entanglement in and ingestion of marine debris and plastics is a threat that likely kills 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 or lost 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 2018) 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 2018); however associated oil spills are likely to modify habitat off nesting beaches and reduce prey availability for all life stages (Pretorius 2018). 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. 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 (including plastics and oil spills) poses a threat to the DPS.

8.2.5.5 Oceanographic Regime Shifts

Oceanographic regime shifts do not appear to be a threat for the SW Indian DPS. Saba *et al.* (2015) describes this population as stable relative to the East Pacific population, in part due to this region's lower propensity for ecosystem collapse associated with oceanographic regime shifts. Post-nesting females disperse widely from their nesting beaches, foraging in waters off the Benguela and Angulhas Currents, the Subtropical Convergence, the upwelling region off the coast Africa, and deep-sea eddies in the SW Indian Ocean (Lambardi *et al.* 2008). Lambardi *et al.* (2008) concludes that these turtles do not target previously visited foraging areas but rather opportunistically use whatever suitable oceanographic features they encounter off nesting beaches. Therefore, alternative options may be available to them if one foraging area becomes less productive. Furthermore, at least one oceanographic variable is stable in the area: Robinson *et al.* (2017) reports that waters adjacent to nesting beaches exhibit a consistent increase in sea surface temperature over the leatherback turtle nesting season, from October to March. We conclude that oceanographic regime shifts are not a threat to this DPS.

8.3 Extinction Risk Analysis

After reviewing the best available information, we conclude that the SW Indian DPS has a high extinction risk, as summarized in the following paragraphs. The total index of nesting female abundance for this DPS is 149 females at monitored beaches. Such a limited nesting population size places this DPS at risk 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 increase the risk of extinction; however, productivity metrics, such as body size, clutch size and frequency, and hatching success, appear to be at or above average for the species. 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 other 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 all demographic factors, we find the DPS to be at an increased risk of extinction, likely as a result of past threats.

Current threats also contribute to the extinction risk of this DPS, as summarized in Table 17. 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 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. Boat 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 likely result in minor reductions in productivity; however, these threats are likely to increase over time as climate change and tourism increases. Climate change may result in further coastal erosion in the beaches causing increased nest inundation. Predation of eggs and hatchlings is also a threat: though it 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 trend.

Table 17. Threats to the SW Indian DPS. Exposure refers to the individuals affected by the threat. Impact refers to how the threat affects the demographic factor(s). The primary threat is identified with an asterisk.

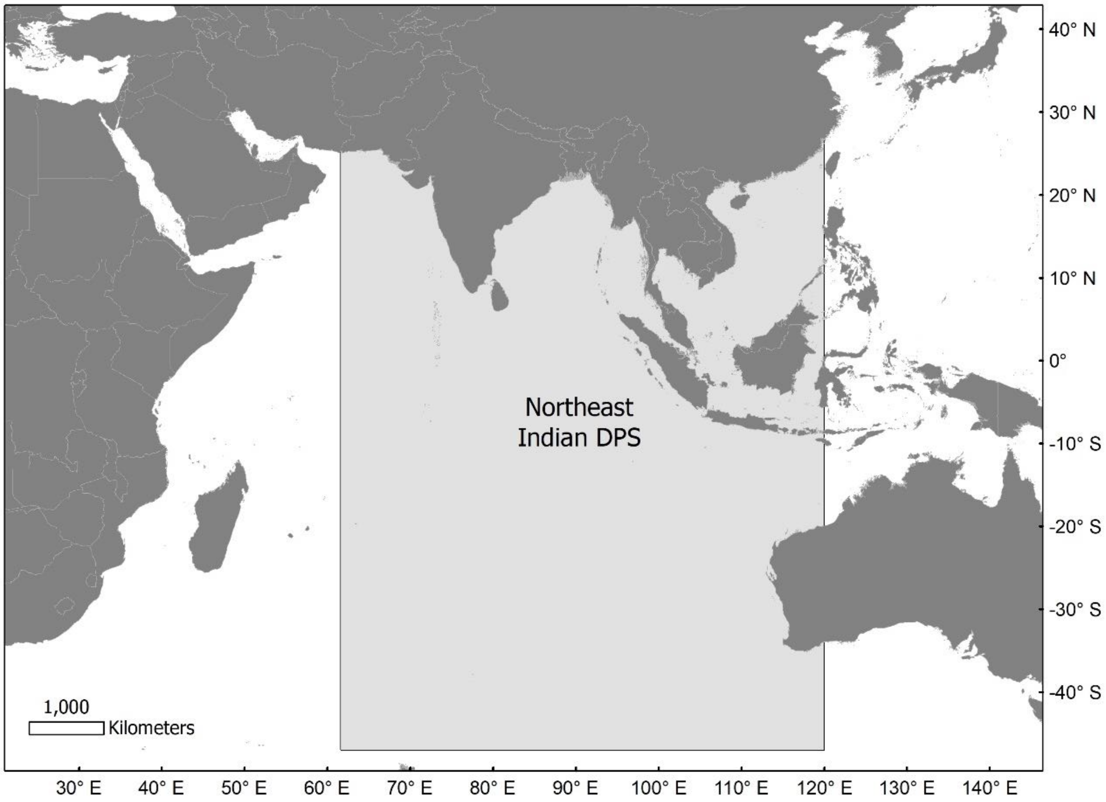
Threat	**Exposure**	**Impact**
Destruction or modification of habitat	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Overutilization	Few eggs and nesting females; few turtles at sea	Loss of nesting females (abundance) and reproductive potential (productivity)
Predation	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Inadequate regulatory mechanisms	Eggs and turtles of all life stages	Some laws are poorly enforced
Fisheries bycatch*	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) and reproductive potential (productivity)
Vessel strikes	Adults off nesting beaches	Loss of individuals (abundance) and reproductive potential (productivity)
Pollution	Eggs and turtles of all life stages	Lethal (abundance) and sublethal (productivity) effects
Climate change	Eggs and turtles of all life stages	Reduction of nesting and hatching success (productivity)

Thus, we find that the SW Indian DPS is at a high level of extinction risk. Its limited nesting female abundance places its continued persistence in question. The slightly declining nest trend and lack of spatial structure and diversity further contribute to our concern over its continued persistence. While we acknowledge that this small population had an increasing or stable nest 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. Now, fisheries bycatch is the primary clear and present 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. We conclude that the SW Indian DPS meets the definition for high risk of extinction (see Section 1.1.3.3 Extinction Risk Assessment). The moderate risk definition does not apply because the DPS is at a high risk of extinction now (i.e., at present), rather than on a trajectory to become so in the foreseeable future. We have high confidence in our conclusion because the abundance is extremely low and the threat of fisheries bycatch is large relative to population size.

9.0 Northeast Indian DPS

We define 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 (Figure 33). 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 restricts 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.

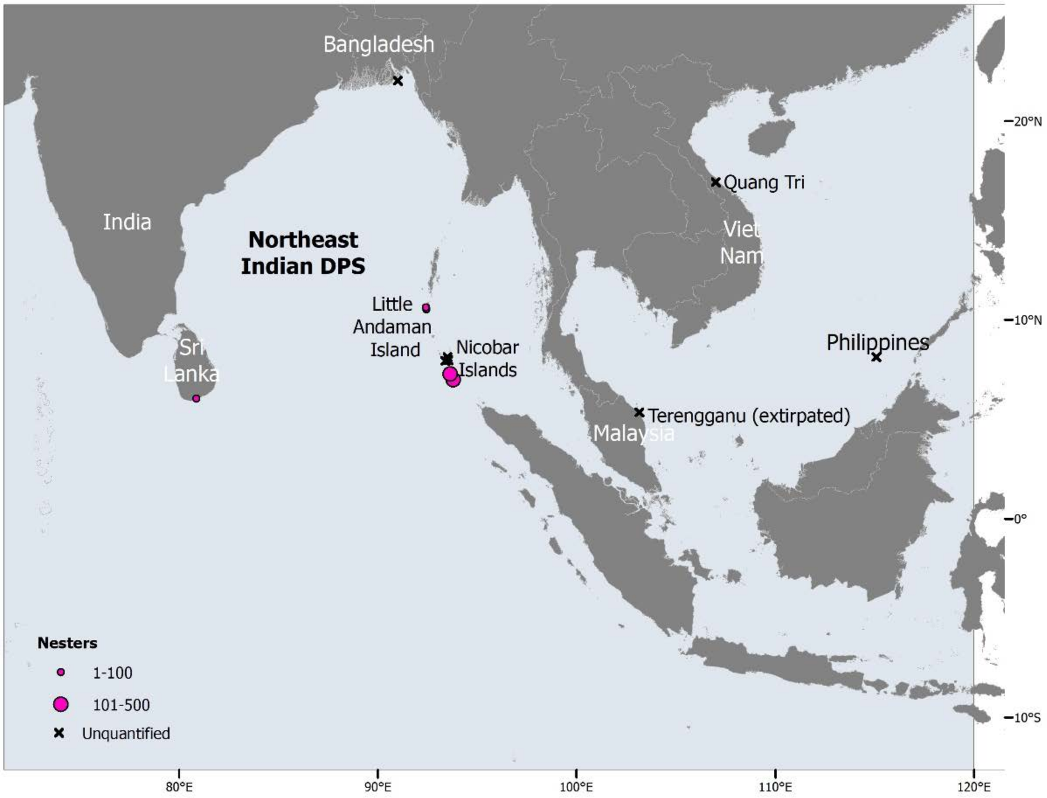
Figure 33. NE Indian DPS boundary map.



The range of the DPS (i.e., all areas of occurrence) extends throughout the Indian Ocean and possibly into the Pacific Ocean as well. Leatherback turtles may occur in the waters of the following nations: Pakistan, India, Sri Lanka, Bangladesh, Myanmar, Thailand, Malaysia, Indonesia, Australia, Brunei, Cambodia, Vietnam, China, Philippines, and Taiwan. 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).

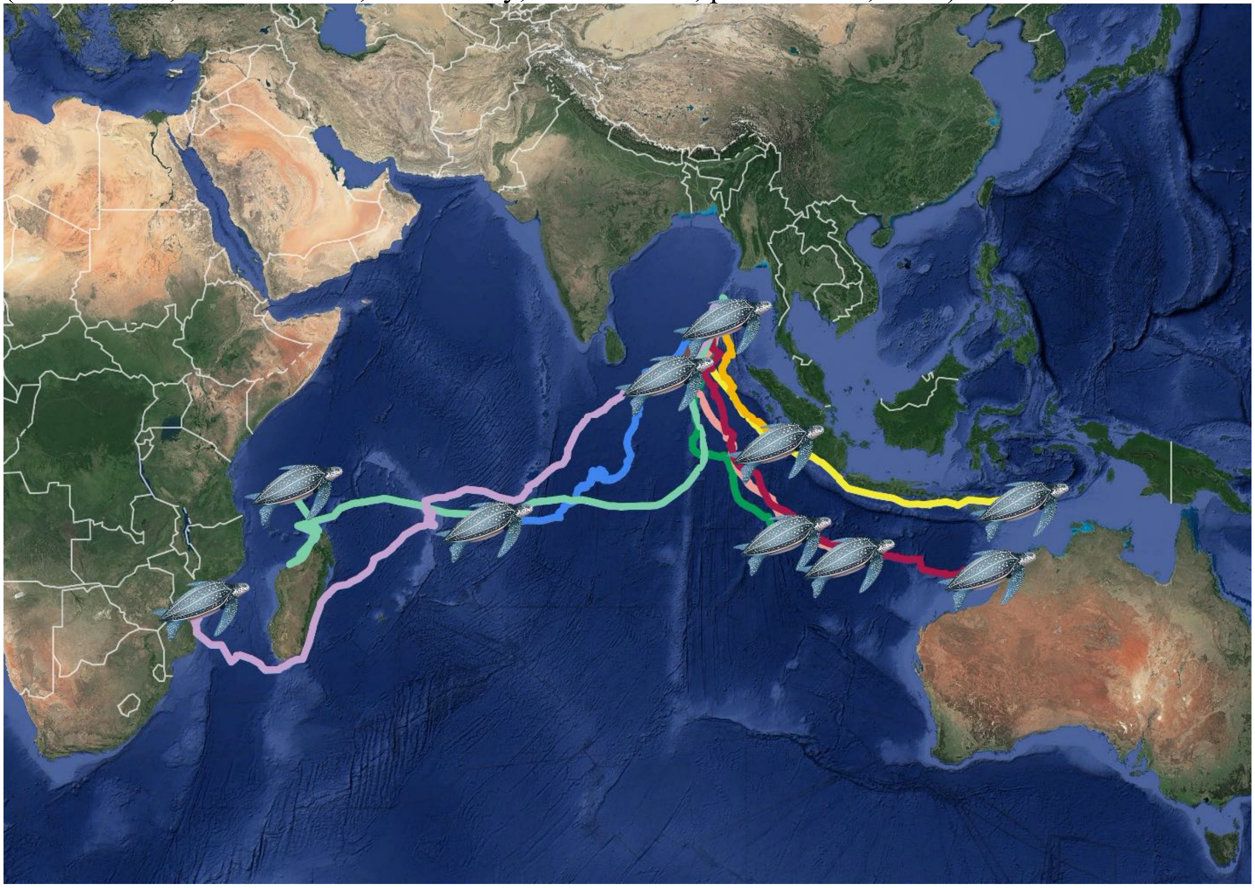
Leatherback turtles of the NE Indian DPS nest on beaches scattered throughout the NE Indian Ocean (Figure 34). The largest abundance of nesting occurs on beaches of the Andaman and Nicobar Islands in India. The sandy beaches of the Andaman and Nicobar Islands consist of soft limestone formed of coral and shell (Lal 1976; Bandopadhyay and Carter 2017). A moderate amount of nesting occurs in Sri Lanka, and even less in Thailand and Sumatra, Indonesia (Hamann *et al.* 2006; Nel *et al.* 2015).

Figure 34. Nesting sites of the NE Indian DPS. Size of circle represents the index of nesting female abundance based on current and outdated (yet best available) information. An "X" indicates that nesting was documented but not quantified.



Information on this DPS is limited, but foraging appears to occur throughout the Indian Ocean (Hamann *et al.* 2006, Andrews *et al.* 2006), as demonstrated by telemetry data (Namboothri *et al.* 2012a; Swaminathan *et al.* 2019; Figure 35). 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, as tracked by satellite telemetry 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.* 2012a; Swaminathan *et al.* 2017; Swaminathan *et al.* 2019). Stranding data also indicate the use of diverse foraging areas. Fifteen individuals stranded or were caught in fishing gear along the coastal mainland of India (Shanker 2013). Leatherback turtles have also stranded along the coasts of Mindanao, Philippines and Pakistan (Firdous 2001; Lucero *et al.* 2011).

Figure 35. Satellite tracks of 10 post-nesting females from Andaman and Nicobar Islands. Each track shown in a different color; leatherback icon denotes location of last transmission (Namboothri, Swaminathan, Choudhury, and Shanker, pers. comm., 2019).



9.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the NE Indian DPS.

9.1.1 Abundance

We estimated the total index of nesting female abundance of the NE Indian DPS to be 109 females. We based this index on the following data, summarized in Table 18 and Table 19 and explained in detail below. This number represents an index of nesting female abundance for this DPS because it only includes available data from recently and consistently monitored nesting beaches. Additional nesting occurs at other locations but is unquantified. Nesting surveys were not conducted throughout the nesting season at many locations, and only two surveys are available for Galathea, Great Nicobar Island.

Table 18. Available nesting data for the NE Indian DPS. Number of nests for the first and last years monitored at surveyed nesting beaches. We also include the highest and lowest number of nests recorded (or other units, as identified). We calculated the index of nesting female abundance using the data referenced as follows: summing the number of nests over the most recent remigration interval (i.e., 3 years; see Table 20) divided by the clutch frequency (3.8 clutches per season; see Table 20). We also provide an index of nesting female abundance for each nation when available data met our criteria.

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Bangladesh				Unquantified
Bangladesh	2001	1	Islam 2002	Unquantified
India*				109
South Bay, Little Andaman Island	25 (2008) 7 (2018)	High: 64 (2013) Low: 7 (2018)	Swaminathan *et al.* 2018	(7+55+29)/3.8 = 24
West Bay, Little Andaman Island	91 (2010) 62 (2018)	High: 176 (2017) Low: 7 (2017)	Swaminathan *et al.* 2018	(62+176+85)/3.8 = 85
Galathea, Great Nicobar Island	2011	146	Swaminathan *et al.* 2017	Unquantified
Great Nicobar Island	2016	775	Swaminathan *et al.* 2017	Unquantified
Little Nicobar Island	2016	229	Swaminathan *et al.* 2017	Unquantified
Kamorta Island	2016	6	Swaminathan *et al.* 2017	Unquantified
Katchal Island	2016	57	Swaminathan *et al.* 2017	Unquantified
Nancowry	2016	1	Swaminathan *et al.* 2017	Unquantified
<u>Indonesia</u>				Unquantified
Sumatra, Java, Bali		Occasional nest	Hamann *et al.* 2006	Unquantified

Nation, nesting beach, and length	Number of nests (first and last years monitored)	High and low number of nests (year)	Reference	Index of nesting female abundance
Mentawai island of Sipora off West Sumatra	2019	45 nests	Turtle Foundation Newsletter 2019	Unquantified
Malaysia				Unquantified
Terengganu, Rhu Cikgu beach in Rantau Abang	~10,000 (1950s) ~1 (2018)	High: 10,000 (1950s) Low: 0 (2009)	Zulkifli *et al.* 2004; K. Ibrahim pers. comm., 2004; TUMEC 2006; Pilcher pers. comm., 2018	Unquantified
Myanmar				Unquantified
Honeymoon Beach	2017	1	Platt *et al.* 2017	Unquantified
Sri Lanka				Unquantified
Godawaya and Rekawa Beaches	2001, 2002, 2005, 2010	may consist of 100 to 200 females annually (based on a year of data)	Ekanayake *et al.* 2002, Nel 2012	Unquantified
Thailand				Unquantified
Phanga and Phuket Provinces		<20	Nel *et al.* 2015	Unquantified
Philippines				Unquantified
Philippines		Occasional nest	Lucero *et al.* 2011	Unquantified
Vietnam				Unquantified

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Quang Tri Province, and Khank Province	2005, 2007, 2013, and 2014	1–2	Cuong pers. comm., 2018	Unquantified

* Some data are based on partial nesting seasons

Table 19. The number of nesting sites by index of nesting female abundance. We estimated the total index of nesting female abundance (for the DPS) by summing the indices of nesting female abundance from Table 18. We calculated the percentage at the largest nesting site by dividing that site's index of nesting female abundance (Table 18) by total index of nesting female abundance for the DPS.

Index of nesting female abundance	**Number of nesting sites**
Unquantified*	14
1–10	
11–50	1
51–100	1
101–500	
501–1,000	
1,001–5,000	
5,001–10,000	
10,001–20,000	
>20,000	
Total number of sites	16
Total index of nesting female abundance (DPS)	109
Confidence in total index of nesting female abundance	Low (nesting sites were not monitored for entire season)
Largest nesting site, percentage of total index	West Bay, Little Andaman Island 78 percent

*Sites included in Table 18 but not included in the total estimate for the DPS because recent data are not available over one migration interval. These sites may represent additional nesters in the DPS but data are outdated or not consistently monitored.

Our total index of nesting female abundance is likely an underestimate because we did not have adequate data from most nesting beaches. The IUCN Red List assessment did not provide an estimate of the total number of mature individuals as monitoring was not sufficient (Tiwari *et al.* 2013). Currently, the largest nesting aggregation occurs in the Andaman and Nicobar Islands in India. There is nesting in Sri Lanka, which may consist of about 100 to 200 nesting females per year, with even smaller nesting numbers in Thailand and Sumatra, Indonesia (Hamann *et al.* 2006; Nel 2012). 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; Islam *et al.* 2011). 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.

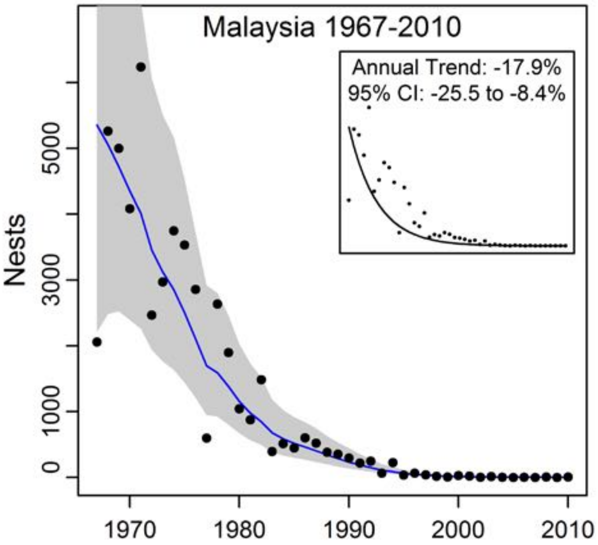
Malaysia once hosted the DPS's largest nesting aggregation (Chan and Liew 1996), which is now considered functionally extinct (i.e., extirpated) (Pilcher *et al.* 2013), as a result of continuous, large-scale egg harvest and fisheries bycatch (Chan and Liew 1996; Eckert *et al.* 2012). The major nesting 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, more recently, to one or no nests annually. The number of females nesting in Vietnam has also decreased dramatically, from approximately 500 nesting females in the 1960s to two to three nests in 2005 and 2007 (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.

The total index of nesting female abundance (n = 109) places the DPS at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017a). 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. Therefore, the total index of nesting female 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 limited nesting female abundance is a major factor in the extinction risk of this DPS.

9.1.2 Productivity

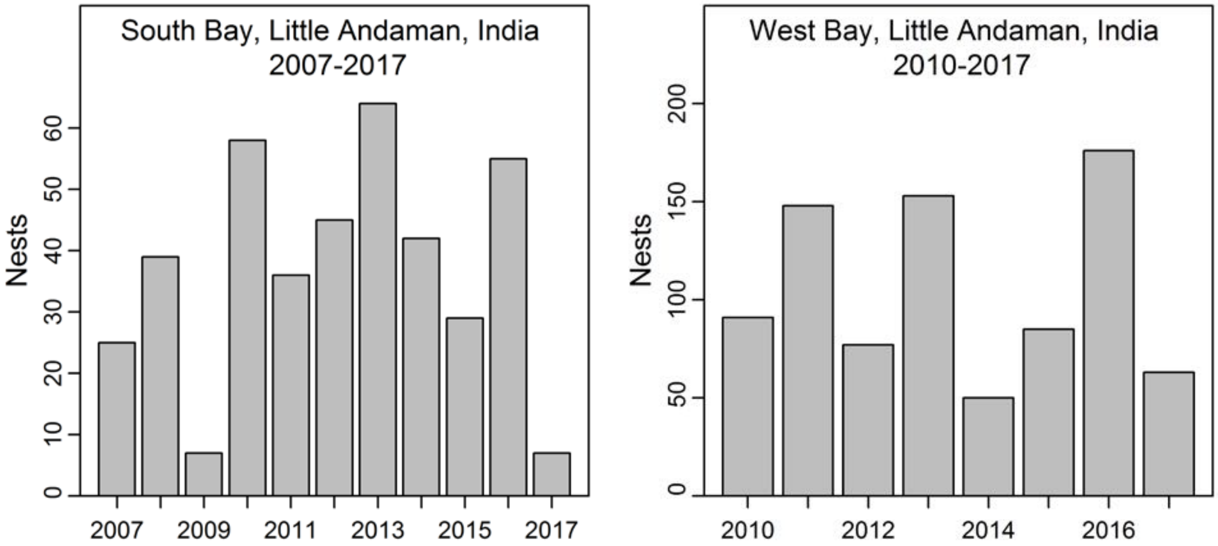
The NE Indian Ocean DPS has exhibited a drastic population decline with extirpation of its largest nesting aggregation in Malaysia (Figure 36). The only recent monitoring data for this DPS are from India, as there are no time series available from Sri Lanka or Vietnam, and consistent monitoring efforts in Malaysia ended in 2010. However, we conducted trend analyses for both India and Malaysia, as Malaysia's historical abundance was greater than 10 times the recent peak at West Bay, Little Andaman in India, which is currently the largest nesting site for the DPS (78 percent of nesting female abundance). As with all DPSs, we report the BSSM trend analysis results as the median and CI, which reflects that there is a 95 percent chance that the trend falls between the low and high CI values. The wider the CI, the less confident we are in the estimated median trend. The higher the "f statistic" the more confident we are in the sign (positive or negative) of the estimated median trend. 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). To run the trend model in natural log space, we changed the number of nests from 0 to 1 for 2007 and 2009; this is within the range of observation error and does not impact the trend estimate. The drastic decline of nests observed in Malaysia is 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.

Figure 36. Previous nest trend at Terengganu, Malaysia (extirpated). The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (nests). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the inter-annual variability.



For India, data were available from two sites on Little Andaman Islands (South Bay and West Bay); however, they did not meet our criteria for trend analysis. The nest count data from West Bay only spanned eight years (2010/2011 through 2017/2018; mean annual nests = 105), falling below our 9-year threshold. While there were 11 years of data from South Bay (2007/2008 through 2017/2018), the mean annual nests of 37 fell below our trend criterion of 50 nests. Thus, we include a bar graph for each site, neither of which exhibits an apparent trend (Figure 37).

Figure 37. Bar graphs of nests at South Bay and West Bay, Little Andaman Islands, India.



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 (Table 20). We have low confidence in these estimates due to limited sampling.

Table 20. Productivity parameters for the NE Indian DPS.

Productivity	**Variable by nation**	**Reference**
Size of nesting female (CCL cm)	Sri Lanka: 151.9	Kapurusinghe 2006a
	Little Andaman Islands: 159.2	Bansal *et al.* 2017
	Vietnam: 120–180	Pham Thuoc 2001
Remigration interval (years)	India: 2.5	Andrews 2002
Clutch size (eggs)	Sri Lanka: 82.8–100.5	Kapurusinghe 2006a
	Vietnam: 70–90	Pham Thuoc 2001
Clutch frequency (nests per season)	India: 3.8 Sri Lanka: 4.9	Andrews 2002, Eckert *et al.* 2015; Bhaskar 1993
Internesting interval (days)	[Andaman and Nicobar Islands]: 11	Eckert *et al.* 2015
Nesting Season	November to March	Eckert *et al.* 2015

The overall nest trend has drastically decreased over the past several decades. The productivity metrics are average for the species. Overall, we have low confidence in productivity metrics and trend for this DPS, due to an unknown percent of hatching success and the limited availability of consistent nesting estimates. We conclude that low nesting activity places the DPS at elevated extinction risk.

9.1.3 Spatial Distribution

For the NE Indian DPS, nesting is limited to a few scattered nesting beaches. Currently, the majority of the nesting occurs on 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 results have not been published (Namboothri *et al.* 2010). Further genetic sampling has been recommended for all the Andaman and Nicobar Islands, as well as northern and eastern Australia, Mozambique, Sri Lanka, Sumatra, Java, Thailand, and Vietnam (Dutton *et al.* 1999, 2007).

The wide distribution of foraging areas likely buffers the DPS 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 (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.

9.1.4 Diversity

Genetic diversity of the NE Indian DPS is relatively high based on analyses of the previously large 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 the diversity, when taken in context with nesting female abundance, provides little resilience to the DPS.

9.2 ESA Section 4(a)(1) Factors

The following sections describe and assess the 4(a)(1) factors or threats. For each, we evaluate the best available information on the threat. We also describe, if possible, the exposure and impact of each threat.

9.2.1 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 are threats to the NE Indian DPS.

Currently, the highest numbers of leatherback nesting occur in the Andaman and Nicobar Islands, which lie closest to the epi-center of the earthquake (Subramaniam *et al.* 2009). In 2004, a major earthquake occurred off the west coast of Sumatra, Indonesia, resulting in a tsunami. This tsunami destroyed many of the beaches that hosted over 1,000 nests (Subramaniam *et al.*

2009). It severely modified the nesting habitat at many islands, causing erosion at some and accretion at others. It also permanently altered large stretches of beach, particularly in the Nicobar Islands. Post-tsunami surveys of nesting beaches in the Andaman Islands showed modified beaches with some areas showing signs of nesting (Andrews *et al.* 2006).

Sand mining and coastal development as a result of tourism related activities are main threats to the 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 coastal development including tourism (large hotels and restaurants), urban and industrial development, and artificial lighting (Kapurisinghe 2006b). Artificial lighting modifies the quality of nesting beaches because artificial lights, visible from the nesting beach, 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. Nests moved to hatcheries are still under threat from hatchery practices, which have resulted in skewed sex ratios and low hatching success (Chan and Liew 1996; Kapurisinghe 2006b; Rajakaruna *et al.* 2013; Phillott and Kale 2018). Along the mainland of India, granite blocks and embankments prevent turtles from approaching many beaches (Andrews *et al.* 2006). Intense coastal development occurs in Bangladesh without environmental review (Pilcher 2006). The beaches in Bangladesh are under threat from coastal development stemming from the tourism industry, and alterations of sand dunes and nesting beaches are a major threat to sea turtles (Islam 1999 as cited in Islam *et al.* 2011). Recreational activities and related lighting on these beaches decrease the quality of nesting habitat and hinder turtles from nesting successfully (Islam 2002).

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 on Terengganu (Rantau Abang Turtle Sanctuary; Ma'Daerah Turtle Sanctuary; PasirTemit, Hulu Terengganu; PasirLubokKawah, Hulu Terengganu; PasirKumpal, Dungun), Perek (Pantai Jabatan, Perak River), Sabah (Turtle Islands Park), Sarawak (Talang-Satang National Park). Coastal development continues to threaten all other nesting beaches (Chan 2004, 2006).

The beaches in Indonesia are being lost due to erosion from high tides and monsoons (Obermeier 2002). In Vietnam, most of the beaches have a large amount of marine debris, which includes glass, plastics, polystyrenes, floats, nets, and light bulbs. This debris can entrap turtles and impede nesting activity. With increasing tourism, coastal development is expected to increase on the beaches of the Son Tra peninsula and beaches in Quan Lan and Minh Chau (Ministry of Fisheries, 2003).

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. We were unable to find quantitative estimates of this reduction. The most abundant nesting aggregations are protected from development, but they experience erosion; while the other beaches are susceptible to high anthropogenic threats. Based on the information presented above, we find that habitat loss and modification pose a threat to the DPS.

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

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

Nearly complete, regular egg harvest was the main threat that contributed to the extirpation in Malaysia (Chan and Liew 1996). At Terengganu Malaysia, in the early 1960s, the nesting beaches were leased to the highest bidder, and nearly all leatherback eggs were harvested. In the 1980s, the State Fisheries Department banned the commercial sale and consumption of leatherback eggs and 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; Gomez and Krishnasamy 2010). Excessive egg harvest, both legal and illegal, also caused population declines or losses in India, Sri Lanka, and Thailand (Ross and Barwani 1982).

Current harvest of turtles and eggs continues in portions of the DPS 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; Hewavisenthi 2001; Rajakaruna *et al.* 2013; Phillott *et al.* 2018). Egg harvest also continues in Thailand. There is protection of nests from commercial egg harvest on the Andaman and Nicobar Islands, and in the Andaman Islands, a ban on hunting and harvesting of turtles started in 1977. However, indigenous groups of people, the original inhabitants of the Andaman and Nicobar Islands, are exempt from the Indian Wildlife Protection Act (Shanker and Andrews 2004). Bhaskar observed intense egg harvest at Delgarno, Trilby and East Turtle Islands (Namboothri *et al.* 2012b).

The consumption of turtles is also a threat to the DPS. In Myanmar, despite regulations that prohibit the eating of turtle meat and eggs (Hamann *et al.* 2006), there is illegal trade of turtle meat caught at sea, including leatherback turtles (Murugan 2007). In Sri Lanka, the historically high direct take of turtles at sea is now low (Kapurushinghe 2006a). Records indicate that turtle meat and parts were 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; Agastheesapillai 1996; Chandrasekar and Srinivasan 2013). We were unable to find historical or current estimates of leatherback turtle harvest.

Exposure to egg and turtle poaching remain high throughout the range of the DPS. Poaching of nesting females or post-nesting females (i.e., 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.

9.2.3 Disease and Predation

While we could not find any information on disease, the best available data on the NE Indian DPS indicate that egg and hatchling predation occurs on several nesting beaches by multiple predators. During a 2016 survey of the Nicobar region, Swaminathan *et al.* (2017) reported that approximately 57 percent (n = 1,223) of leatherback nests were lost to depredation by feral dogs, water monitor lizards, or feral pigs. The Asian water monitor lizard takes 68.6 percent of leatherback nests in the Andaman Islands (Sivasundar and Prasad 1996). 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). Egg predation by feral pigs was identified as a major threat in Indonesia (Maturbongs *et al.* 1993; Maturbongs 1995; Sivasundar and Prasad 1996). In Sri Lanka, egg predators include feral dogs, water and land monitor lizards, jackals, wild boars, mongooses, and ants. Crabs are natural predators of eggs and hatchlings (Kapurusinghe and Ekanayake 2000). Adult turtles are also subject to predation.

A large number of eggs and hatchlings, and a small number of nesting females, 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 nesting females reduces abundance and productivity, while egg and hatchling predation mainly impact productivity. We conclude that predation poses a threat to the NE Indian DPS.

9.2.4 Inadequacy of Regulatory Mechanisms

The NE Indian 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.

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 that promotes enforcement; and/or are not legally-binding. Lack of implementation or enforcement by some nations may render 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).

In addition to these international regulatory mechanisms, nearly all nations where the NE Indian DPS occurs have some level of national legislation directed at sea turtle protection. 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 tribes of the Andaman and Nicobar Islands are exempt from these laws. India also 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 Malaysia, the nesting beach at Rantau Abang in Terengganu State is protected; however, this nesting aggregation is functionally extinct. In 1994, the surrounding marine waters off 38 offshore islands in 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). The use of TEDs will be required in Malaysia by 2020. 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. Habitat protection includes: 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). Under the Wildlife Protection Ordinance 1998, all marine turtles 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).

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. The use of TEDs will be required in Myanmar by late 2018.

In Pakistan, the Baluchistan Wildlife Protection Act 1974 No.19/1974, The Azad Jammu and Kashmir Wildlife Act 1975 No.23/1975 and The Sindh Wildlife Protection Ordinance 1972 No.5/1972 confer total protection on the leatherback turtle in Baluchistan, Azad Kashmir and Sind respectively. 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). The nesting beach in Yala Reserve is protected. Under Section 30 of the Fauna and Flora Ordinance (as amended), it is an offence 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. Section 30 (1) of this Act also empowers the minister in charge of the subject of Fisheries and Aquatic

Resources to make regulations to prohibit or regulate the export from or import into Sri Lanka, of turtles or their derivatives (Parliament of the Democratic Socialist Republic of Sri Lanka 1993).

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. Though the regulatory mechanisms provide some protection to the species, many are inadequate to sufficiently 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 poaching remain threats to the DPS. We consider the magnitude of such threats in other sections. Although regulatory mechanisms partially address the direct and incidental take of leatherback turtles, in the remaining nesting sites, the regulatory mechanisms do not sufficiently minimize the threat. The regulatory mechanisms to reduce the level of bycatch are insufficient to ensure the survival of the NE Indian DPS. We conclude that the inadequacy of the regulatory mechanisms is a threat to the NE Indian DPS.

9.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

The NE Indian DPS is negatively affected by both natural and anthropogenic impacts. Fishery bycatch (longline, gill net, and trawl fishing) occurs throughout the range of the DPS. Additional threats to this DPS include: pollution, climate change, and natural disasters.

9.2.5.1 Fisheries Bycatch

Leatherback turtle bycatch from gill nets, trawls, purse seines, and longline fisheries is a significant cause of sea turtle mortality in the NE Indian DPS (Wright and Mohanty 2002; Hamann *et al.* 2006; Project GloBAL 2007; Bourjea *et al.* 2008; Abdulqader 2010; Wallace *et al.* 2010b). The magnitude of trawl, gill net, and longline fisheries within the NE Indian DPS is great with little substantive sea turtle protection measures in place to reduce sea turtle bycatch mortality.

Gill net fisheries pose a major threat to the DPS. A survey conducted at 16 main fishing ports in Sri Lanka indicated that 431 leatherback turtles were caught in gill nets 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, catch in drift gill nets and bottom longlines, respectively. Along the coast of Andaman and Nicobar Islands, where the largest nesting aggregations occur, the main types of fisheries are gill nets and purse seines (Shanker and Pilcher 2003; Chandi *et al.* 2012). In Bangladesh, gill nets, set bag nets, trawl nets, seine nets, hook and line and other net types of gear capture turtles (Hossain and Hoq 2010).

Trawl fisheries also pose a large threat to the DPS. In Malaysia, the Fisheries Act (1985) prohibited capture of sea turtles by any type of fishery. However, this led to interactions that were not reported (Yeo *et al.* 2011). Regulations in 1991 prohibited fishing in waters adjacent to Rantau Abang during the leatherback nesting season (Chan 2006). In India, trawl nets require TEDs; however, the fisheries are reluctant to use them (Murugan 2007). Trawl fishing is also common in Bangladesh, and the use of TEDs is not required (Khan *et al.* 2006). On June 20, 2019, the European Union passed a regulation (PE-CONS 59/1/19 Rev 1) that requires shrimp

Exhibit 8 to Declaration of Adam Keats

trawl fisheries to use a turtle excluder device in European Union waters of the Indian and West Atlantic Oceans.

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 the SW and NE Indian DPSs (Louro 2006). In Vietnam, longline fisheries continue to capture leatherback turtles; however, a circle hook program has minimized 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 Economic Exclusive Zone 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 (Ciccione 2006).

In Thailand, one of the main causes of decline in the turtle population is bycatch in trawl, drift gill net, 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).

We conclude that individuals of this DPS are exposed to high fishing effort throughout their foraging range and in coastal waters near nesting beaches. Records suggest that mortality is also high. Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when nesting females are bycaught and killed. We conclude that fisheries bycatch is a major threat to the NE Indian DPS.

9.2.5.2 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). Ghost nets in the Maldives primarily drift from fisheries in the Bay of Bengal (e.g., Sri Lanka and India; 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 reuse and recycling fishing gear, the threat continues throughout much of the range of the DPS. 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 be lethal to sea turtles through ingestion or entanglement (Kapurusinghe 2006; Das *et al.* 2016). Ingestion of plastics by leatherback turtles has increased dramatically over the last 65 years. Since 1969, 37.2 percent of leatherback turtles necropsied were found to have ingested plastics (Mrosovsky *et al.* 2009).

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 has 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 at relatively high levels in the waters of Malaysia (including Terengganu) with high levels found in leatherback pancreases and livers (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. For pollution, we were unable to determine how exposure impacts individuals. At present, we think that pollution poses a threat to the NE Indian DPS.

9.2.5.3 Natural Disasters

The Andaman and Nicobar Islands lie close to the epi-center of the earthquake that triggered the 2004 Tsunami (Subramaniam *et al.* 2009). The 2004 Tsunami caused drastic changes in the coastline and the leatherback nesting beaches (Ramachandran *et al.* 2005; Alfred *et al.* 2006, Andrews *et al.* 2006). One meter of the coastline was lost, and the beaches of South Bay, Little Andaman Islands were severely modified, resulting in very low leatherback nesting in 2005 and 2006 (Namboothri 2012b). Natural disasters have the potential to affect several important nesting beaches. They impact the DPS by affecting nesting and hatching success (i.e., productivity) at numerous beaches, for an entire season. We conclude that natural disasters pose a threat to the NE Indian DPS.

9.2.5.5 Climate Change

Similar to other areas of the world, climate change and sea level rise threaten the NE Indian DPS. A significant rise in sea level would reduce nesting habitat, as the majority of nesting occurs on islands. Over the long term, the DPS is likely to be threatened by the alteration of thermal sand characteristics from global warming (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 male. As temperatures increase, there is concern that incubation temperatures will reach levels that 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 the nesting conditions for the entire DPS because most of the nesting occurs on island beaches. Impacts likely range from small changes in nesting metrics to large losses of productivity. As there are already small impacts due to coastal erosion and sea level rise, we conclude that climate change is a threat to the NE Indian DPS.

9.3 Extinction Risk Analysis

After reviewing the best available information, we conclude that the NE Indian DPS has a high extinction risk, as summarized in the following paragraphs. The total index of nesting female abundance for this DPS is 109 females at monitored beaches; however, we have low confidence in this estimate because several nesting sites were not included due to lack of consistent, standardized monitoring over an entire nesting season. The once large nesting aggregation in Malaysia is now functionally extinct. Such a low nesting population size 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. This DPS exhibits average productivity metrics, such as body size, clutch size and frequency. The DPS has some spatial distribution and diversity, with multiple foraging sites and relatively high genetic diversity; however, most nesting occurs on islands. Based on these demographic factors, we find the DPS to be at an increased risk of extinction as a result of past threats.

Current threats also contribute to the risk of extinction for this DPS, as summarized in Table 21. The primary threats to the DPS include fisheries bycatch and the harvest of turtles and eggs. There are not many nests to exploit, but if found, the eggs are 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 from trawl and gill nets in Malaysia, India, Thailand, Sri Lanka, Bangladesh, and Indonesia is a high threat. Additional threats include erosion, predation, and natural disasters. Erosion on the Andaman and Nicobar Islands, as a result of tsunami damage, has significantly reduced available nesting habitat. Additional habitat modifications include high human population and activity on beaches that would likely prevent nesting in Malaysia, if turtles returned, and concrete blocks that would block nesting female access to beaches in India. Additional threats include pollution and climate change, which is likely to increase. These threats affect the DPS by lowering abundance (i.e., fisheries bycatch and overutilization) and reducing productivity (erosion, egg harvest, and egg predation). Though many regulatory mechanisms are in place, they do not reduce the impact of these threats to levels that ensure the continued existence of the DPS.

Table 21. Threats to the NE Indian DPS. Exposure refers to the individuals affected by the threat. Impact refers how the threat affects the demographic factors. The primary threats are identified with asterisks.

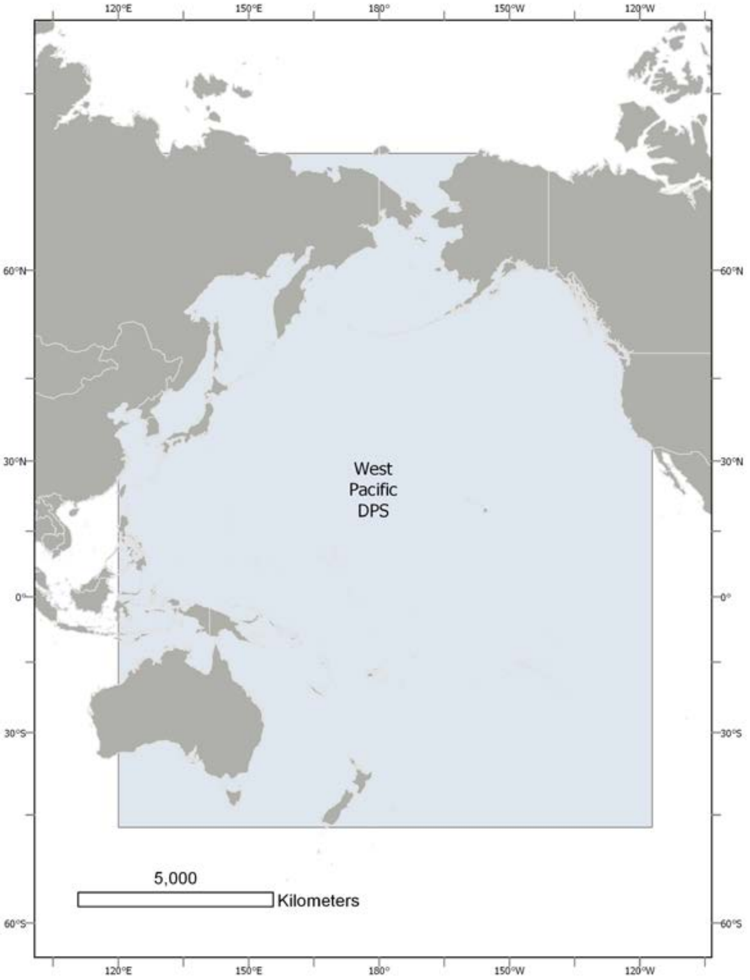
Threat	**Exposure**	**Impact**
Destruction or modification of habitat	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
*Overutilization	Eggs and few nesting females; few turtles at sea	Loss of nesting females (abundance) and reproductive potential (productivity)
Predation	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Inadequate regulatory mechanisms	Eggs and turtles of all life stages	Some laws are poorly enforced
*Fisheries bycatch	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) including loss of nesting females (productivity)
Natural disasters	Eggs and hatchlings and largest nesting aggregations	Reduction of nesting and hatching success (productivity)
Pollution	Eggs and turtles of all life stages	Lethal (abundance) and sublethal (productivity) effects
Climate change	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)

Thus, we find that the NE Indian DPS is at a high level of extinction risk. Its nesting female abundance places its continued persistence in question. Dramatic declines in nesting female abundance over the past several decades contribute to our concern over its continued persistence. 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 clear and present 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. We conclude that the NE Indian DPS meets the definition for high risk of extinction (see Section 1.1.3.3 Extinction Risk Assessment) because of its imminent and substantial demographic risks. The moderate risk definition does not apply because the DPS is at a high risk of extinction now (i.e., at present), rather than on a trajectory to become so in the foreseeable future. We have high confidence in our conclusion because the abundance is extremely low and the threats are significant.

10.0 West Pacific DPS

We define the West Pacific DPS as leatherback turtles originating from the West Pacific Ocean, with the following boundaries: south of 71° N, north of 47° S, east of 120° E, and west of 117.124° W (Figure 38). 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 areas off waters of Chile and Peru (Dutton *et al.* 1999).

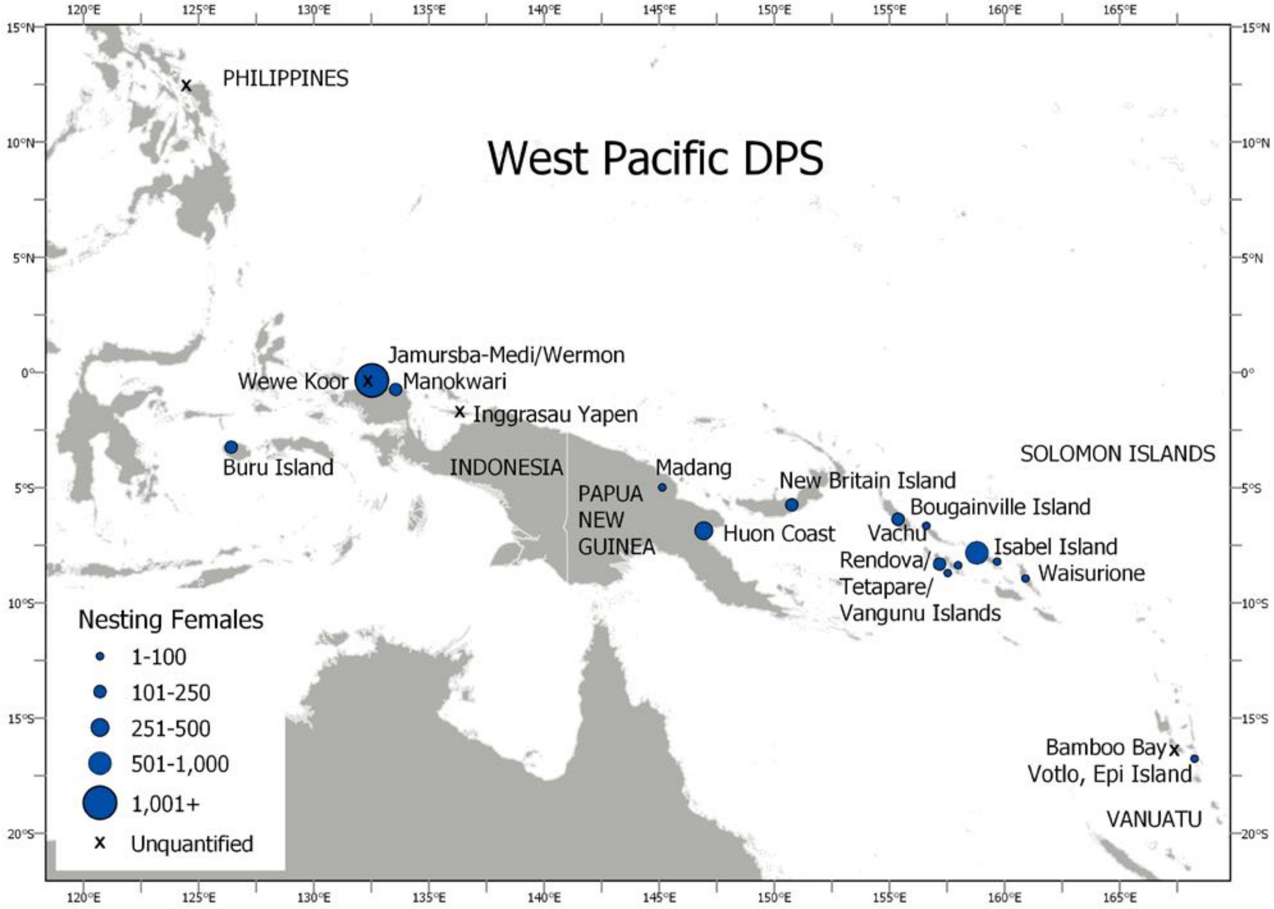
Figure 38. West Pacific DPS boundary map.



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. Leatherback turtles of the West Pacific DPS migrate through the EEZs of at least 32 nations, spending between 45 and 78 percent of the year on the high seas including in the U.S. EEZs of California and Hawaii (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 specific foraging areas, likely as a result of their nesting season (Benson *et al.* 2011; Harrison *et al.* 2018) and post-hatchling dispersal patterns (Gaspar *et al.* 2012; Gaspar and Lalire 2017).

Leatherback turtles of the West Pacific DPS nest in tropical and subtropical latitudes primarily in Indonesia, Papua New Guinea, and Solomon Islands, and to a lesser extent in Vanuatu (Figure 39; 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), and relatively new sites in the Solomon Islands (Jino *et al.* 2018; TNC-Solomon Islands 2018 unpublished) 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).

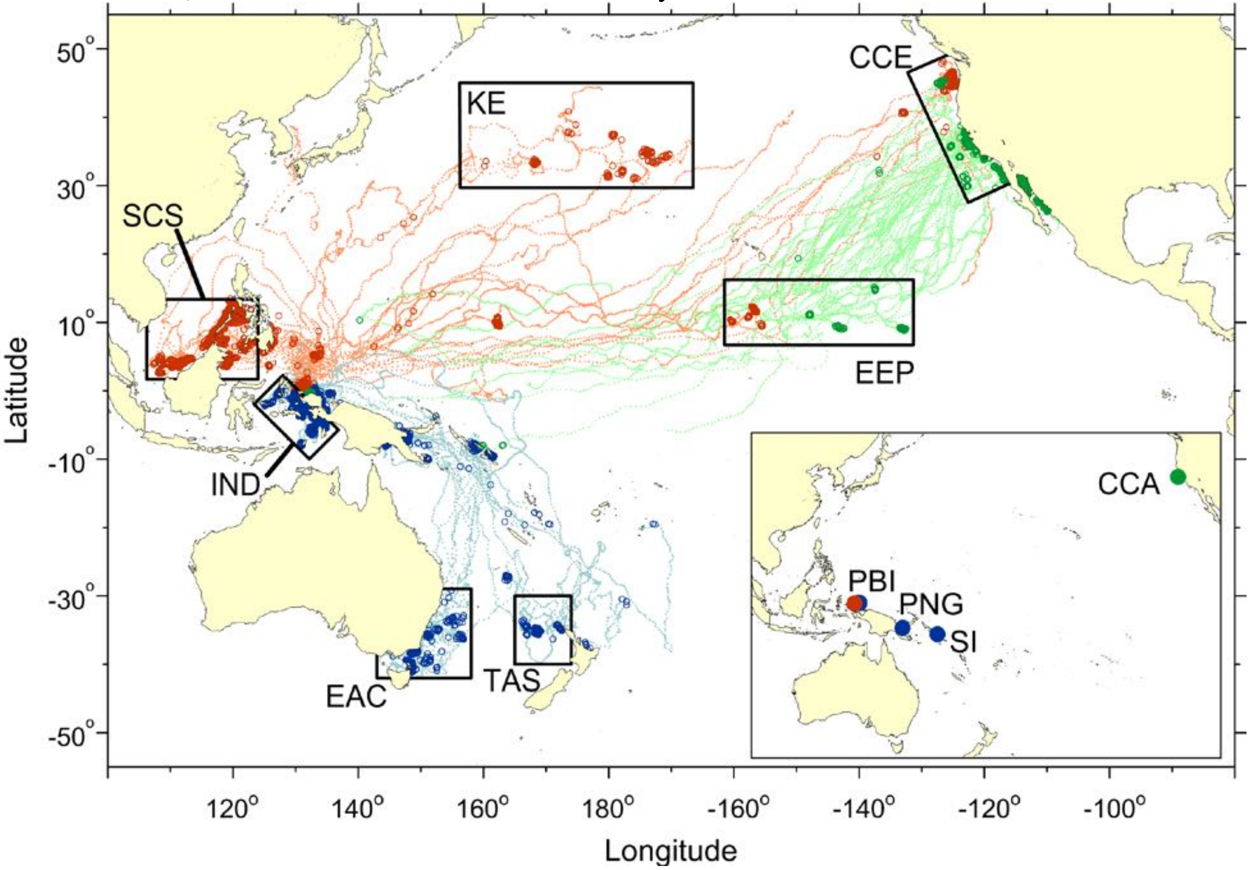
Figure 39. Nesting sites of the West Pacific DPS. Size of circle represents the index of nesting female abundance based on current and outdated (yet best available) information. An "X" indicates that nesting was documented, or suspected, but not quantified.



Nesting beaches 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; Benson *et al.* 2007b; Benson *et al.* 2011; Harrison *et al.* 2018; Figure 40). 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 SWFSC unpublished). It is unclear what proportion of the West Pacific DPS might use this area and how important it might be to this DPS.

Figure 40. West Pacific leatherback turtles, satellite tracked from nesting beaches or at sea. Figure 1 from Benson *et al.* (2011), used with permission from S. Benson (2018) showing 126 satellite tag deployments presented as probability of transit. Large, darker circles indicate Area Restricted Search behavior; small, lighter dots indicate transiting behavior. Color of track indicates deployment season: red = summer nesting females, blue = winter nesting females, green = deployments at central California foraging grounds. Inset shows deployment locations: PBI = Papua Barat, Indonesia, PNG = Papua New Guinea, SI = Solomon Islands, CCA = central California. Black boxes represent ecoregions for which habitat associations were quantitatively examined: SCS = South China, Sulu and Sulawesi Seas, IND = Indonesian Seas, EAC = East Australia Current Extension, TAS = Tasman Front, KE = Kuroshio Extension, EEP = equatorial eastern Pacific, and CCE = California Current Ecosystem.



Summer post-nesting females from Indonesia and Solomon Islands (and likely from Papua New Guinea although data are currently lacking) migrate in one of three predominant directions: westward through various passes into the Indonesian, Sulu and Sulawesi, and South China Seas; eastward along the equatorial currents; or northeastward into the Kuroshio Extension Current (or North Pacific Transition Zone) heading towards the central California ecoregion (Benson *et al.* 2007a; Benson *et al.* 2011; Lontoh 2014; Harrison *et al.* 2018; Jino *et al.* 2018; TNC-Solomon 2018, unpublished). Research to understand foraging dynamics off the central California coast has been ongoing since 1995 (Benson *et al.* 2007c; Benson *et al.* 2018a). Based on simulated modeling of oceanic currents and habitat-driven movements, Gaspar and Lalire (2017) hypothesize that juveniles migrating across the Pacific may reach sexual maturity after 15 years, the mean age at which active turtles reach the California ecoregion. Approximately 30 to 60 percent of Jamursba-Medi summer nesting females (n=78 in 2007 and 2010) foraged in waters off California, in the North East Pacific Ocean (Seminoff *et al.* 2012). Lontoh (2014) sampled additional Jamursba-Medi nesting turtles in 2011 resulting in a sample size of 207 leatherback turtles, demonstrating that the foraging ground composition differed between nesting seasons. Stable isotope analysis combined with satellite telemetry found that animals sampled in 2010 foraged largely within the North East Pacific Ocean and North Pacific Transition Zone (proportions of 48 and 38 percent, respectively), whereas the South China Sea was dominant in 2011 (43 percent) with other animals (roughly 30 percent each) utilizing the North Pacific Transition Zone and North East Pacific Ocean (Lontoh 2014; Seminoff *et al.* 2012).

There are also connections between the North East Pacific Ocean and the Solomon Islands; for example, one satellite tracked female tagged in California in September 2007 migrated to Santa Isabel Island, Solomon Islands to nest during the summer of 2008 (TNC-Solomon Islands 2018 unpublished; S. Benson, NMFS, pers. comm., 2018). Once in their foraging habitats, West Pacific leatherback turtles do not appear to undertake systematic seasonal movements, and some individuals may remain virtually ‘stationary’ for many months, including those in the central California ecoregion and adjacent to the Kei Islands, Indonesia which was occupied year-round (Benson *et al.* 2011).

Leatherback turtles migrate through and forage in the waters of the Philippines (Benson *et al.* 2007a; Benson *et al.* 2011). In 2005, Salinas *et al.* (2009) found a female in San Fernando (close to El Nido, Philippines) that had been previously tagged at Jamursba-Medi in July 2003. The Marine Research Foundation (MRF) used aerial transects to survey foraging areas 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 25 reports compiled by the local NGO, Marine Wildlife Watch of the Philippines, from 2010 to 2018 (Bagarinao 2011; Cruz 2006; MRF 2010; MWWP 2018 unpublished).

10.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the West Pacific DPS.

10.1.1 Abundance

We estimated the total index of nesting female abundance of the West Pacific DPS to be 1,277 females. We based this index on the data summarized in Table 22 and Table 23 and explained in detail below. This number represents an index of nesting female abundance for this DPS because it only includes available data from recently (as of 2014) and consistently monitored (over the remigration interval) nesting beaches: Jamursba-Medi and Wermon, Indonesia. It does not include nesting females from other beaches of Indonesia, Papua New Guinea, Solomon Islands or Vanuatu because these areas have not been consistently monitored for nesting in recent years.

However, these locations may host 25 to 50 percent of nests (see Table 23). Therefore, actual nesting female abundance could be higher, given the potential for unidentified or unmonitored nesting beaches.

Table 22. Available nesting data for the West Pacific DPS. Number of nests (or other units, as identified) for the first and last year of data received represents the range of monitoring years for continuous datasets. We also include the highest and lowest number of nests (or other units, as identified). When recent data were available, we calculated the index of nesting female abundance using the best available data referenced as follows: summing the number of nests over the most recent remigration interval (i.e., 3 years 2015–2017; see Table 24) divided by the clutch frequency (5.5 clutches per season as per Tapilatu *et al.* 2013; see Table 24). When recent data were not available, we reported the number (or range) of nests based on historic information. In these cases, available information is included even if it does not fit the above criteria in an effort to provide some measure of nesting abundance. These numbers are represented in Figure 39 but are not included in Table 23 to calculate the total index of nesting female abundance.

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Indonesia				1,277
Jamursba-Medi, 18km	4,000+ (1981) 1,354 (2017)	High: 4,000+ (1981) Low: 781 (2013)	Tapilatu *et al.* 2013; Tiwari *et al.* in prep.	(1113+1510+1354)/ 5.5 = 723
Wermon, 6km	2,994 (2002) 860 (2017)	High: 2,994 (2002) Low: 650 (2012)	Tapilatu *et al.* 2013; Tiwari *et al.* in prep.	(869+1318+860)/5.5 = 554
Manokwari region, 4 sites	131 (2008) 116 (2011)	High: 135 (2009) Low: 84 (2010)	Suganuma 2012	Unquantified
Buru Island, 10km	203 (2017)	N/A	WWF 2018	Unquantified
Inggrasau Yapen (Japen Island)	Suspected nesting		Tapilatu *et al.* 2017	Unquantified
Wewe Koor	Suspected nesting		Dutton *et al.* 2007	Unquantified
Papua New Guinea				

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
Huon Coast, monitoring effort/distance inconsistent (1 to 7 sites = 3.2 to 35km	249 (2005) 199 (2012)	High: 527 (2010) Low: 193 (2011)	Pilcher 2013; WPRFMC 2015	Unquantified
Bougainville Island	Aerial surveys (2005–2007)	High: 160 (estimated); Low: 68 (range 41–107)	Dutton *et al.* 2007; Benson *et al.* 2007b	Unquantified
New Britain Island	Aerial surveys (January 2004 and 2007)	140 (estimated)	Dutton *et al.* 2007	Unquantified
Madang Province	Aerial surveys (January 2004 and 2007)	50 (estimated)	Benson *et al.* 2007b	Unquantified
Solomon Islands				
Isabel Island (northwest): Sasakolo (1.5km) and Litogarhira (3km)	1989 to current	High: 650 (2007) Low: 315 (2011)	Dutton *et al.* 2007; Tiwari 2011 unpublished site visit	Unquantified
Isabel Island (southeast): Haveo and Sosolilo	2013 to 2016/2017	High: 53 females (2016) Low: 52 females (2013)	TNC-Solomon Islands 2018 unpublished	Unquantified
Rendova, Western Province	2003 to 2017	High: 235 (2003) Low: 29 (2017)	Pilcher 2010b; TDA 2013; Solomon Islands Community Conservation Partnership 2018 unpublished	Unquantified
Tetapare	2002 to 2012/2013	30–50 nests/yr (estimated);	Goby *et al.* 2010; TDA 2013	Unquantified

Nation, nesting beach, and length	**Number of nests (first and last years monitored)**	**High and low number of nests (year)**	**Reference**	**Index of nesting female abundance**
		44 nests (2012/2013)		
Vangunu Island	June 2011 to July 2014	23 nests and 11 females	Jino *et al.* 2018	Unquantified
Vachu, Choiseul Island		50 (estimated)	Dutton *et al.* 2007	Unquantified
Waisurione, Malaita Island	2014–2015	12 (estimated)	Williams *et al.* 2014	Unquantified
Vanuatu/Other Areas				
Votlo Beach, Epi Island	2002 to 2003; 2008 to 2011; 2014 to 2015	31 to 41 nests/yr (estimated); 15 nests and 3 females tagged (2014/2015)	Petro *et al.* 2007; WSB 2009, 2011, 2016	Unquantified
Bamboo Bay, Malekula Island	Suspected nesting		Dutton *et al.* 2007	Unquantified
Philippines	Random nesting events reported		Cruz 2006; MRF 2010; MWWP 2018	Unquantified

Table 23. The number of nesting sites by index of nesting female abundance. We estimated the total index of nesting female abundance (for the DPS) by summing the indices of nesting female abundance from Table 22. We calculated the percentage at the largest nesting site by dividing that site's index of nesting female abundance (Table 22) by the total index of nesting female abundance for the DPS.

Index of nesting female abundance	**Number of nesting sites**
Unquantified	18
1–10	
11–50	
51–100	
101–500	
501–1,000	2
1,001–5,000	
5,001–10,000	
10,001–20,000	
>20,000	
Total number of sites	20
Total index of nesting female abundance (DPS)	1,277 females
Confidence in total index of nesting female abundance	Moderate (25 to 50 percent of nests were not included in estimate)*
Largest nesting site, percentage of total index	Jamursba-Medi and Wermon Beaches, combined 100 percent

* Sites included in Table 22 but not included in the total index for the DPS because recent data (i.e., since 2014) on nests or females are not available over one migration interval (3 years).

Our total index of nesting female abundance is 1,277 females. 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.* 2013a). The IUCN estimate is higher than ours because it included males. Curtis *et al.* (2015) estimated an annual nesting female abundance of at least 318 females (or 954 nesting females over a 3-year remigration interval). Dutton *et al.* (2007) estimated that 1,113 females may have nested annually, or conservatively 2,700 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). Given the published decline in nesting activity since 2002 (Tapilatu *et al.* 2013), our current estimate is consistent with the declining trend (See 10.1.2 Productivity). We estimate that Bird's Head Peninsula beaches represent 50 to 75 percent of the West Pacific DPS. While our estimated total index may not be accurate because calculations do not take into account nesting activity in areas where recent data are not available (Table 22), it 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 only relatively recently with standardized methods in Papua Barat first implemented in 2002 (Hitipeuw *et al.* 2007; Tapilatu *et al.* 2013). Outside of 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, and 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 foreign 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 the following paragraphs, we summarize available abundance data by nation.

Indonesia

In Indonesia, aerial surveys provided the first indication of leatherback nesting in Papua (i.e., Irian Jaya; Salm 1981). At that time, Salm (1981) 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 (a complex of three beaches that span 18 km), and Wermon, a 6 km beach approximately 30 km east of Jamursba-Medi where a total of 1,371 nesting females have been tagged between 2002 and 2011.

The primary nesting season at Jamursba-Medi occurs during the summer from 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; 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 use 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). Historically, approximately 60 percent of nesting activity occurred at Jamursba-Medi and 40 percent of activity at Wermon (Tapilatu *et al.* 2013), and current nesting activity remains proportionally similar (Table 22).

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 if interchange exists 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. 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 *et al.* 2017).

Papua New Guinea

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 have been 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, 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 satellite tagged animals seen visiting a number of areas during one nesting season (Benson *et al.* 2007b). For example, several Huon Coast nesting females visited other nearby beaches and at 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 (Spring 1982; Prichard 1982; Dutton *et al.* 2007; Benson *et al.* 2007b). 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 (Dutton *et al.* 2007; Spring 1982). Approximately 50 nests may be laid annually along the north coast of the Madang Province (TIRN 2017; Benson *et al.* 2007b). 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).

Solomon Islands

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 (Dutton *et al.* 2007; Benson *et al.* 2018b; Pita 2005). 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-Solomons 2018 unpublished). Leatherback turtle monitoring began 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 population 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.* 2018b). 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).

The Tetepare Descendants' Association (TDA) has monitored nesting activity opportunistically since 2002 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 remain key challenges 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 resulted in 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).

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 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); 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).

Vanuatu and Other Pacific Areas

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 inwater 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 (Dobb 2002; Limpus 2009); however, current information was not available for this Report, 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 nest 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 (i.e., a Facebook post), the female was tagged and led back to sea (MWWP 2018 unpublished). Given the low site fidelity of this DPS (Benson *et al.* 2007b), it is not surprising that leatherback turtles might distribute nests among various areas throughout the region.

The index of nesting female abundance of the West Pacific DPS (n = 1,277) places it at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017a). 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. Low site fidelity and dispersal of nests among various beaches may help to reduce population level impacts from threats which may disproportionately affect one area over another. However, due to its small size, the DPS has restricted capacity to buffer such losses. The nesting female abundance is likely an indicator of past and current threats, which we describe later in the Report. Given the intrinsic problems of small population size, we conclude that nesting female abundance is a major factor in the extinction risk of this DPS.

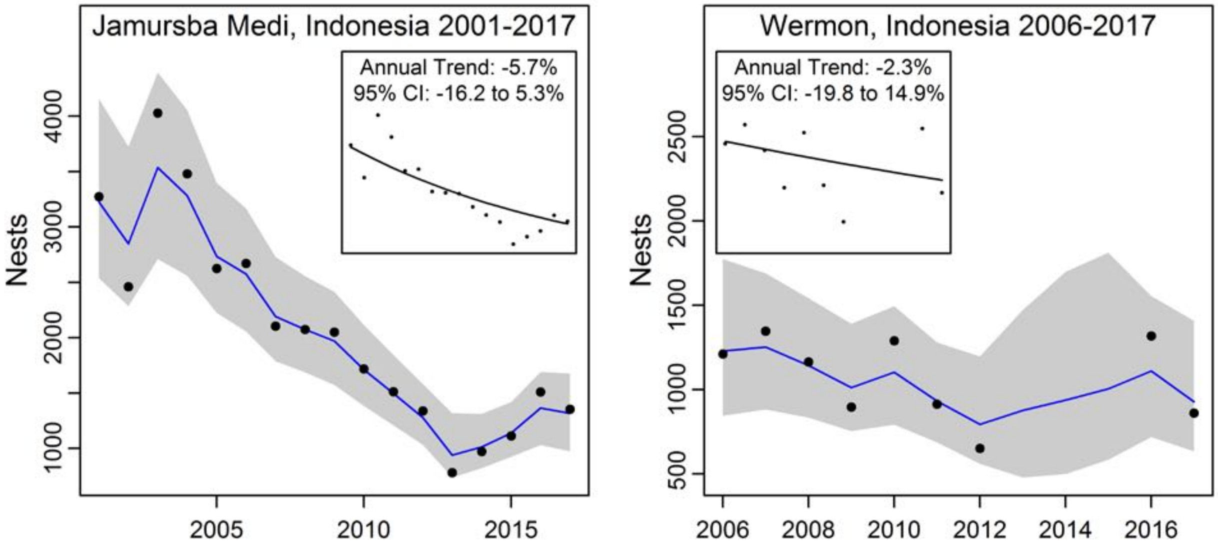
10.1.2 Productivity

The West Pacific DPS exhibits a declining nest trend. Long-term monitoring data for this DPS are geographically limited to the Bird’s Head Peninsula in West Papua, Indonesia and the Huon Coast of Papua New Guinea. 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). Data collection in Papua New Guinea spanned 8 years and ended prior to 2014; therefore, we produced a bar graph rather than a trend for the Huon Coast. As with all DPSs, we report the BSSM trend analysis results as the median and CI, which reflects that there is a 95 percent chance that the trend falls between the low and high CI values. The wider the CI, the less confident we are in the estimated median trend. The higher the “f statistic” the more confident we are in the sign (positive or negative) of the estimated median trend.

Indonesia

Only two beaches (Jamursba-Medi and Wermon) met our standards for trend analyses (Figure 41). 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 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 2013–2015 due to low or insufficient effort; data from 2002 to 2005 were not included in this analysis due to 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 likely represent 50 to 75 percent of nesting for this DPS, we consider these declining trends to be representative of the entire DPS.

Figure 41. Nest trends at Jamursba-Medi and Wermon, Indonesia. The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (nests). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the interannual variability.



Our trend data for Indonesia yield similar results to other published findings. The IUCN Red List found a decreasing trend of 7 percent (Tiwari *et al.* 2013a). Tapilatu *et al.* (2013) identified a 5.5 percent annual rate of decline at Jamursba-Medi between 1984 and 2011 and an 11.6 percent annual rate of decline at Wermon between 2002 and 2011. This population decline is also evident in a foraging habitat. A 27-year aerial survey study indicates a decline in the number of leatherback turtles foraging off central California (Benson *et al.* 2018a). From 1995 to 2003, an estimated 12 to 379 individuals (mean = 178) foraged off California (Benson *et al.* 2007c); however, from 2004 to 2017, an estimated 23 to 112 individuals foraged in this same area, representing a decline of 5.6 percent annually (Benson *et al.* 2018a).

At Jamursba-Medi, nesting data have been collected 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, 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 identify alternative livelihood strategies and monitor nesting activity. While the overall nest trend has continued to decline, there appears to be an increase in nesting since 2013 (Tiwari *et al.* in prep; Figure 47). Continued monitoring over the next 5 years will be critical to determine if declining trends have reversed.

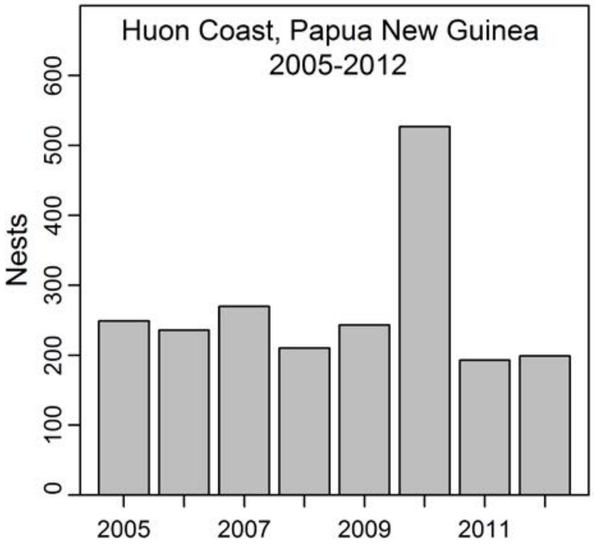
The first comprehensive surveys at Wermon beach in 2002 found almost as many nests laid on Wermon as on Jamursba-Medi (Hitipeuw and Maturbong 2002). At that time, it was hypothesized that the decline at Jamursba-Medi may have been offset by an increase at Wermon (Hitipeuw *et al.* 2007). However, Tapilatu *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; however, similar to Jamursba-Medi there may have been a slight increase in recent nesting activity (Tiwari *et al.* in prep; Figure 47).

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 (Tapilatu *et al.* 2017).

Papua New Guinea

We consider the Huon Coast to be one nesting area and not individual nesting beaches due to the exchange of females and evidence of multiple beach use among females in Papua New Guinea (Benson *et al.* 2007b). Because we did not have sufficient, consistent data from the Huon Coast for a trend analysis, we provide a bar graph of standardized data collected for eight nesting seasons (Figure 42). Within the Huon Coast, there are seven sites which provide approximately 35 km of nesting beach habitat. Combined, these Huon Coast beaches do not exhibit an apparent trend in the bar graph of annual nest counts (mean annual nests = 258) over the 8-year data collection period (2005/2006 to 2012/2013). However, the data were uncertain for Kamiali Beach during the 2005/2006 season, and monitoring efforts were incomplete (to an unknown degree) in 2011/2012 and 2012/2013.

Figure 42. Bar graph of nest data from the Huon Coast, Papua New Guinea. Nesting activity occurs during the winter months November to February. Hence, data recorded in 2005 were collected from November 2005 to February 2006, and so forth for subsequent years.



Nesting activity along the Huon Coast was relatively stable with an average of 258 nests laid per year (range: 193 to 527) between 2005 and 2012 (Pilcher 2013; Benson *et al.* 2015; WPRFMC 2015). 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 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 is approximately 30 km south of the city of Lae. In 1996, the Kamiali Wildlife Management Area was declared a leatherback turtle protected area, 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 longest running, most consistently monitored nesting beach within the Huon Coast nesting beach complex with 194 females 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 at Kamiala and Busama Beaches, villagers and historic information indicates that leatherback nesting activity has declined and was significantly greater in past decades (Benson *et al.* 2007b, 2015; Hirth *et al.* 1993; Kinch 2006; Bellagio Sea Turtle Conservation Initiative, 2008).

Solomon Islands
In the Solomon Islands, it is not possible to estimate nest trends or provide a bar graph 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, and for at least 9 years for trend analysis). 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.* 2018b; Jino *et al.* 2018). However, due to the remoteness of these areas and lack of systematic surveys, and likely numerous 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 Vangunu 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 nearby 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).

Vanuatu
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 2014/1015 nesting season, three females laid 15 nests (WSB 2011, 2016). Again, it is not possible to estimate nest trends or provide a bar graph 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, and for at least 9 years for trend analysis).

In addition to an overall declining nest trend, the West Pacific DPS exhibits low hatching success, due in-part to a combination of past and current threats (i.e., beach erosion, predation, and beach temperatures) described in 10.2 ESA Section 4(a)(1) Factors. Other productivity parameters (with the exception of annual female survivorship) appear to be similar to species' averages (Table 24).

Table 24. Productivity parameters for the West Pacific DPS.

Productivity	**Variable by nation**	**Reference**
Size of nesting female	Papua New Guinea: 160.1 cm CCL (range 150 – 190 cm); 169.5 cm CCL (range 155 – 186 cm)	Pilcher 2009; Hirth 1993
	Indonesia: 161 cm CCL (range 145 – 178 cm); 160.8 ± 0.9 cm CCL for Northeast Pacific foragers; 156.9 ± 1.0 cm CCL for North Pacific Transition Zone foragers; 156.3 ± 1.0 cm CCL for South China Sea foragers	Hitipeuw and Maturbong 2002; Lontoh 2014
Female survivorship (percent)	Papua New Guinea: 0.85 (95 percent CI: 0.66 – 0.95)	Pilcher and Chaloupka 2013
Remigration Interval (years)	Papua New Guinea: 3 (range 1–7)	Pilcher 2010
	Indonesia: 2 – 4	Lontoh 2014
Clutch size (eggs)	Papua New Guinea: 94 (SD 18.1)	Pilcher 2011
	Indonesia, Jamursba-Medi: 79.6 (SD 16.3) n=48; Wermon: 76.2 (SD 16.1) n=51	Tapilatu and Tiwari 2007
	Solomon Islands: 94.6	Jino *et al.* 2018
Clutch frequency (nests per season)	Papua New Guinea: 2 – 5	Pilcher 2011
	Indonesia: 5.5 (SD 1.6)	Tapilatu *et al.* 2013
	Solomon Islands: 5–6	Jino *et al.* 2018
Internesting interval (days)	Papua New Guinea: 11; 15.2 (computed for 20 renesting occurrences)	Benson *et al.* 2007b; Pilcher 2009

Productivity	**Variable by nation**	**Reference**
	Indonesia: 9	Lontoh 2014
Incubation period (days)	Papua New Guinea: 58.2 (SD 5.85)	Pilcher 2007
	Indonesia: 61.5 (SD 4.7)	Tapilatu and Tiwari 2007
Hatching success (percent)	Papua New Guinea: ~60 (project dependent)	Pilcher 2011
	Indonesia, Jamursba-Medi: mean = 25.5 (SD = 32, range = 0 to 85, n = 48); Wermon: mean = 47.1 (SD = 23.6, range = 3.8 to 100, n = 52)	Tapilatu and Tiwari 2007
Sex ratio	California foraging habitat: 3:1 female: male Papua New Guinea: Male-biased sex ratios (variable); Indonesia: Female-biased sex ratios, with some areas of beach possibly producing both males and females	Benson *et al.* 2011 Steckenreuter *et al.* 2010; Pilcher 2010 Tapilatu *et al.* 2013b

Population demographics for leatherback turtles occurring in Indonesia and Papua New Guinea are similar, with the mean size of nesting females around 160 cm CCL (although there is variability based on foraging dynamics), and a remigration interval between 2 to 4 years. However, the remigration interval can range widely (1 to 7 years or longer) likely due to lack of spatial monitoring coverage throughout the vast and remote region, environmental conditions, or low site fidelity (reviewed by Eckert *et al.* 2012; Benson *et al.* 2007b). Benson *et al.* (2011) and Lontoh (2014) found that variation in body size and reproductive output is associated with foraging region, which is likely a reflection of productivity and energetic costs associated with migration. For example, leatherback turtles that foraged in the northeast Pacific Ocean exhibited greater body size and longer remigration intervals than those that foraged in the South China Seas or North Pacific Transition Zone. Further, leatherback turtles that foraged in the North Pacific Transition Zone laid more clutches and had shorter remigration intervals than others (Lontoh 2014).

In Indonesia, Tapilatu *et al.* (2013) estimated an average clutch frequency of 5.5 ± 1.6 nests per female. Although clutch estimates per female are lower in Papua New Guinea than Indonesia, this may be due to inconsistent monitoring effort or nesting at unmonitored beaches (Benson *et al.* 2007b; Pilcher 2009). Female survival has only been estimated for leatherback turtles nesting at Kamiali Beach, Papua New Guinea. At Kamiali, the female annual survival probability was 85 percent and constant over a 10-year (2000 to 2009) mark-recapture study (Pilcher and Chaloupka

2013). This rate is lower than those estimated for two Atlantic rookeries in St. Croix and French Guiana (0.893 and 0.91, respectively; Dutton *et al.* 2005; Rivalan *et al.* 2005), possibly due to greater anthropogenic impacts or lower site fidelity (Pilcher and Chaloupka 2013).

The DPS exhibits 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.* 2013a). Overall, we have moderate confidence in productivity and trend for this DPS: while multiple sources identify long-term or historic declines, inconsistent data collection prevents high confidence of current levels of decline at all nesting beaches. However, the bulk of information points to substantial declines across the DPS over the long term. The decline may reflect past and current threats that exceed the population's productivity metrics. A population growth rate below replacement levels would further reduce nesting female abundance, even if the threats remained constant; increasing or additional threats would further worsen this scenario. We conclude that the declining nest trend and low reproductive output place the DPS at elevated extinction risk.

10.1.3 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 (Benson *et al.* 2011), 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 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 which lead them to foraging and developmental habitats (e.g., the North Pacific Transition Zone); they 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 foraging habitats in Asia and the Central North Pacific Ocean, while winter nesting females migrate to 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 (e.g., Sulu Sulawesi and South China Seas) in more tropical habitats 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 *et al.* 2017; NMFS SWFSC unpublished data).

The wide distribution and variance in foraging strategies likely buffers the DPS 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.

10.1.4 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 provides 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 2017a). 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 to only a portion of the DPS. We conclude that diversity within the DPS provides it with some resilience to threats.

10.2 ESA Section 4(a)(1) Factors

The following sections describe and assess the 4(a)(1) factors or threats. For each, we evaluate the best available information on the threat. We also describe, if possible, the exposure and impact of each threat.

10.2.1 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 energy 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 (Warmamedi 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 becomes a concern.

At Wermon, the inundation of nests from high tides is a major 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 tides (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 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 2017/2018 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 2011/12 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; Jino *et al.* 2018; TDA 2013). 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 are threats, but likely not a major or primary threats to the DPS at this time.

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

The primary threat to the West Pacific DPS is the legal and illegal harvest 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 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 the ability of villagers to internalize the foreign concept of conservation, or "wise" use, due to perceived traditional rights (Kinch 2006; Gjersten and Pakiding 2012; von Essen *et al.* 2014). Additionally, there are nuances related to indigenous harvest (and the definition thereof), which may not be prohibited in these nations.

Turtle poaching includes nesting females on beaches and turtles in their foraging areas and has been documented in all four countries where this DPS nests (Bellagio Sea Turtle Conservation Initiative, 2008; Jino *et al.* 2018; Kinch 2009; Petro *et al.* 2007; Suarez and Starbird 1996; Tiwari *et al.* 2013a; WWF 2018). Egg poaching is a well-documented past and current threat and is prolific throughout the range of the DPS (NMFS and USFWS 2013; Bellagio Sea Turtle Conservation Initiative 2008; Tiwari *et al.* 2013a; Tapilatu *et al.* 2017).

In Indonesia, the poaching of turtles and eggs continues, 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 the 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 up to 100 individuals 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 enumerator 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), with 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 the Kei Islands, three mtDNA samples were assigned to Birds Head Indonesian region, and the fourth sample was not definitive (66 percent probability to Indonesia, 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). Additionally, one female was documented migrating from Wermon Beach to Kei Kecil in March 2007; she was seen but not taken because she was deemed to be on a "special mission" due to her satellite transmitter (Hitipeuw and Lawalata 2008).

In Papua New Guinea, egg and turtle 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; Pritchard 1979; Spring 1982; 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 increased from zero to approximately 60 percent (Pilcher 2009; 2011; 2013; WPRFMC 2015). However, 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 Project, pers. comm., 2019).

In Vanuatu and the Solomon Islands, the poaching of females and collection of eggs is also well documented (NMFS and USFWS 2013; Bellagio Sea Turtle Conservation Initiative, 2008). 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 when encountered (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). Historically, nearly all nesting females and eggs were poached on Redova for consumption (Tiwari 2011 unpublished). 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 DPS, reducing current and/or future reproductive potential. Egg harvest reduces productivity; the persistent, and near-exhaustive (at some locations) collection of eggs guarantees that future population recruitment (i.e., nesting female abundance) will be reduced or eliminated. Given the declining nest trend and current index of 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 and is a significant threat to the DPS. We conclude that overutilization is a major, and the primary, threat to the West Pacific DPS, accelerating its risk of extinction.

10.2.3 Disease and Predation

While we could not find any information on disease, 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 (*Varanus salvator)* has been documented with feral pig predation the most detrimental (Bellagio Sea Turtle Conservation Initiative, 2008; Hitipeuw and Maturbongs 2002; Tapilatu and Tiwari 2007). 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). Efforts are needed to reduce and prevent pig predation and have become a programmatic focus (Suganuma 2005; Hitipeuw *et al.* 2007; Tiwari *et al.* in prep). Intensive management efforts at Jamursba-Medi reduced feral pig predation of nests to five percent during the 2016 and 2017 nesting seasons (Tiwari *et al.* in prep).

In Papua New Guinea, predators of eggs include feral dogs, monitor lizards (*Varanus indicus*), and ghost crabs (*Ocypode cordimanus*) (Kinch 2009). Predation 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; Hirth 1993; Kinch 2009). To protect nests, the 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 in this DPS have a clutch size of over 94 eggs and can nest at least two times in a season, 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 major threat to the West Pacific DPS.

10.2.4 Inadequacy of 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, especially given customary marine tenure systems. Lack of enforcement or implementation 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 is not known throughout the communities or perceived to 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), and the protection of all sea turtle species came into effect in 1999 (Government Regulation No. 7 on Preserving Flora and Fauna Species; Zinudin *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 (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 during the months of May through July (Kinch 2006). This is typically the nesting season for hard-shelled sea turtle species, with leatherback turtles nesting 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 of major cities. Kinch (2006) describes the cultural and socio-economic dynamics of Papua New Guinea villagers, who do not recognize foreign or “western” concepts of sustainability, protection, or conservation.

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 (SPREP 2007). 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 of national regulations near impossible to enforce. 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-based 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).

The Western and Central Pacific Fisheries Commission (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 ≤ 10°, 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.* 2013a). Furthermore, 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 methods, even though approximately 20 percent of longline effort consists of shallow sets (Clarke 2017; WCPFC-TCC14 2018). Further, many RFMO members are not meeting the five percent observer coverage requirement resulting in limited bycatch reporting (WCPFC-TCC14-2018).

In summary, regulatory mechanisms exist to protect leatherback turtles and their eggs throughout the range of this DPS. However, the implementation of laws or management measures are inadequate to reduce the threat that they were designed to address due to a lack of enforcement, adherence to regulations, or inclusion of provisions for indigenous harvest. Regulations are also misaligned with established traditional practices and management systems. The implementation of the WCPFC 2008 longline fishery sea turtle conservation measure is inadequate due to its narrow scope and limited compliance. As a result, harvest and international bycatch remain major threats to the DPS. We consider the magnitude of such threats in other sections. In summary, we consider the inadequacy of the regulatory mechanisms to be a threat to the DPS.

10.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

Other threats affect the West Pacific DPS, with fisheries bycatch being a major threat. Additional threats include: pollution; vessel strikes; natural disasters; and climate change.

10.2.5.1 Fisheries Bycatch

Fishery bycatch in coastal and pelagic fisheries is a major threat to this 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. In their global study of sea turtle bycatch, where data were available, Wallace *et al.*

(2013) found that both longline and net gear were high risk, but low bycatch impact for West Pacific leatherback turtles.

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.* 2012a; 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 those turtles 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 and 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 taken 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 following 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 voyages (months) 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 U.S. Hawaii longline fisheries (NMFS 2018).

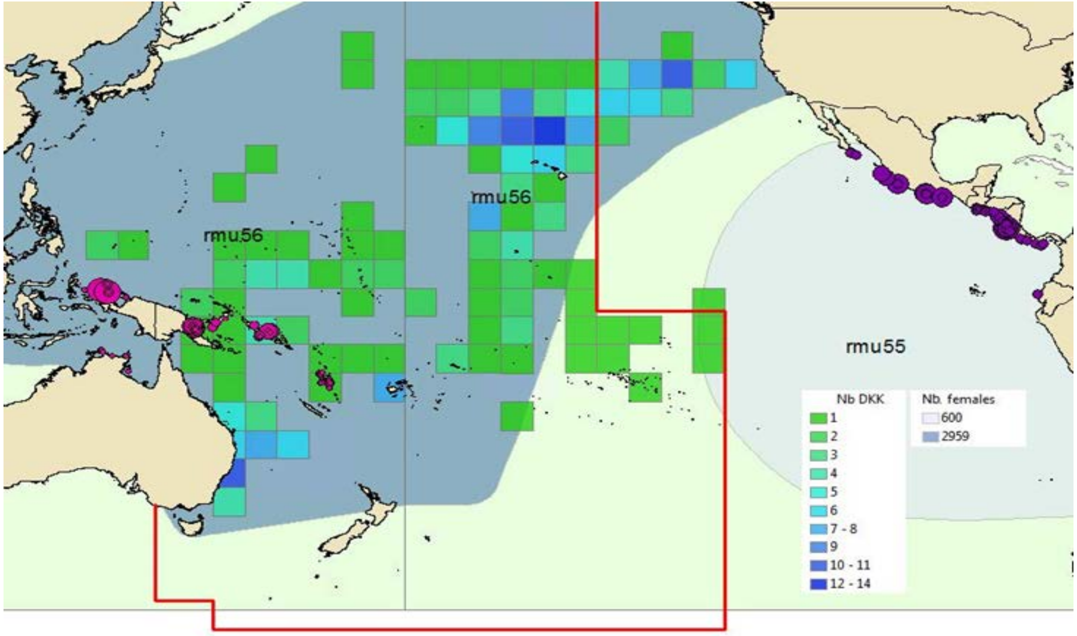
10.2.5.1.1 International Pelagic Fisheries

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 (NMFS 2004; 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 ºN and 31 ºS) 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), or 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), which 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 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 (WCPFC-TCC14-2018). 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 (WCPFC-TCC14-2018).

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 (Figure 43; 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 (Figure 43; 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 just shallow-set swordfish fisheries alone (Clarke 2017).

Figure 43. Leatherback bycatch by 34 Pacific fleets between 1989 and 2015. Figure 1 from Clarke 2017, Western and Central Pacific Fisheries Commission and Pacific Community.



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 bycatch impact 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, Alfaro-Shigueto *et al.* 2011; Peckham *et al.* 2007), 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).

10.2.5.1.2 Southeast Asian Fisheries

Waters of Southeast Asia are heavily fished by a variety of gill nets, 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). Between 2004 and 2007, NMFS provided funding to World Wildlife Fund-Indonesia to support capacity building in the commercial tuna longline fishery to implement an observer program, undertake circle hook trials, and promote turtle-safe handling measures (Zainudin *et al.* 2007). It is unclear if measures supported during that time continue, although Indonesia began taking a more active role in RFMOs and the Indian Ocean Southeast Asian Sea Turtle Memorandum of Understanding (IOSEA MOU), and initiated a national action plan to reduce bycatch in the Indonesian commercial tuna longline fishery in 2005 (Zainudin *et al.* 2007). 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). Recently, NMFS has been collaborating with Indonesia's Ministry of Marine Affairs and Fisheries and local universities to better understand the effects of the region's small-scale fisheries on protected marine megafauna. From 2013 through 2016, rapid assessments (i.e., interview-based surveys) have been conducted in 34 districts, and consisting of over 1,100 fishermen interviews, which characterized fishing vessels, fishing gear, scope of fishing operations, and bycatch rates (e.g. sea turtles, marine mammals, elasmobranchs). An assessment of the gillnet fisheries based in West Kalimantan found that several hundred sea turtles are caught each year in those fisheries (primarily green and olive ridley turtles; WWW 2018). NMFS has been working with Indonesia to expand this work into other coastal gillnet fisheries throughout the Indonesian archipelago with a focus on identifying and reducing leatherback bycatch. Fortunately, cultural and traditional perspectives can be (and perhaps have been) helpful in managing fishery interactions. For example, Indonesian longline fishermen try to minimize interaction with sea turtles as they believe that sea turtles on-board will reduce fish catch (Zainudin *et al.* 2007; WWF 2018). Also, if the vessel's captain is Chinese or Taiwanese, fishermen typically release sea turtles as they believe that sea turtles are divine creatures and must be respected (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 2009).

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 (Stobutzki *et al.* 2006; Robins *et al.* 2002), 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) under 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 turtle 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 ≤ 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; Patterson *et al.* 2013 in Clarke *et al.* 2014).

In New Zealand, there have been 288 stranding and bycatch (commercial and recreational) records 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 overage during the peak time of leatherback abundance in New Zealand waters (January to March). Strandings can also 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 floats 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).

10.2.5.1.3 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.

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 increased data confidence by significantly increasing 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 (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 (see, NMFS regulations at 50 CFR Part 665; 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 Cir. 2017) was arbitrary and capricious. 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; 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), an extrapolation based on 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 new emerging threat with two documented leatherback takes occurring in 2015 and 2016. Dungeness crab fishing effort is high, and the fishery has shifted into the Central California region, which overlaps somewhat with leatherback foraging habitat (S. Benson, NMFS, pers. comm., 2018).

10.2.5.1.4 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.* 2007a, 2011, 2018). Of 59 leatherback turtles caught in East Pacific fisheries, an estimated 15 percent of individuals sampled originated from the West Pacific DPS (NMFS SWFSC unpublished; 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). See also Chapter 11 (Section 11.2.5.1) for additional information and discussion on fishery impacts in the Eastern Pacific.

We conclude that individuals of this DPS are exposed to high fishing effort throughout their foraging range, and likely in coastal waters near nesting beaches or enroute 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, for example in Southeast Asia (WCPFC-SC13-2017; Wallace *et al.* 2013a; Alfaro-Shigueto *et al.* 2011; Lewison *et al.* 2004; Lewison *et al.* 2014). 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; NMFS 2018a; Swimmer *et al.* 2017). Mortality reduces abundance by removing individuals from the population; it also reduces productivity when nesting females are bycaught and killed. While interactions in U.S.-managed fisheries cannot be discounted, we conclude that international fisheries bycatch is a major threat to the West Pacific DPS.

10.2.5.2 Pollution

Pollution includes contaminants, marine debris, and ghost fishing gear. Leatherback turtles can ingest marine 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, however the impact is unquantified.

Leatherback turtles feed exclusively on jellyfish and other gelatinous organisms and as a result may be prone to ingesting plastic items resembling their food source (Schuyler *et al.* 2013, 2015). Lebreton *et al.* (2018) estimated plastic debris accumulation to be at least 79,000 (45,000 to 129,000) tons in the Great Pacific Garbage Patch, a 1.6 million km^2 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. Additionally, Clukey *et al.* (2017) found no plastics in the gastrointestinal tracts of three leatherback carcasses from Pacific longline fisheries captured between 2012 and 2016. It is surprising that no ingested debris was found in these studies given the amount of debris believed to be within the pelagic foraging environment. While this may be a consequence of the small sample sizes, it could indicate minimal marine debris ingestion. Future necropsy and gut content analysis of beach stranded animals would be valuable to help understand or quantify this potential threat.

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 no ingested marine debris. Impacts could include death and injury; however, quantitative estimates of such impacts are not available. We conclude that while pollution may pose a threat to individuals of the DPS, the level of impact is currently unknown based on information available at this time.

10.2.5.3 Vessel Strikes

Vessels strikes pose a threat to the West Pacific DPS. Of leatherback strandings documented in central California between 1981 and 2016, 11 were determined to be the result of vessel strikes (7.3 percent of total; NMFS unpublished data).

The range of the DPS overlaps with many high-density vessel traffic areas and it is possible that the vast majority of vessel strikes are undocumented. However, we were not able to find any information on vessel strikes for other locations. In Hawaii, five leatherback turtles stranded in nearshore coastal waters between 1982 and 1993, but none were attributed to vessel strike (NMFS unpublished data). Leatherback strandings have also been documented in the Philippines, Australia, and New Zealand (Mackay *et al.* 2014; MWWP 2018 unpublished), but again none were directly attributed to vessel strike.

Though there exists potential for exposure, we are only aware of 11 vessel strikes in California. Vessel strikes resulting in mortality would lower the abundance of the DPS. We conclude that boat strikes pose a threat to individuals of the DPS, although the impact to the DPS is currently unknown.

10.2.5.4 Natural Disasters

Natural disasters have the potential to threaten the DPS, although their impact is unquantified. Natural disasters within the range of this DPS include: tsunamis, typhoons, earthquakes, and flash floods. Such natural climatic or 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 a few locations, but leatherback turtles have undoubtedly evolved to sustain such natural impacts and stochastic events which have been ongoing for millions of years. However, 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). In addition, 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 have the potential to pose a threat, but based on available information they are not a current threat to the DPS.

10.2.5.5 Climate Change

Climate change is likely to affect 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 ± 4.2 days, and nests laid in February hatched in 55.8 ± 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 suggest a female-biased population at Jamursba-Medi. However, the mean incubation period of 61.5 ± 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 (2005 to 2012) ranged from 26.5 to 34.9ºC 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 in 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 the DPS. Impacts are likely to affect productivity. As we are already seeing negative impacts and low hatching success due to high beach temperatures and coastal erosion, we conclude that climate change is a threat to the West Pacific DPS.

10.3 Extinction Risk Analysis

After reviewing the best available information, we conclude that the West Pacific DPS has a high extinction risk, as summarized in the following paragraphs. The total index of nesting female abundance is 1,277 females at two currently monitored beaches over the most recent remigration interval. These beaches may represent 50 to 75 percent of total DPS nesting activity. This estimate 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.

Current threats contribute to the high risk of extinction of this DPS, as summarized in Table 25. 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 reduce productivity and contribute to low hatching success. Additional threats include: pollution and marine debris, vessel interactions, and natural disasters. Climate change is an increasing threat that results in reduced productivity; high (lethal) beach incubation temperatures have already resulted in nest failure, which contributes to low hatching success and perhaps has already skewed sex ratios. Though many regulatory mechanisms exist, they are inadequate to sufficiently reduce the threats.

Table 25. Threats to the West Pacific DPS. Exposure refers to the individuals affected by the threat. Impact refers to how the threat affects the demographic factors. The primary threats are identified with asterisks.

Threat	**Exposure**	**Impact**
Destruction or modification of habitat	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Overutilization*	Eggs and nesting females; turtles at sea	Loss of nesting females (abundance) and reproductive potential (productivity)
Predation	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Inadequate regulatory mechanisms	Eggs and turtles of all life stages	Some laws are poorly enforced
Fisheries bycatch*	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) including loss of nesting females (productivity)
Vessel strikes	Foraging juveniles and adults	Loss of individuals (abundance)
Pollution	Turtles of all life stages	Lethal (abundance) and sublethal (productivity) effects
Natural disasters	Some eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Climate change	Eggs and turtles at all life stages	Reduction of nesting and hatching success (productivity)

Thus, we find that the West Pacific DPS is at a high level of extinction risk. Its nesting female abundance and significant declining nest trend contribute to our concern over its continued persistence. The DPS faces several clear and present threats. Past egg and turtle harvest initially reduced the abundance and productivity of this DPS; ongoing harvest pressure continues to inhibit productivity and abundance and therefore remains a primary threat. Fisheries bycatch is also a primary threat that reduces abundance by removing mature and immature individuals from the population. Predation and low hatching success is also a threat to productivity. We conclude that the West Pacific DPS meets the definition for high risk of extinction (see Section 1.1.3.3 Extinction Risk Assessment). The moderate risk definition does not apply because the DPS is at a high risk of extinction now (i.e., at present), rather than on a trajectory to become so in the foreseeable future. We have high confidence in our conclusion of high extinction risk because of the limited abundance, declining trends, and impact of several major threats.

11.0 East Pacific DPS

We define 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 (Figure 44). 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.

Figure 44. East Pacific DPS boundary map.

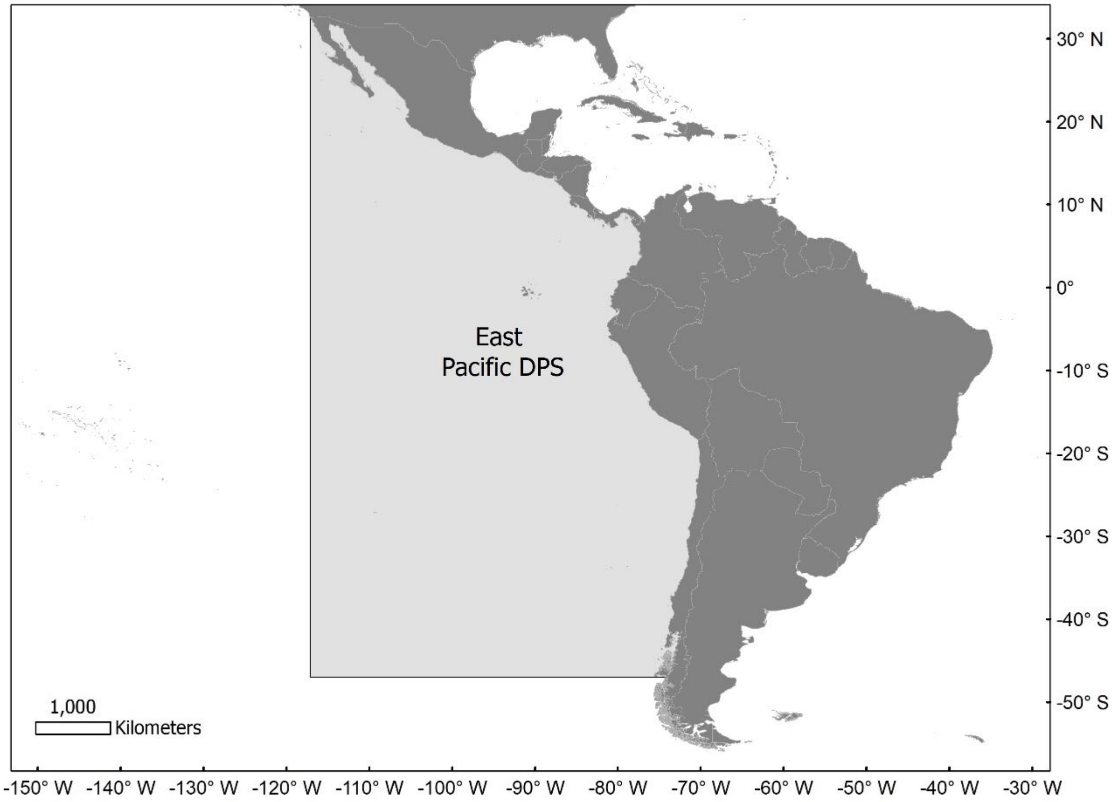
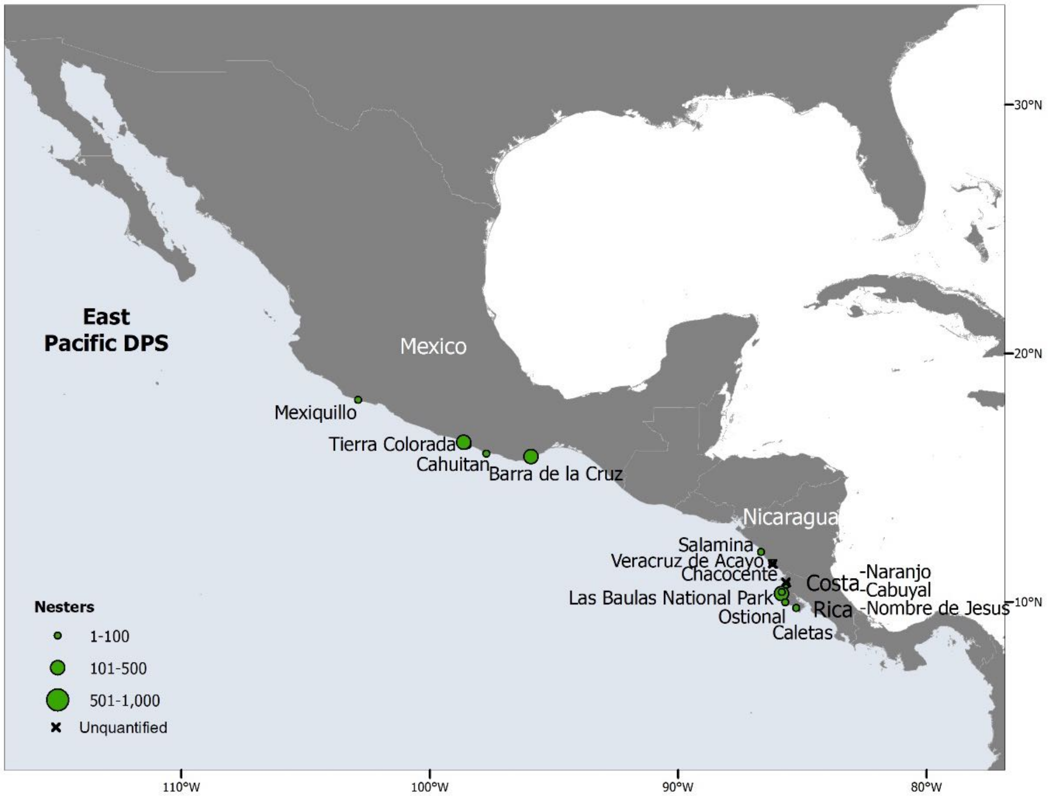


Exhibit 8 to Declaration of Adam Keats

The range of the DPS (i.e., all areas of occurrence) is focused 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 (Figure 6) and two turtles captured by the Hawaii longline fishery, genetically assigned to 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.* 2013b).

Leatherback turtles of the East Pacific DPS nest primarily on beaches in Mexico, Costa Rica, and Nicaragua (Figure 45). 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 70 to 75 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). The smallest nesting females recorded in Mexico have been 120 cm CCL (Sarti *et al.* 2007). 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.* 2017c). In Nicaragua, extremely 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).

Figure 45. Nesting sites of the East Pacific DPS. Size of circle represents the index of nesting female abundance. An "X" indicates that nesting was documented but not quantified.



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.* 2007a,b; Eckert and Sarti 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 and Sarti (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 (Figure 46). Additional tracking of 46 post-nesting females from Las Baulas National Park over a 3-year period (2004/2005–2006/2007) 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 (Figure 47). 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 and Sarti (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.* 2008). 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.

Figure 46. Satellite tracks of seven post-nesting tagged at Mexiquillo from January to September 1997. Figure 2 from Eckert and Sarti (1997; Marine Turtle Newsletter, *http://www.seaturtle.org/mtn/archives/mtn78/mtn78p2.shtml*).

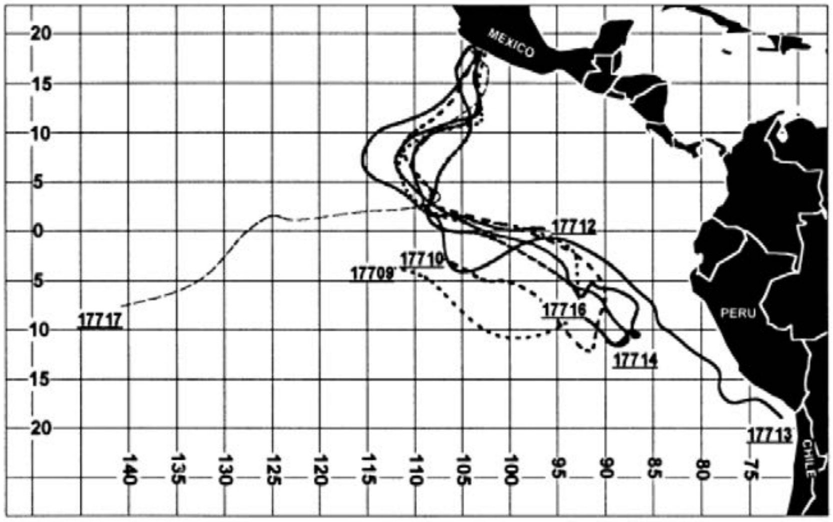
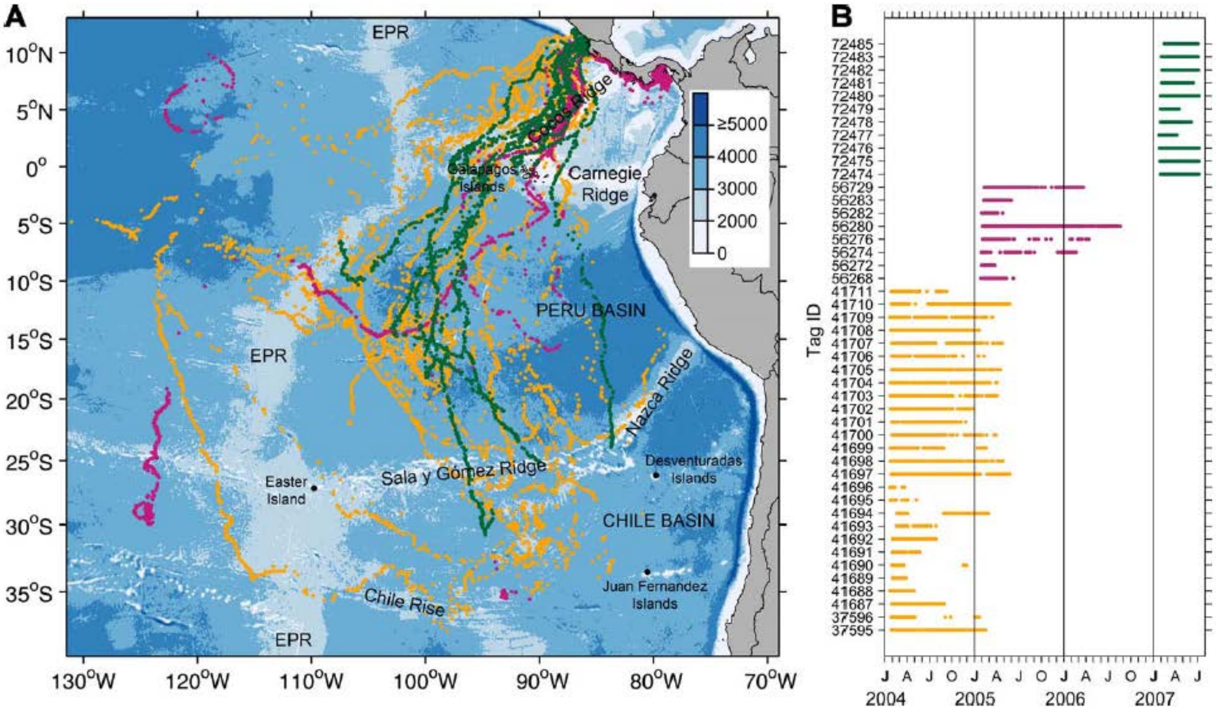


Figure 47. Satellite tracks for 46 post-nesting females tagged at Playa Grande, Costa Rica from 2004 (n = 27, orange), 2005 (n = 8, purple), and 2007 (n = 11, green), overlaid on bathymetry (in meters). B) Timeline of satellite transmissions for each tag. Image: Figure 1 from Shillinger *et al.* (2008), PLoS Biology, *https://doi.org/10.1371/journal.pbio.0060171*.



In summary, preferred foraging areas for the 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 at higher latitudes (Bailey *et al.* 2012b).

11.1 Demographic Factors

In the sections below, we provide information on the abundance, productivity, spatial distribution, and diversity of the East Pacific DPS.

11.1.1 Abundance

For the East Pacific DPS, we estimated the total index of nesting female abundance to be 755 females. We based this total index on the following data, summarized in Table 26 and Table 27 and explained in detail below: monitoring data from Mexico provided by L. Sarti, Mexican Commission for Natural Protected Areas (i.e., Sarti 2018); monitoring data from Nicaragua provided by V. Gadea (FFI 2018); and monitoring data (The Leatherback Trust 2018) and published nesting data from Costa Rica (Santidrián Tomillo *et al.* 2017b). This is an index of nesting female abundance 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.

Table 26. Available nesting data for the East Pacific DPS. For each monitored nesting beach, the number of nests (published estimate of nesting females, in bold) for the first and last year of data received represents the range of monitoring years for continuous datasets. For non-continuous datasets, we identify additional years for which data were available. We also include the highest and lowest number of nests (or other units, as identified). We calculated the index of nesting female abundance by summing the number of nesting females (or nests, when only available) over the most recent remigration interval (i.e., 4 years; see Table 31). For Nicaragua, where only nest data were available, we estimated the index of nesting female abundance by summing the number of nests over the most recent remigration interval (i.e., 4 years); divided by the clutch frequency (i.e., 7.2 clutches per season, the average clutch frequency for Costa Rica, see Table 28; Reina *et al.* 2002b; Wallace *et al.* 2013b).

Nation, nesting beach, and length	Number of nests/nesters (in bold); (first and last years or seasons monitored)	High and low number of nests/nesters (in bold); (year or season)	Reference	Index of nesting female abundance
Mexico				**572**
Mexiquillo, 18 km	41/**15** (1996/1997) 38/**5** (2012/2013)	High: 611/**135** (1999/2000) Low: 20/**4** (2001/2002)	CONANP, 2018	(12 + 46 + 15 + 5) = 78
Playa de Tierra Colorada, 26 km	214/**28** (1996/1997) 305/**64** (2016/2017)	High: 502/**112** (2003/2004) Low: 8/**2** (2002/2003)	CONANP, 2018	(11 + 23 + 22 + 64) = 120
Playa Cahuitán, 12 km	151/**55** (1997/1998) 140/**27** (2016/2017)	High: 413/**114** (2000/2001) Low: 31/**4** (2004/2014)	CONANP, 2018	(20 + 25 + 26 + 27) = 98
Playa San Juan (Chacahua)	33/**10** (2011/2012) 55/**12** (2016/2017)	High: 104/**20** (2015/2016) Low: 29/**3** (2013/2014)	CONANP, 2018	(3 + 11 + 20 + 12) = 46

Nation, nesting beach, and length	**Number of nests/nesters (in bold); (first and last years or seasons monitored)**	**High and low number of nests/nesters (in bold); (year or season)**	**Reference**	**Index of nesting female abundance**
Bahia de Chacahua, 11 km	34/**3** (2013/2014) 40/**11** (2016/2017)	High: 40/**11** (2016/2017) Low: 4/**2** (2014/2015)	CONANP, 2018	(3 + 2 + 5 + 11) = 21
Playa Barra de la Cruz – Playa Grande, 8 km	20/**6** (1996/1997) 240/**61** (2016/2017)	High: 365/**96** (2015/2016) Low: 3/**2** (2002/2003)	CONANP, 2018	(17 + 35 + 96 + 61) = 209
Costa Rica				**165**
Las Baulas National Park: Playa Ventanas (1.0 km), Playa Grande (3.6 km) and Playa Langosta (1.3 km)	**1,504** (1988/1989) **23** (2015/2016)	High: **1,504** (1988/1989) Low: **20** (2014/2015)	The Leatherback Trust, 2018	(36 + 25 + 20 + 23) = 104
Naranjo	106/**18** (1971) 20/**5** (2015)	High: 106/**18** (1971) Low: 1/**1** (2014)	Santidrián Tomillo *et al.* 2017	Unquantified
Cabuyal	1/**2** (2011) 28/**6** (2015)	High: 28/**6** (2015) Low: 1/**2** (2011)	Santidrián Tomillo *et al.* 2017	(5 + 2 + 2 + 6) = 15
Nombre de Jesús	5/**4** (2010) 12/**8** (2015)	High: 19/**12** (2014) Low: 5/**1** (2012)	Santidrián Tomillo *et al.* 2017	(1 + 3 + 12 + 8) = 24
Ostional	19/**2** (2004) 8/**4** (2014)	High: 44/**8** (2009) Low: 19/**2** (2004)	Santidrián Tomillo *et al.* 2017	(4 + 7 + 6 + 4) = 21
Caletas	18/**3** (2002) 9/**1** (2015)	High: 18/**3** (2002) Low: 0/0 (2012/2014)	Santidrián Tomillo *et al.* 2017	(0 + 0 + 0 + 1) = 1
Nicaragua				**18**
Salamina-Costa Grande	15 (2007/2008) 18 (2017/2018)	High: 25 (2014/2015) Low: 17 (2015/2016)	FFI and Quelantaro Reserve, 2018	(25 + 17 + 20 + 18)/7.2 = 11.11 (12)

Nation, nesting beach, and length	**Number of nests/nesters (in bold); (first and last years or seasons monitored)**	**High and low number of nests/nesters (in bold); (year or season)**	**Reference**	**Index of nesting female abundance**
Veracruz de Acayo (Chacocente Wildlife Refuge)	24 (2002/2003) 9 (2017/2018)	High: 84 (2006/2007) Low: 2 (2016/2017)	FFI, 2018	(23 + 5 + 2 + 9)/7.2 = 5.42 (6)
El Salvador Playa Puntilla	6 nests (1999)		Sarti *et al.* 2000	Unquantified
Guatemala	Less than 10 nests (1999)		Sarti *et al.* 2000; Muccio and Flores 2015	Unquantified

*Nesting female abundance for Mexiquillo was only available for 2009 to 2013, due to safety reasons.

Table 27. The number of nesting sites by the index of nesting female abundance. We estimated the index of nesting female abundance by summing the indices of nesting female abundance from Table 26. We calculated the percentage at the largest nesting site by dividing that site's index of nesting female abundance (Table 26) by the index of nesting female abundance for the DPS.

Index of nesting female abundance	**Number of nesting sites**
Unquantified*	3
1–10	2
11–50	6
51–100	2
101–500	3
501–1,000	0
1,001–5,000	0
5,001–10,000	0
>10,000	0
Total number of sites	**16**
Total index of nesting female abundance (DPS)	**755**
Confidence in total index of nesting female abundance	**Moderate since the most recent 4 years of data from Mexiquillo are not available and approximately 30 percent of nesting is unmonitored**
Largest nesting site, percentage of total index	**Playa Barra de la Cruz (Mexico): 28 percent**

*Sites included in Table 26 but not included in the total estimate for the DPS because recent data are not available over one migration interval. These sites may represent additional nesting females in the DPS, but data are outdated, or sites are not consistently monitored.

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.* 2002b) 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.* 2002b; Santidrián Tomillo *et al.* 2007) to obtain a total number of adult females that included nesting as well as non-nesting turtles. This estimate is lower than our total index of nester abundance because of the clutch frequency and methodology used by Wallace *et al.* (2013b). 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.* 2013b). 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 estimate represent approximately 70 to 75 percent of total nesting in that nation (Gaona Pineda and Barragan Rocha 2016); however, our estimate 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.* 2017b), 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 most of the nesting and consistent monitoring. Small numbers of females also nest at Juan Venado National Reserve, which is not consistently monitored (V. Gadea, FFI, pers. comm., 2018).

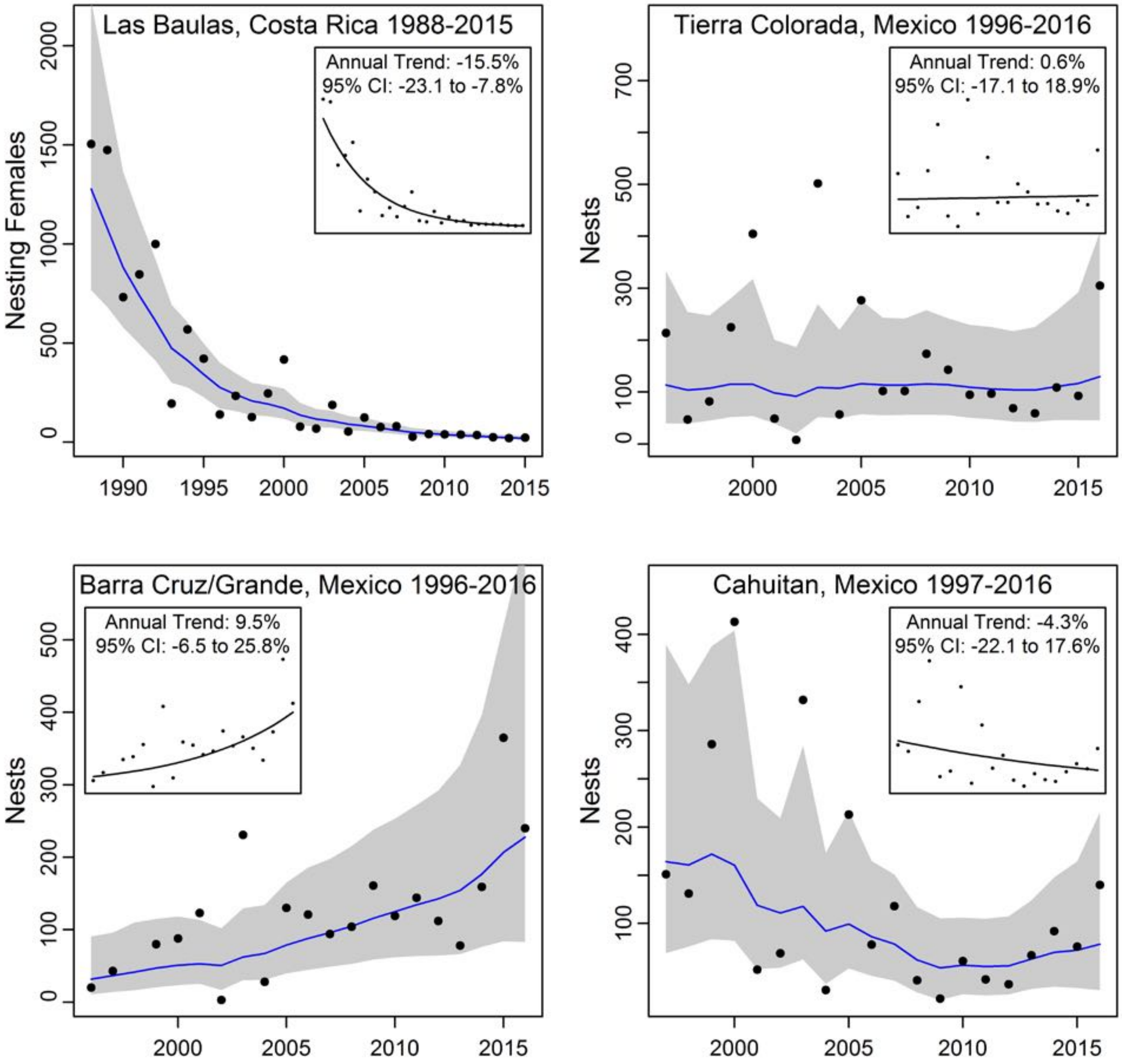
Nesting is rare in other nations (Sarti *et al.* 1999). Nesting is very uncommon in Ecuador with one record of a female attempting to nest (according to local reports) in Atacames, a province of Esmeraldas (Salas 1981). Sarti *et al.* (1999) reported six nests at Playa Puntilla, El Salvador, but overall nesting in that country is low and/or unknown. 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). Recent records are unknown, although important nesting sites were identified to be Hawai beach, La Candelaria, Taxico, Santa Rosa, and the zone adjacent to the border with El Salvador (Chacón and Aráuz 2001), 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 (n = 755) places the DPS at elevated risk for environmental variation, genetic complications, demographic stochasticity, negative ecological feedback, and catastrophes (McElhany *et al.* 2000; NMFS 2017b). 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. 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 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, likely elevates the extinction risk of this DPS.

11.1.2 Productivity

The East Pacific DPS exhibits a decreasing trend since monitoring began, with a 97.4 percent decline since the 1980s or 1990s (depending on nesting beach; Wallace *et al.* 2013b); despite intense conservation efforts, the decline in nesting had not been reversed as of 2011 (Benson *et al.* 2015). Using population viability analysis, the Laúd OPO Network (2020) found that the DPS will be extirpated in less than 60 years under status quo conditions. We found a declining nest trend at some of the remaining, small nesting aggregations (Figure 48). We base our conclusion on nest count data consistently collected in a standardized approach for nine or more years. We find these data to be representative of the DPS because they include the largest nesting beaches. Based on these data, we conclude that this DPS has experienced drastic reductions in overall nesting female abundance. Historical abundance at Las Baulas, Costa Rica (previously the largest nesting site) at its peak was seven times the current abundance at Playa Barra de la Cruz/Playa Grande, Mexico (currently the largest nesting site). As with all DPSs, we report the BSSM trend analysis results as the median and CI, which reflects that there is a 95 percent chance that the trend falls between the low and high CI values. The wider the CI, the less confident we are in the estimated median trend. The higher the "f statistic" the more confident we are in the sign (positive or negative) of the estimated median trend.

Figure 48. Nest trends at East Pacific beaches. The BSSM trend analysis is represented by the blue line (median model prediction) and grey shading (95 percent credible interval). Black dots are original data points (nesting females or nests). Model predicted values are based on estimates for both a long-term trend parameter and an inter-annual variability parameter. Inset plot shows the long-term trend parameter isolated from the inter-annual variability.

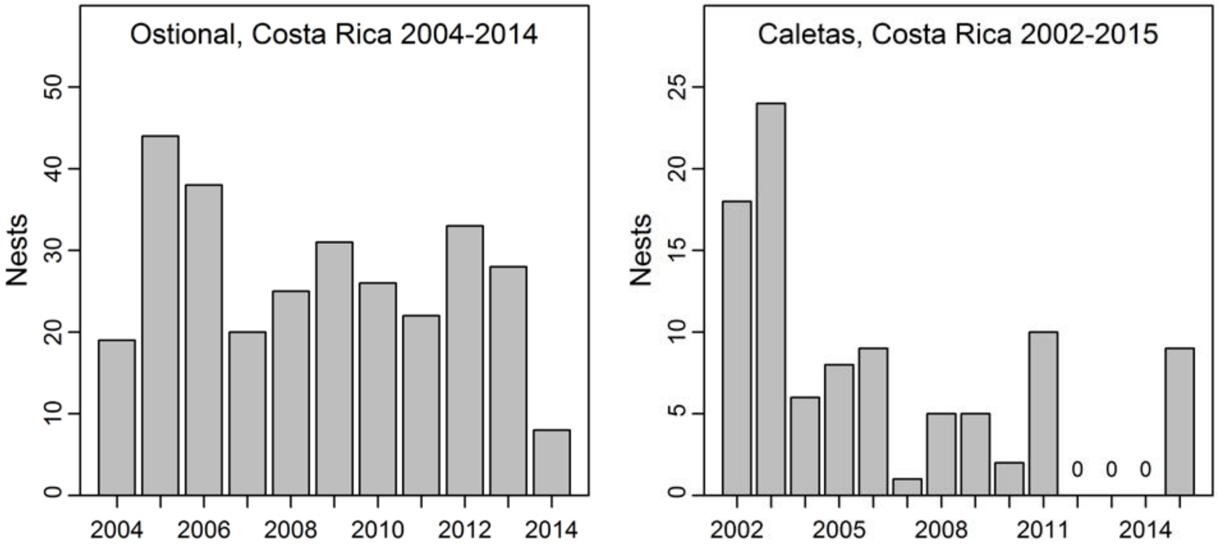


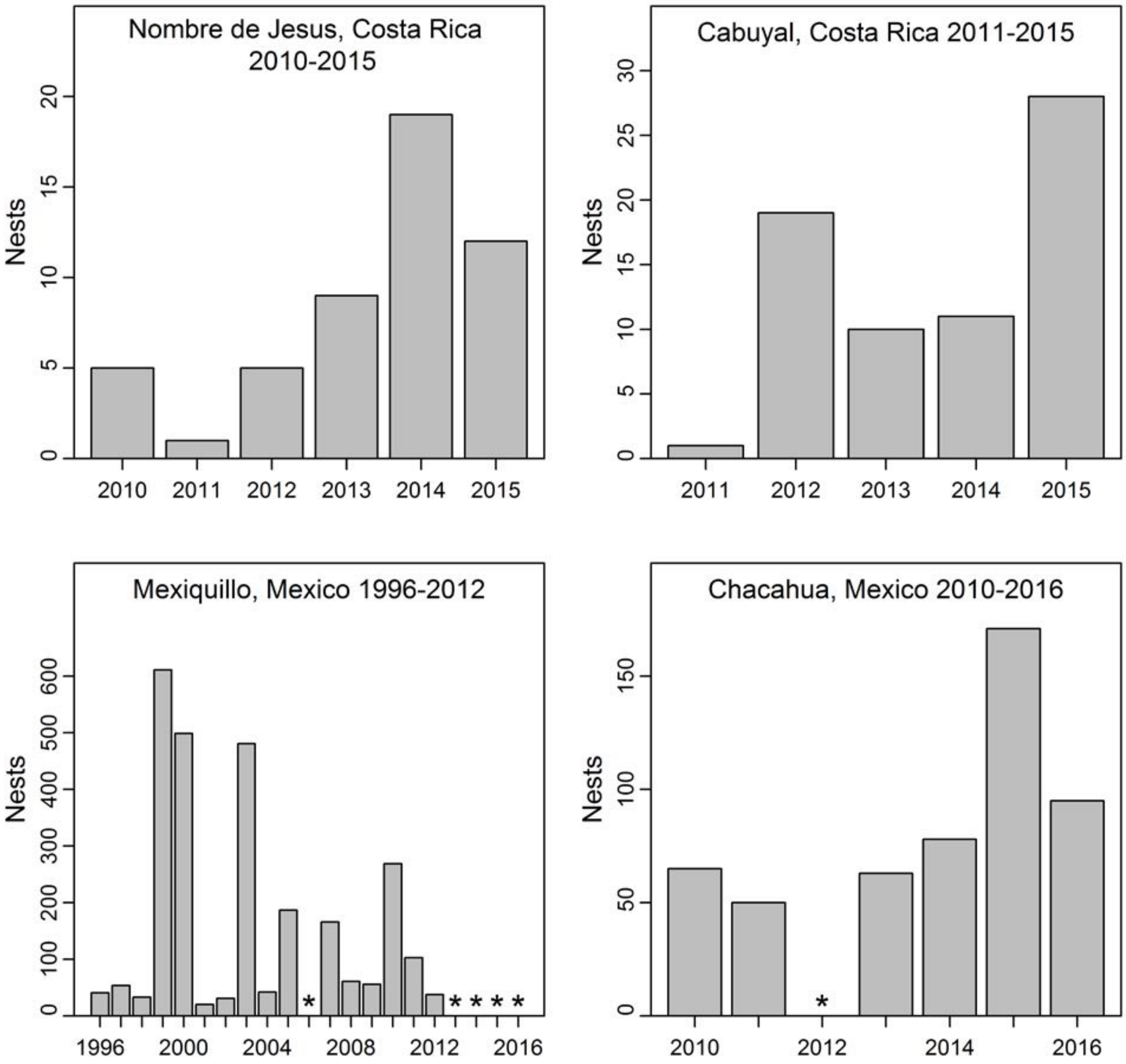
The primary nesting beach complex in Costa Rica, Las Baulas, presents the strongest evidence for decline for this DPS, as it hosts the majority of nesting females in recent decades. From 1988/1989 through 2015/2016, the trend in number of nesting females at Las Baulas was −15.5 percent annually (sd = 3.8 percent; 95 percent CI = −23.1 to −7.8 percent; f = 0.998; mean annual nesting females = 315). Bar graphs for the secondary beaches of Ostional (2004–2014; mean annual nests = 6) and Caletas (2002–2015; mean annual nests = 2) suggest possible slight declines, while bar graphs for Nombre de Jesus (2010–2015; mean annual nests = 5) and Cabuyal (2011–2015; mean annual nests = 3) show no apparent trend (Figure 53); however, all four of those sites have relatively low numbers that may fall within the range of typical inter-annual variability. Data from Naranjo were too intermittent and unreliable to include in this report.

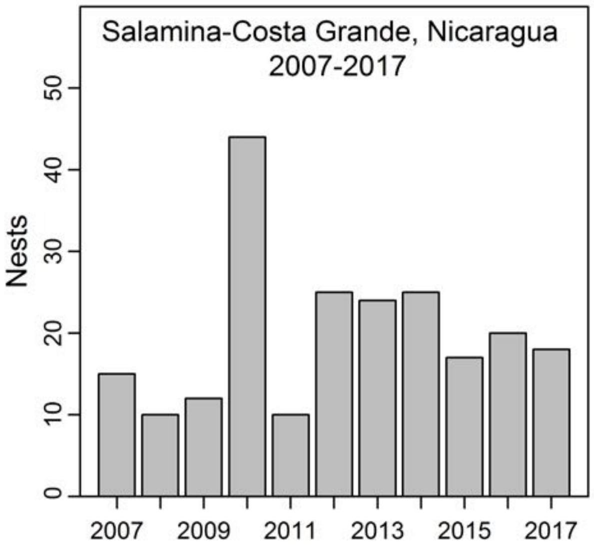
In Mexico, the median trend in annual nest counts estimated for Playa Tierra Colorada was 0.6 percent (sd = 8.9 percent; 95 percent CI = −17.1 to 18.9 percent; f = 0.536; mean annual nests = 153) from the 1996/1997 season through 2016/2017. Over the same time period, Playa Barra de la Cruz /Playa Grande 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). For Mexiquillo, standardized data collection ended prior to 2014 (1996/1997–2012/2013; mean annual nests = 168); thus, we include the data in a bar graph (Figure 53). We also include a bar graph of nest counts from Chacahua (2010/2011–2016/2017, excluding 2012/2013; mean annual nests = 87). Neither bar graph provides clear, compelling evidence for a trend (Figure 54). In Nicaragua, the mean annual nest counts at Salamina-Costa Grande and Veracruz de Acayo were both lower than our 50 nest threshold for trend analysis. Therefore, we provide bar graphs for each site (Figure 53). Data for Salamina-Costa Grande span 2007/2008 through 2017/2018 (mean annual nests = 20), with the bar graph demonstrating no obvious trend. At Veracruz de Acayo, the bar graph may reflect a decline from 2002/2003 to 2017/2018 (mean annual nests = 20); however, there is uncertainty in such an inference.

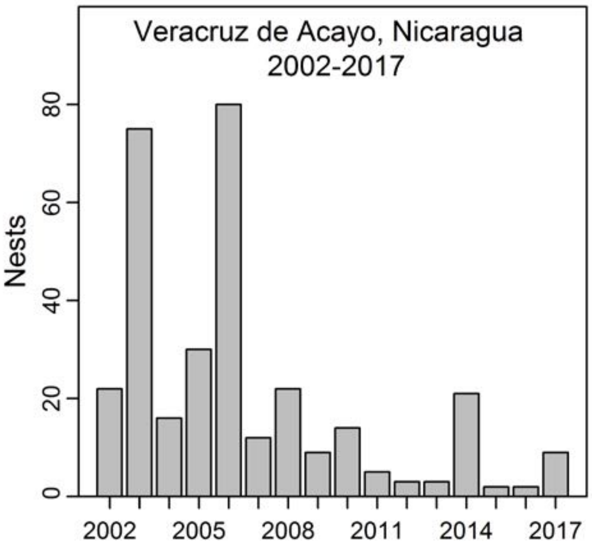
We did not perform a trend analysis on the other nesting data because we did not have 9 years of data collected in a consistent and standardized manner (i.e., methodology has changed in recent years). Instead, we include bar graphs of these data (Figure 49).

Figure 49. Bar graphs of nest data for East Pacific beaches. Asterisks indicate years when data were not available. Zeros (0) indicate years when no nests were observed.




Nombre de Jesus, Costa Rica 2010-2015
Nests
0
5
10
15
20
2010
2011
2012
2013
2014
2015
Cabuyal, Costa Rica 2011-2015
Nests
0
5
10
15
20
25
30
2011
2012
2013
2014
2015
Mexiquillo, Mexico 1996-2012
Nests
0
100
200
300
400
500
600
1996
2000
2004
2008
2012
2016
*
* * * *
Chacahua, Mexico 2010-2016
Nests
0
50
100
150
2010
2012
2014
2016
*





Our trend data yield similar results to other published findings. The IUCN Red List concluded that this population is decreasing and has declined, by 97.4 percent, over the past three generations (Wallace *et al.* 2013b).

11.1.2.1 Mexico

The number of nests at Mexico nesting beaches has declined precipitously in recent decades (Figure 10; 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). During that aerial survey, Pritchard (1982) recorded that a large number of nesting females were killed on the beaches as they attempted to nest, a "distressingly high level of poaching" both for meat and eggs. 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 four kilometers 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.

Concern regarding reported declines at major nesting beaches began in the early 1980s through the early 1990s. For example, researchers from the National University of Mexico recorded 3,000 to 5,000 nests annually from 1982 through 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. Sarti *et al.* (1994) reported the following reasons for the decline: intense egg harvest and killing of adult females, both at the nesting beaches and in open waters; natural fluctuation in the reproductive biology of leatherback turtles; or movement to other beaches by nesting females.

The bar graphs do not include the early estimates of nesting. 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 and Sarti 1997). The bar graphs provide data from these surveys. In 1995, "Proyecto Laud" (Leatherback Project) was formed to estimate the population size using comprehensive surveys. This group identified the primary nesting beaches as well as nesting beaches with lower nesting densities but which were still important to protect. In 1995 and 1996, they estimated approximately 1,100 females nesting throughout Mexico. The next two seasons, they estimated between 236 and 250 nesting females, and declines continued. Furthermore Sarti *et al.* (2000) noted that during the 1980s, 30 percent of the nesting females per season were remigrants, but that since the mid-1990s, there has been little evidence of remigration. Sarti *et al.* (2007) reported a decline in the proportion of remigrants found at Mexiquillo (from 1995 to 1999), coinciding with low numbers of nesting females in all of the index beaches. From 1999 through 2004, the proportion of remigrants rose, with the average percentage of all the index beaches at 22.4 percent from 1996 through 2004. Currently, based on data from 2014 through 2018 (preliminary) between 100 and 250 females nest at all the protected beaches in Mexico.

11.1.2.2 Costa Rica

During the 1980s, researchers realized that the beaches of Playa Grande, Playa Ventanas and Playa Langosta collectively hosted the largest remaining Pacific leatherback populations in Costa Rica. Since 1988, leatherback turtles have been studied at Playa Grande, which at the time was considered the fourth largest known leatherback nesting colony in the world. From 1988 to 1998, the number of nesting females per season declined from 1,367 to 117 females (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 fewer than 30 nesting females in recent years at all beaches (i.e., through 2016; The Leatherback Trust 2018). Low nesting during the 2001/2002 and 2002/2003 seasons was likely due, in part, to the effect of the ENSO transitions (Saba *et al.* 2007; Reina *et al.* 2009); however, nesting has not recovered to the 2000/2001 levels.

Since tagging began at Playa Grande in 1994, there has been a low nesting female return rate (16 to 25 percent) in the 5 or 6 years following tagging (Santidrián Tomillo *et al.* 2007). Santidrián Tomillo *et al.* (2017b) calculated an annual mortality rate of 22 percent for nesting females at La Baulas National Park. Over the same period, the return rate at St. Croix was approximately 60 percent (McDonald and Dutton 1996 *in* Reina *et al.* 2002a) and the annual mortality rate was 11 percent (Dutton *et al.* 2005). Thus, comparatively few leatherback turtles return to nest on east Pacific nesting beaches, likely indicating high mortality rates during non-nesting years. Bell *et al.* (2003) found that while leatherback turtles at Playa Grande had a high rate of fertility (93.3 ± 2.5 percent), embryonic death was the main cause of low hatching success in this population. At Playa Grande, temperatures above 30.0 °C produce exclusively female hatchlings, while below 29.0 °C, all hatchlings are male (Binckley *et al.* 1998). Hatching success is very dependent on ENSO in this region. At least in Costa Rica, hatching success can change dramatically depending on precipitation levels, which are influenced by ENSO (Santidrián Tomillo *et al.* 2012 and consequences at the population level in Saba *et al.* 2012).

While the nesting beaches at Las Baulas National Park represent the majority of nesting leatherback turtles (approximately 75 percent, based on the most recently reported five years of nesting (2011 to 2015; Santidrián Tomillo *et al.* 2017b), several other beaches once hosted small numbers of nesting females; however, females rarely if ever nest on these beaches. In the 1990s, 10 secondary beaches (defined as those where turtles nest regularly, are used by the same subpopulation of turtles that nest on the index beach(es) and are of secondary importance due to lower intensity of nesting activity) were identified, including (from north to south): Potrero Grande, Naranjo, Cabuyal, Matapalo, Camaronal, Caletas, Hermosa, Carate, Río Oro and Pejeperro/Piro. Playa Caletas on the Nicoya Peninsula hosted up to 20 nesting females nightly during the early 1990s, based on local interviews, and around 40 nesting events occurred during the 1997/1998 nesting season. On nearby Playa Coyote, locals reported three to five instances of nesting during 1997/1998. A 1999 survey on Playa Caletas reported 5 leatherback body pits between October 1 and December 11, with two body pits containing eggs or eggshells (Squires 1999). Currently, however, Caletas hosts very low (if any) and sporadic nesting. Compared to the 10 secondary beaches identified in the early 1990s, based on preliminary information gathered at a 2013 workshop held in Costa Rica, only a handful of beaches continue to host a small number of nesting females, albeit sporadic. These beaches are restricted to the Nicoya and Santa Elena Peninsulas: Naranjo, Cabuyal, Nombre de Jesús and Ostional. Naranjo has been monitored for the longest period of time, with 18 nesting females in 1971 and 3.7 ± 2.3 nesting females annually between 2011 and 2015. Over the same time period, a mean of 3.4 ± 1.9 females nested on Cabuyal, a mean of 5.2 ± 4.6 nested on Nombre de Jesús, and a mean of 5.3 ± 1.5 nested on Ostional. An estimated mean of 0.4 ± 0.5 nested on Caletas, with no nesting females during the 2012 to 2014 nesting seasons (Santidrián Tomillo *et al.* 2017b).

11.1.2.3 Nicaragua

In Nicaragua, researchers conducted aerial surveys and estimated 488 nests in the 2005/2006 nesting season. In the early 1980s, researchers monitored leatherback nesting at the Chacocente Wildlife Refuge on the northern coast of the Rio Escalante. From October through December, 1980, 108 females nested on Playa Chacocente, while during January, 1981, 100 females reportedly nested in a single night on Playa El Mogote (in Arauz 2002). Similar to many beaches along the eastern Pacific, the abundance of nesting females has decreased. An aerial survey of Playa El Mogote during the 1998/1999 revealed a nesting density was only 0.72 turtles per kilometer (Sarti *et al.* 1999 *in* Arauz 2002). During the 2000/2001 nesting season, community members near Playa El Mogote noted that 210 leatherback nests had been deposited. Of these, 31 nests produced hatchlings, while the rest were poached (85 percent poaching rate). During the 2001/2002 nesting season (monitored from October through March), leatherback turtles successfully nested 29 times. Of these, six nests were protected in a hatchery and 23 were poached (79.3 percent poaching rate) (Arauz 2002). Due to the high level of poaching in this area, when possible, researchers from Flora & Fauna International began re-locating leatherback nests beginning in 2002 through 2004, with a total of 98 leatherback nests protected during this time, with low emergence rates (22.2 percent) (Urteaga and Chacón 2008).

Nesting beach protection and monitoring were conducted at Playa Veracruz since 2002, at Juan Venado since 2004, and Salamina since 2008. During this period (2002 to 2010), up to 420 leatherback nests were recorded, with 94 percent protected. A total of 48 individual females have been identified using PIT and flipper tags (Urteaga *et al.* 2012). More recent estimates have been reported at Veracruz, where Salazar *et al.* (2019) recorded 340 nests, protecting 93.5 percent of them (89.7 percent in a hatchery and 3.8 percent in situ) between 2002 and 2014. Authors noted a downward trend even though there was a slight uptick during the 2014/2015 season. As shown in Table 26, the number of nests recorded at Salamina and Veracruz de Acayo have fluctuated between 17 and 25 and between 2 and 23 from 2014 to 2018 (V. Gadea, FFI, pers. comm., 2019), with no apparent trend. Given that the observed number of nesting females in 1980/1981 were primarily recorded on index beaches, the more recent number of nests indicate a significant decline in nesting in Nicaragua.

11.1.2.4 Productivity Parameters

Productivity parameters are well-documented for the DPS (Table 28). In Mexico, the mean size of nesting females was 144 cm CCL (range of 120 to 168 cm) (Sarti *et al.* 2007), and the estimated clutch frequency was 5.5, with an average clutch interval of 9.7 days. The average total fecundity per female was estimated to be 341 eggs per season, with a maximum of 744 eggs deposited in a season (Sarti *et al.* 2007). On average, females deposit 62 eggs per clutch, with 92 percent oviposition success. Compared to leatherback turtles nesting in the Atlantic Ocean, eastern Pacific nesting females are smaller and produce fewer eggs per clutch. Hatching success fluctuates between 35 and 52 percent (minimum = 0 to 30 percent; maximum = 58 to 78 percent). In rare events, 100 percent hatching success has been reported in individual clutches (Sarti *et al.* 2007). In Costa Rica, at Las Baulas National Park, the estimated clutch frequency is between 4.9 and 9.5 per season, with each female laying 61.8 to 65.6 eggs per clutch (Eckert *et al.* 2012). The average remigration interval is estimated to be 3.7 years. Hatching success was estimated to be approximately 45 percent (Santidrián Tomillo *et al.* 2007, 2012, 2014). At Playa Langosta, between 1999 and 2004, the size of nesting females ranged from 125 to 164 cm (CCL) (Piedra *et al.* 2007). In Costa Rica, at Las Baulas, the average remigration interval of female leatherback turtles was 3.7 ± 1.4 years. For the purpose of determining the number of nesting females in the East Pacific DPS, we used 4 years as the remigration interval, which is longer on average than other DPSs, which may re-migrate every two to three years. In Mexico, Proyecto Laúd began tagging efforts in the 1980s, and until the mid-1990s, only females nesting at Mexiquillo were tagged, primarily with flipper tags (Sarti *et al.* 2007). Since then, females on the index beaches have been tagged with both flipper and PIT tags, which has elucidated the proportion of returning females (remigrants) to neophyte nesting females. In the late 1990s, the proportion of remigrants reported on the index beaches was around 25 percent, which could indicate that there is high mortality of adults during their migration to and from foraging areas; alternatively, it could indicate a low detection rate or high dispersion rate to unmonitored beaches.

Table 28. Productivity parameters for the East Pacific DPS .

Productivity	**Variable by Nation**	**Reference**
Size of nesting females (cm CCL)	Mexico (Michoacán): 143.8 ± 6.88 (range 120 to 168)	Sarti *et al.* 2007
	Mexico (Jalisco): 144.5 (range 135–151)	Castellanos-Michel *et al.* 2006
	Costa Rica (Playa Grande): 147 ± 0.48 (range 133–165)	Price *et al.* 2004

Productivity	**Variable by Nation**	**Reference**
	Costa Rica (Playa Langosta): 144.9 ± 6.7 (range 125 to 164)	Piedra *et al.* 2007
Annual mortality (percent)	Mexico (Mexiquillo 1984–1995): 22	Sarti *et al.* 1996
	Costa Rica (Las Baulas): 22–35	Santidrián Tomillo *et al.* 2008
Remigration Interval (years)	Mexico: 3	Garcia-Munoz 2000 *in* Sarti *et al.* 2007
	Costa Rica: 3.7 ± 1.4	Santidrián Tomillo *et al.* 2007
	Nicaragua: 3.3	FFI 2018 (pers. comm.)
Clutch size (average eggs)	Mexico: 62–64	Sarti *et al.* 2007; Proyecto Laud unpublished data *in* Pineda and Barragan Rocha, 2016
	Costa Rica (Las Baulas): 65	Santidrián Tomillo *et al.* 2007
Clutch frequency (nests per season)	East Pacific: 7.2	Reina *et al.* 2002; Wallace *et al.* 2013b
	Mexico: 5.5 (estimated; range: 3–12)	Sarti *et al.* 2007
	Costa Rica (Playa Grande): 4.9–9.45	*In* Eckert *et al.* 2012
Internesting interval (mean days)	Mexico: 9.7	Sarti *et al.* 2007
	Costa Rica: 9–9.6 (Playa Grande and Playa Langosta)	*In* Eckert *et al.* 2012
Incubation period (days)	Mexico: 57 to 60	Proyecto Laud unpublished data *in* Pineda and Barragan Rocha, 2016
	Costa Rica: 59.3 ± 2.5	Santidrián Tomillo *et al.* 2009 *in* Eckert *et al.* 2012
Hatching success (percent)	Mexico: 35 to 52	Sarti *et al.* 2007
	Playa Grande, Costa Rica: 47	Santidrián Tomillo and Swiggs 2015
Sex ratio	Mexico: 1.1:1 female:male	(A. Barragan, Kutzari, pers. comm., 2019).
	Costa Rica (Playa Grande): 85 percent female	Santidrián Tomillo *et al.* 2014

11.1.3 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.* (2017c) found a contraction of Costa Rica's overall nesting distribution since the 1990s.

The best available genetic data indicate a high degree of connectivity among nesting aggregations that comprise a single management unit without population subdivision. Dutton *et al.* (1999) did not find any genetic differentiation between nesting populations in Mexico (Playa Mexiquillo) and Costa Rica (Playa Grande) based on 496 bp mtDNA control region sequences. Preliminary results from a more comprehensive survey using longer (more informative) 763 bp mtDNA sequences and nuclear DNA (microsatellites) from 3 index nesting beaches in Mexico, also failed to find genetic differentiation (Barragan and Dutton 2000; Dutton *et al.* unpublished).

Further investigation with microsatellite, single nucleotide polymorphisms, and more extensive sampling of these Pacific rookeries may yet resolve finer scale population structuring within these regions; however, results are not available yet (Dutton and Shanker 2015).

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 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 Colorada 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 combined. Based on 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 3 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.* 2017c).

The foraging range of the DPS extends into coastal and pelagic waters of the southeastern Pacific Ocean; however, foraging is not as widely separated as compared to the West Pacific DPS. This relatively low diversity in foraging strategy, as indicated by satellite telemetry (Eckert and Sarti 1997; Shillinger *et al.* 2008) and genetic studies (Dutton *et al.* 2013b) leaves the DPS less resilient and more vulnerable to perturbations in ocean conditions due to climate change, ENSO, or the Pacific Decadal Oscillation. East Pacific leatherback turtles 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 areas of Central and South America could have greatly reduced coastal foraging in this population (Saba *et al.* 2008). 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 modify nesting habitat or limit prey availability. Nesting aggregations are largely connected; however, there is less exchange among distant nesting beaches.

11.1.4 Diversity

The East Pacific DPS exhibits some genetic diversity, as demonstrated by mtDNA haplotypic diversity (h = 0.66–0.71; Dutton *et al.* 1999). Such diversity provides 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 2017b). 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, which 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 occur in the southeastern Pacific Ocean, where they are similarly exposed to the effects of climate change, ENSO, or the Pacific Decadal Oscillation. Diverse foraging strategies may provide some resilience against local reductions in prey availability or catastrophic events, such as oil spills, by limiting exposure. Thus, the DPS has limited resilience to threats.

11.2 ESA Section 4(a)(1) Factors

The following sections describe and assess the 4(a)(1) factors or threats. For each, we evaluate the best available information on the threat. We also describe, if possible, the exposure and impact of each threat.

11.2.1 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 the East Pacific DPS. We discuss sand warming associated with climate change in Section 11.2.5.4 Climate Change. 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 (discussed in Section 11.2.5.1 Fisheries Bycatch).

Coastal 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 the increase in other environmental damages to sand associated with human development, including significant differences in 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 (Playa Tamarindo 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 more or less 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 for monitors on all-terrain vehicles (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 have reduced the threat of poaching, establishment of tourism-focused coastal development that does not comply with the existence of management plans could threaten 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; however, we were unable to find quantitative estimates of this reduction. We conclude that habitat loss and modification is a threat to the DPS.

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

The primary cause of the historical decline of the East Pacific DPS was the harvest of nesting females and eggs in Mexico (and poaching of eggs and nesting females following the 1990 law protecting all sea turtles) and Costa Rica (and primarily poaching of eggs at Las Baulas National Park following the presidential decree in 1991 and permanent law establishing the park in 1995), and Nicaragua. To reduce the harvest of turtles and eggs, several regulatory mechanisms and protections have been established in the three nations hosting nesting beaches, as detailed below. 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 and cooler nests due to a lower number of metabolizing embryos may have reduced hatching success (Sieg *et al.* 2011). However, in Costa

Rica only nests in danger of tidal inundation are moved (approximately 10 percent of nests); most nests remain *in situ*.

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 killed at nesting beaches and in open waters (Sarti *et al.* 1994; Sarti *et al.* 1998). The harvest of turtles and eggs is now prohibited as a result of national legislation. Although the levels of poaching have been consistently reduced over the years, poaching pressure is still high wherever beach patrols do not occur (Lopez and Sarti 2016 *in* Santidrián Tomillo *et al.* 2017b). For example, Mexiquillo produced hatchlings every season in the 1980s; however, even with efforts to protect the nests in place, less than half of the total number of clutches (30 to 40 percent) were successfully protected. Nichols (2003) notes that leatherback turtles were once harvested off Baja California but their meat is now considered inferior for human consumption. Leatherback turtles are not generally captured for their meat or skin, but the poaching of nesters has been known to occur 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 protections afforded to the nesting beaches at Las Baulas National Park, illegal poaching of eggs still occurs, though quite rarely. The black market for eggs remains strong, where local bars throughout Guanacaste and elsewhere continue to offer shots of raw sea turtle egg yolks which accompany beer or liquor (Wallace and Piedra 2012).

In 1991, the Parque Nacional Marino Las Baulas was created and subsequently ratified by law in 1995. Prior to that decree, the largest nesting beach in this area, Playa Grande and the adjacent estuary (Tamarindo) were established as the Tamarindo Wildlife Refuge. 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 to near 0 percent in 1993/1994 (Santridián Tomillo *et al.* 2007). Since 1993, environmental education and conservation efforts through active law enforcement has greatly reduced egg poaching in some areas of Costa Rica (Chaves *et al.* 1996).

In Nicaragua, prior to conservation efforts in the early 2000s, poachers took nearly 100 percent of the nests at the three nesting beaches. Nesting beach protection has occured 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). Recently Flora & Fauna International (pers. comm., 2018) stated that while Veracruz de Acayo and Salamina are protected at 100 percent, Isla Juan Venado is not permanently monitored; therefore, there is some poaching. Poaching occurs at high levels at other beaches, such as Playa El Mogote, in Nicaragua. 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).

The extensive and prolonged effects of comprehensive egg harvest levels of nearly 90 percent for about two decades have depleted the leatherback turtle population in Costa Rica and Mexico (Sarti Martínez *et al.* 2007; Santidrián Tomillo *et al.* 2008; Wallace and Saba 2009). Currently, nesting females and eggs are exposed to poaching. Though conservation efforts have reduced the levels of both, 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.

11.2.3 Disease and 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 on Mexiquillo, in 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 Eastern Pacific leatherback turtles throughout their life stages. Eggs and hatchlings are eaten by crabs, ants, birds, reptiles, mammals, and fish. In the Nicoya Peninsula, on the Pacific coast of Costa Rica, Squires (1999) documented evidence of potential nest predation by dogs, coyotes, and raccoons. As adults, principal predators at sea include killer whales, crocodiles, 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 (e.g., female reproductive organs).

In Costa Rica, during the 1993/1994 season several nests were lost to predation and infestation by maggots (Schwandt *et al.* 1996). 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).

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). 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 (Sarti *et al.* 2007), such as feral dogs and jaguars. Some of these efforts are funded through the ESA and Marine Turtle

Conservation Act of 2004. Without these protections, predation on nesting females, eggs, and hatchlings is likely to increase slightly. We conclude that predation is a threat to this DPS.

11.2.4 Inadequacy of Regulatory Mechanisms

The East 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. Despite efforts to reduce impacts, poaching and fisheries bycatch continue to be major threats to this DPS.

11.2.4.1 International Regulatory Mechanisms

Several international regulatory mechanisms apply to this DPS. The Inter-American Convention for the Protection and Conservation of Sea Turtles, 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.

The Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC) is the only binding international treaty dedicated exclusively to sea turtles and sets standards for the conservation of sea turtles and their habitats, with an emphasis on bycatch reduction. The Convention area is the Pacific and Atlantic waters of the Americas, and there are currently 16 member nations to the Convention. In 2004, at the 2nd Conference of the Parties of the Inter-American Convention for the Conservation of Sea Turtles (INF-16-04), the parties resolved to conserve and protect leatherback turtles through protection of eggs, bycatch reduction of individuals on the high seas, and increased awareness and education of the public on the population status of leatherback turtles and the need to recover the species. In 2015, at the 7th Conference of the Parties, the parties 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 bycaught leatherback turtles in fisheries; and 6) agree to a 5-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://savepacificleatherbacks.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 Pacifico (Red Laúd OPO) (*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. 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. Pollution is a threat to the East Pacific DPS.

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 observer forms 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 5 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 and fish bait significantly reduce 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. In 2018, the United States continues to propose this resolution, in addition to requiring a reduction in fishing activity adjacent to the nesting beaches off Mexico and Costa Rica.

Throughout the world, 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*.

11.2.4.2 National Regulatory Mechanisms

Most nations within the range of the DPS have laws prohibiting the harvest of turtles and eggs. This applies to nesting turtles and those captured at sea. Though 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 are bycaught, the turtles are released and not retained (e.g., Chile; Donoso and Dutton 2010).

Several protected areas occur 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, or those that contain an extensive coastline that is difficult to monitor and protect. Protected nesting beaches include:

- Mexico (Mexiquillo (until 2013); Playa de Tierra Colorada, Playa Cahuitán, Playa San Juan, Bahia de Chacahua, Playa Barra de la Cruz)
- Costa Rica (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))
- Nicaragua 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 of these boundaries during their inter-nesting period, which makes them vulnerable to fisheries outside of 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^2, 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. Expansion of the marine protected areas would reduce bycatch and vessel collision.

In the following paragraphs, we provide information on the regulatory mechanisms within each nation.

Chile

In November 1995, Chile passed a presidential decree (No. 225), which established a closed season for the harvest of sea turtles and/or nesting habitats. The Decree was updated in February 2007 to become a permanent closure on the use of sea turtles and their products. In 2012, the Quarapara Tortugas Marinas Chile was established to promote, coordinate, and develop research that contributes to the knowledge and conservation of sea turtles in Chile (Alavarez *et al.* 2013).

Colombia

The most important national legislation in Colombia affecting sea turtles along their Pacific coast was the 1986 National Agreement for the Protection of National Resources and Nature in the South Pacific Region.

Costa Rica

The key legislation in Costa Rica protecting turtles was Presidential Decree No. 8325, which was passed in 2002, the Law of Protection, Conservation, and Recuperation of Marine Turtles.

Ecuador

The most important legislation in Ecuador for the protection of sea turtles was Law RO 51, which was passed on December 12, 1990, which protected all sea turtles in Ecuadorian national waters. In addition to the wildlife protection laws for Ecuador, shrimp trawling was partially banned in February 2012 by Ministerial Agreement No. 020. This was later modified by the Ministerial Agreement No. 425 in October 2012 to fully shut down the Ecuadorian shrimp fleet.

Guatemala

The first national effort for sea turtle protection was the Presidential Agreement, passed on October 26, 1971, which declared the closure of capture, circulation, and commercialization of sea turtles, including leatherback turtles. An additional governmental agreement was passed on February 17, 1981 that prohibited the capture, circulation, and commercialization of all species of sea turtles that inhabit and reproduce on the Guatemalan coasts (Muccio and Flores 2015). Other Guatemalan national legislation includes: 1) Law of Protected Areas (Congressional Decree 4-89 of the Republic of Guatemala) that regulates everything related to the use and management of protected areas and wildlife, including CITES species such as sea turtles; and 2) Fisheries Law (Decree 80-2002) that mandates the use of TEDs and establishes greater sanctions for violations of the TED law. This law was updated by the Ministerial Agreement 46-2005.

Given that Guatemala became a signatory state to the Inter-American Convention for Sea Turtles, which established parameters for those species threatened with extinction, Guatemala established a program that includes limits on the commercialization of eggs and prohibits the take of leatherback turtles (Muccio and Flores 2015).

Honduras

The primary wildlife law for sea turtles in Honduras is the General Law of the Environment (Decree 104-93) that provides national regulations for sea turtle use.

Mexico

In 1990, Mexico recognized the importance of protecting sea turtles through the passage of a presidential decree, which banned the use or sale of sea turtle products throughout all of Mexico (Aridjis 1990). Signed by then-President Carlos Salinas de Gortari, this was a monumental declaration on the part of the Mexican government that prohibited the harvest of sea turtles at all life stages in Mexico, with mandatory fines and jail time for individuals found with sea turtle products.

An additional law that served to protect sea turtles was a modification of the official Mexican Regulation NOM-002-PESC-1993, passed in 1997, to mandate the responsible management of shrimp fisheries throughout Mexico by implementing the use of turtle excluder devices. While we have no documented takes of leatherback turtles in shrimp trawls, we know that they are taken in the southeastern United States so the threat is possible, particularly if effort is adjacent to the nesting beaches. In 2004, the Official Mexican Emergency Regulation NOM-EM-002-PESC was passed that provided technical specifications for the turtle excluder devices used by the Mexican shrimp trawl fleet.

Nicaragua

Nicaragua prohibits the tradition of consuming sea turtle eggs through the passage of Law No. 651 and Ministrial Resolution No. 043-2005. However, the harvesting and consumption of sea turtle eggs continue throughout the coastal areas of the Pacific coast of Nicaragua. One vital piece of legislation was the declaration of a protected area for the nesting beaches in the Rio Escalante-Chacocente region by National Decree No. 1294 in 1983, and the declaration of a wildlife refuge in the Pearl Keys area in 2010.

Panama

The most important law that provides protection for sea turtles is Wildlife Law (1995) and Environmental Law (No. 41). Law No. 003 was declared in 2009, which adopted the Code of Conduct for Responsible Organization of the United States Food and Agriculture Organization (FAO) and its Annexes on Fishing and the International Plan of Action is also taken to Prevent, Deter, and Eliminate Fishing Illegal, Unreported, and Unregulated of the FAO.

Peru

In 1997, the Ministry of Agriculture in Peru, through Resolution 1710-77-AG/DGFF categorized leatherback turtles, along with other hard-shelled turtles as in a “Vulnerable Situation” due to excessive hunting and habitat destruction. In addition, Resolution RM-1065-76-PE banned the capture of leatherback turtles in coastal waters due to the indiscriminate fishing/retention of this species. Then in 1995, RM-103-95-PE, the Ministry of Fisheries banned the capture of all species of marine turtles (Morales and Vargas 1996 [Marine Turtle Newsletter No. 75 1996]).

Sea turtle protection was first mandated in Peru in 2001 by the Forestry and Wildlife Law 014-2001, which established protective measures for wildlife resources and established interactions for wildlife protection violators. This was updated in 2004 by Law 034-2004, which approved the categorization of all sea turtles in Peruvian waters as threatened, thus prohibiting their hunting, capture, possession, transportation, or exportation for commercial purposes.

United States
There are numerous laws in the United States that promote the protection and conservation of sea turtles. The Magnuson-Stevens Fishery Management and Conservation Act also is a national instrument, although it has larger implications in the international arena by mandating the responsible fishing practices and bycatch mitigation within fleets/nations that export fish products into the United States.

The Marine Turtle Conservation Act is also 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 nations, with emphasis on protecting nesting populations and nesting habitat. Funds from this Act have provided financial support for monitoring and protection throughout the nesting range of this DPS.

The National Environmental Policy Act of 1969 also has a role in sea turtle protection as it requires the review of federal actions to assess their environmental impact and the development of various alternatives for carrying out the activity to reduce impacts to the natural environment.

11.2.4.3 Summary

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 are inadequate to sufficiently 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. In summary, we consider the inadequacy of the regulatory mechanisms to be a threat to the DPS.

11.2.5 Other Natural or Manmade Factors Affecting its Continued Existence.

Other threats affect the East Pacific DPS, with fisheries bycatch being a major threat. Additional threats include: pollution; oceanographic regime shifts, and climate change.

11.2.5.1 Fisheries Bycatch

Bycatch in commercial and recreational fisheries, both on the high seas and coastally, 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 (except probably hatchlings), and reducing the range of migrating and foraging leatherback turtles.

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. Fishing effort was highest in the central South Pacific Ocean (south of Hawaii), which overlaps with the foraging range of this DPS. With observers on only a fraction of longline vessels in the eastern tropical Pacific Ocean, and only a recent requirement through an IATTC resolution, these estimates may be considered 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 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 has been estimated by several researchers (e.g., Spotila *et al.* 1996; 2000).

In their global study of sea turtle bycatch, where data were available, Wallace *et al.* (2013a) found that longline bycatch had a low impact, but that net bycatch had a high impact on 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 Ecuadoran, 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 and Sarti 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 or two 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 (only vessels observed) since 2014, and they 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). Artisanal fisheries are characterized by small boats, less than or equal to 15 m in length, and non-mechanized gear. 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.* 2007a). 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 mahi mahi (*Coryphaena hippurus*), blue sharks (*Prionace glauca*), mako sharks (*Isurus oxyrinchus*), and rays (*Myliobatis* spp.) (Alfaro-Shigueto *et al.* 2007a). 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 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 (2009–2014). Of 953 carapaces recorded, leatherbacks comprised only 1.4% 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 centimeters (range: 80–135, n = 10), 70 percent of the leatherbacks were juveniles, 30 percent were sub-adults and 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 in four fisheries from 2000 to 2007, as follows: 40 turtles (with a range of 37 to 44) in the driftnet fishery, with 80 percent released alive; six turtles 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 with primarily juveniles and subadults (mean CCL was 125.1 ± 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 that survive the gear interaction 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 leatherback turtles was 117 ± 10.65 cm, while in 2005, the mean CCL was 109.27 ± 14.4, possibly indicating overexploitation due to systematic and sustained harvests, particularly during El Niño years (de Paz 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 (de Paz 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° S and 38° S (furthest south was 38°39'S and 84°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 this 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. Mortality reduces abundance, by removing individuals from the population; it also reduces productivity, when nesting females are bycaught and killed. Several studies indicate that fisheries bycatch has contributed to the decline of this DPS. As noted above, there have been efforts by individual nations and regional fishery management organizations to mitigate and reduce the threat of bycatch. However, we conclude that fisheries bycatch remains a major threat to the East Pacific DPS.

11.2.5.2 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 mid-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 containing this aggregation 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. Degradation of plastics throughout the world's oceans leads to a rise in the level of toxins in the area and most of the garbage patch contains microbeads, tiny abrasives less than five micrometers in size usually found in personal hygiene products, clothes, fishing debris from fishermen in the southern hemisphere, and microscopic fragments of larger pieces which have been broken down in the ocean. Efforts to begin cleaning up the South Pacific garbage patch have not yet begun, but The Ocean Cleanup project has begun to clean up the North Pacific garbage patch, initiated in the fall of 2018.

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, yet the impact remains unquantified. Leatherback turtles subsist primarily on jellyfish and other gelatinous zooplankton and may be prone to ingesting plastics resembling their food source (Mrosovsky 1981; Schuyler *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 (Juárez *et al*. 1998).

Since high seas driftnet fisheries were banned through a United Nations moratorium in 1992, active north and south Pacific large scale driftnets no longer pose a threat to leatherback turtles. However, the numbers of driftnets that may still be at large may continue to "ghost-fish" for a variety of marine species, including leatherback turtles. Thus, entanglement in ghost fishing gear is a concern (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. In addition, fishermen are prohibited from releasing salt bags and associated debris into the ocean.

There have been few studies of levels or effects of toxins on leatherback turtles that provide any link 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 (Fe, Pb, Cu, Ni, Cr, Cd, Mn, Zn, and As) and other toxicants (PCBs, OC, DDT and its metabolites) and studied the relationship to hatching success for 30 females. Researchers indicated that baseline data would be important to determine potential influences these substances may have on the population of these turtles. Metal levels were highly variable, but there were no significant differences within and between groups of females. None of the pesticides tested were present in the samples. Overall, researchers were not able to link metal concentrations to hatching/emergence success, hatchling size or mass. They 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, yet the impact remains unquantified. We conclude that pollution is a threat to this DPS.

11.2.5.3 Oceanographic Regime Shift

The East Pacific DPS is affected by oceanographic regime shifts. In the eastern equatorial Pacific Ocean, reductions in productivity parameters are primarily associated with El Niño. For example, Santidrián Tomillo *et al.* (2020) found dramatic declines in the reproductive success of leatherback turtles after the extreme 2015-2016 El Niño event. In addition, sex ratios may become biased up to 100 percent female hatchlings (Santidrián Tomillo *et al.* 2014). In North Pacific Costa Rica hatching success is very low due to dry and hot conditions on the nesting beaches during El Niño years, and is high during la Niña events due to increased precipitation in this area (Santidrián Tomillo *et al.* 2012). 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.

11.2.5.4 Climate Change

Climate change is a potential 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 90–99 percent probability; IPCC 2007) to become warmer and likely (defined as 66 to 90 percent probability; IPCC 2007) drier by 2100 (Saba *et al.* 2012). In addition, climate variability is likely to change the strength and frequency of ENSO events, although there is less agreement on the frequency and magnitude of changes to these events. Using global climate models, Santidrián Tomillo *et al.* (2020) predicted an increase in the occurrence of extreme ENSO events from 0.7 to 2.9 events per decade by the end of the century, which corresponds to a 19 percent projected decline in the reproductive success of the East Pacific DPS. 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 ± 1 percent per decade over the next century of climate change under the A2 scenario, which considered emissions scenario from 2000 to 2100 (Special Report on Emissions Scenarios 2007); similar declines occurred for other scenarios. The nesting population was projected to remain stable up until around 2030 but then reduced 75 percent by the year 2100. Hatching success and emergence rates, which were predicted to decrease in association with projected 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, 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 conservation efforts at the primary nesting beaches in Costa Rica, with the general warming of Central America in the near future, that the chances of a new nesting area emerging with more ideal conditions (i.e., cooler and wetter) is unlikely.

High 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. 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, since 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 will likely continue to increase, resulting in a further decline in the number of hatchlings produced (Santidrián Tomillo *et al.* 2014). An increase in the percentage of females might benefit the DPS in terms of population dynamics; however, such benefits would be tempered by the associated lower emergence and hatching 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 associated with lower emergence and hatching 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 exacerbate population declines. As mentioned previously, the preferred foraging habitat of eastern Pacific leatherback turtles is characterized by relatively low sea surface temperatures and chlorophyll-a. Using information derived from satellite tagged 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.* 2015). Depending on whether this population is able to shift its preferred migratory routes and foraging habitat over time, remigration intervals may shorten or lengthen, which could influence reproductive productivity.

Climate change is a potential threat to the East Pacific DPS given that it could affect 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. The impacts of high sand temperatures already occur. Foraging areas could be impacted via changes in ocean productivity, sea surface temperatures, and availability of prey.

11.3 Extinction Risk Analysis

After reviewing the best available information, we conclude that the East Pacific DPS has a high extinction risk, as summarized in the following paragraphs. Our total index of nesting females (755 females) at monitored beaches 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 regime shifts that expose the DPS to low-productivity events. Based on these demographic factors, we find the DPS to be at an increased risk of extinction as a result of past threats.

Current threats continue to place this DPS at a high risk of extinction, as summarized in Table 29. 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 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 associated and predicted increase in frequency and strength of ENSO (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 sufficiently reduce the impact of these threats. Further, it is important to note that conservation 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 United States through the ESA and Marine Turtle Conservation Fund. A reduction in effort is expected, assuming no protection under the ESA.

Table 29. Threats to the East Pacific DPS. Exposure refers to the individuals affected by the threat. Impact refers to how that threat affects the demographic factor(s). The primary threat is identified with an asterisk.

Threat	Exposure	Impact

Destruction or modification of habitat	Eggs	Reduction of hatching success (productivity)
Overutilization	Eggs and nesting females; turtles at sea	Loss of nesting females (abundance) and reproductive potential (productivity)
Predation	Eggs and hatchlings	Reduction of hatching success and hatchling survival (productivity)
Inadequate regulatory mechanisms	Eggs and turtles of all life stages	Some laws are poorly enforced
Fisheries bycatch*	Adults off nesting beaches; foraging juveniles and adults	Loss of individuals (abundance) and loss of nesting females (productivity)
Regime shift	Foraging juveniles and adults	Reduced productivity
Pollution	Eggs and turtles of all life stages	Lethal (abundance) and sublethal (productivity) effects
Climate change	Eggs and turtles of all life stages	Reduction of nesting and hatching success (productivity); changes in foraging resources

Thus, we find that the East Pacific DPS is at a high level of extinction risk. Its nesting female abundance and declining trend place its continued persistence in question. The productivity metrics further contribute to our concern over its continued persistence. 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. Poaching of eggs likely occurs in a portion of the range, and fisheries bycatch is the major, clear and present threat. These threats reduce abundance and productivity (i.e., imminent and substantial demographic risks) by removing turtles and eggs from the population at rates exceeding replacement. Additional productivity metrics (such as remigration interval), may play a role in this decline; however, the degree to which they affect the population trend is unknown. We conclude that the East Pacific DPS meets the definition for high risk of extinction (see Extinction Risk Assessment). The moderate risk definition does not apply because the DPS is at a high risk of extinction now (i.e., at present), rather than on a trajectory to become so in the foreseeable future. We have high confidence in our conclusion because of the reduced abundance and declining trend, combined with the magnitude of poaching and fisheries bycatch.

12. Appendix I: 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 predator 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 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. However, several conservation efforts have been previously discussed under the sections on Inadequate Regulatory Mechanisms. Therefore, Table 33, and the descriptions below, describe those conservation efforts that have not been previously discussed within this document. These efforts were not considered during the status review process. Instead, these efforts will be considered by the Services as they make their listing determinations.

Table 30. Conservation Efforts.

DPS	Northwest Atlantic	Southwest Atlantic	Southeast Atlantic	Southwest Indian	Northeast Indian	West Pacific	East Pacific
Accra Declaration of the Ministerial Committee of the Gulf of Guinea Large Marine Ecosystem (GOG-LME)-1998 Abuja Declaration of the Guinea Current Large Marine Ecosystem Project-2006		√	√				
African Convention on the Conservation of Nature and Natural Resources (Algiers Convention)	√	√	√	√			
Southwest Atlantic Sea Turtle Network		√					
Association of Southeast Asian Nations Ministers on Agriculture and Forestry					√		
Andaman and Nicobar Island Environmental Team					√		
Central American Regional Network	√						
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 on the Conservation of European Wildlife and Natural Habitats	√			√			
Convention for the Co-operation in the Protection	√	√	√	√			

DPS	Northwest Atlantic	Southwest Atlantic	Southeast Atlantic	Southwest Indian	Northeast Indian	West Pacific	East Pacific
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 of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention)						√	√
Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region (Cartagena 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):	√	√	√	√	√	√	√
Cooperative Agreement for the Conservation of Sea Turtles of the Caribbean Coast of Costa Rica,	√						

DPS	Northwest Atlantic	Southwest Atlantic	Southeast Atlantic	Southwest Indian	Northeast Indian	West Pacific	East Pacific
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)	√						
Eastern Pacific Leatherback Network						√	√
Eastern Tropical Pacific (ETP) Marine Corridor (CMAR) Initiative:						√	√
Food and Agricultural Organization Technical Consultation on Sea Turtle-Fishery Interactions	√	√	√	√	√	√	√
The Indian Ocean Tuna Commission (IOTC)				√	√		
Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA):				√	√	√	
Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC):	√	√				√	√
International Convention for the Prevention of Pollution from Ships (MARPOL)	√	√	√	√	√	√	√
International Union for Conservation of Nature (IUCN)	√	√	√	√	√	√	√
Marine Turtle Conservation Act	√	√	√	√	√	√	√
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					√	√	

DPS	Northwest Atlantic	Southwest Atlantic	Southeast Atlantic	Southwest Indian	Northeast Indian	West Pacific	East Pacific
Association of South East Asian Nations (ASEAN) 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					√	√	
Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region				√			
National Sea Turtle Conservation Project in India					√		
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):	√	√				√	√
Secretariat of the Pacific Regional Environment Programme						√	√
South-East Atlantic Fisheries Organization (SEAFO)			√				
South Atlantic Association		√					
United Nations Convention on the Law of the Sea (UNCLOS)	√	√	√	√	√	√	√
United Nations Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing	√	√	√	√	√	√	√

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 *https://au.int/en/treaties/african-convention-conservation-nature-and-natural-resources*

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 bycatch, 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 Fundacão 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 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 the 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).

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 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.

Centre for Herpetology/ Madras Crocodile Bank Trust: In India, since 1978, the Centre for Herpetology/ Madras Crocodile Bank Trust has conducted sea turtle surveys and studies in the islands. The Centre for Herpetology/ Madras Crocodile Bank 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 Madras Crocodile Bank Trust (MCBT) (Tripathy *et al.* 2012).

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 Memorandum of Understandings (MOUs) include 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 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 as of March 2013. 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*.

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, São Tomé 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." ***Error! Hyperlink reference not valid.*** 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 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 may relate to leatherback turtles is the Protocol Concerning Specially Protected Areas and Wildlife (SPAW) 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 SPAW 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 SPAW 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.

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 Commision 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 (WCPF Convention): The convention entered into force on 19 June 2004. The WCPF 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 Pacifico (Laúd OPO) (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.

Eastern Tropical Pacific Marine Corridor (CMAR) Initiative 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 Technical Consultation on Sea Turtle-Fishery Interactions: While not a true international instrument for conservation, the 2004 Food and Agriculture Organization of the United Nations' (FAO) technical consultation on sea turtle-fishery interactions was groundbreaking in that it solidified the commitment of the lead United Nations 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 gears 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 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 Inter-American Convention for the Protection and Conservation of Sea Turtles. (*http://www.iattc.org/*).

The International Commission for the Conservation of Atlantic Tunas (ICCAT) 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 in 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 are bycaught 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. In 2005, the IOTC adopted Resolution 05/08, superseded by Resolution 09/06 on Sea Turtles which 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 long-liners operating in the Indian Ocean (*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 captured 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 binding international treaty dedicated exclusively to sea turtles and sets standards for the conservation of sea turtles 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: The U.S. Marine Turtle Conservation Act of 2004 was established to support conservation projects that protect and conserve global marine turtle species. The Act granted the U.S. Fish and Wildlife Service the authority to establish the *Marine Turtle Conservation Fund* which provides funding for projects that conserve marine turtles primarily through cooperative efforts to protect, restore, and manage nesting sites.

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 Western Pacific Leatherback Turtles and their critical habitats.
http://awsassets.wwf.or.id/downloads/mou_trinationalpartneshipagreement_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 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*.

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 NGOs 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) reducing mortalities 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 the reduction of bycatch of sea turtles 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, 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 Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing: In 1989, the United Nations 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*.

13. Appendix II: NW Atlantic DPS Regulatory Mechanisms

The following is a list of regulatory mechanisms, by nation, that apply to the NW Atlantic DPS.

Albania

There are no specific national laws protecting or prohibiting take of sea turtles in Albania (Haxhiu, 2010). Albania recognized sea turtles as endangered under order Number 146 (Adoption of Red List Flora and Fauna; 8/5/2007). The Action Plan for the Conservation of Sea Turtles and their Habitats in Albania reviews the existing legislation that could be extended to provide the framework for protection of sea turtles and their habitats. These laws include biodiversity and wildlife, fisheries and aquaculture, and veterinary laws. These laws do not specifically mention sea turtles, but do provide the framework for the prevention of harm to wild animals, migratory species, habitat protections; permit issuance inspection, and control of research activities; and enforcement and penalties (Medasset, 2012). In 2013, Albania adopted the Action Plan for the Conservation of Sea Turtles and their Habitats in Albania (available at *https://issuu.com/medasset/docs/action_plan___en_/1?ffande=2133522/3422466*) prepared by Medasset (Ministria E Mjedisit 2015). The goal of the plan is the conservation and viability of sea turtles in Albanian seas so that sea turtle species and their habitats are fully protected under Albanian national law.

Algeria

Environmental protection in Algeria is based in Law 03-10 (July 19, 2003) on the Protection of the Environment within the framework on Sustainable Development. It sets the framework for protecting biological diversity, among other mandates. No legislation specific to sea turtles was identified.

Anguilla

In Anguilla, national legislation provides complete protection for leatherback turtles and their eggs (Richardson *et al.* 2013). All sea turtles are protected under the Revised Fisheries Protection Regulations f40-1. These regulations prohibit the take or attempt to take sea turtles, including turtle eggs. The regulations prohibit take, slaughter, purchase, sale, and possession of turtles, or a portion of the meat and of eggs. The regulations are in force for a period of 15 years beginning on December 15, 2005. The Fisheries Protection Regulations are supported by the Biodiversity and Heritage Conservation Act c.R55 that prohibits the taking, possession, and trafficking of wildlife species within Schedule 1, including sea turtles.

Antigua and Barbuda

The Fisheries Act of 2006 is the primary legislation regulating sea turtles in Antigua and Barbuda. In 2013, the Fisheries Act was signed and Antigua and Barbuda enacted fisheries regulations. Regulations prohibit the take, sale, purchase, or possession of turtles or turtle eggs; the take, capture, or disturbance of any turtle found on shore, and the disturbance, alteration, or destruction of any turtle nests. The Minister may declare an open season for sea turtle species. However, fishing for, taking, selling, purchasing, or possessing any leatherback turtle, or part thereof, is not allowed during the open seasons. The open season also includes prohibitions related to eggs, hatchlings, turtles found on shore, and nests. The Minister may also publish protections for designated nesting areas.

In 1990, UNEP partnered with WIDECAST to launch the Caribbean's first series of national species recovery plans. Each STRAP 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 (www.widecast.org). One objective of the recovery action plan series is to promote and implement science-based sea turtle management and conservation programs. Antigua and Barbuda published a plan in 1992 (Fuller *et al.* 1992).

Aruba
Law in Aruba has protected sea turtles since 1987. Their nests and eggs have been protected since 1980 through the Marine Environmental Ordinance of Aruba (Marien Milieuverorderning Aruba) AB 1980, No. 18. Under the Ordinance, it is illegal to disturb nests, remove, destroy, possess, deliver transport, buy, or sell turtle eggs. Decree No. 51 of 1987 listed leatherback and other sea turtles. It is prohibited to kill, sell, purchase, deliver, export, import, or possess sea turtles and/or their parts (living or dead). It is also prohibited to use the products to make goods. The Nature Conservation Ordinance (Natuurbescherming Beschermingsverodening) AB 1995, No 2 prescribes penalties for killing a protected species (Bräutigam and Eckert 2006). In addition, sixteen nature areas, including areas beneficial to sea turtles, are to be incorporated into the Arikok National Park (Ministry of Economic Affairs, 2017). Aruba has had a STRAP since 1993 (Barmes *et al.* 1993).

Bahamas
In September 2009, the Fisheries Regulations governing marine turtles were amended to give full protection to all sea turtles found in Bahamian waters by prohibiting the harvesting, possession, purchase, and sale of turtles, their parts, and eggs. The new regulations also prohibit the molestation of sea turtle nests (Bjorndal and Bolten 2010).

Barbados
With the Fisheries (Management) Regulations of 1998, the Government of Barbados enacted sea turtle protections. It is illegal to fish for or ensnare sea turtles, to endanger or disturb any nest, or, to remove eggs from a nest. It is also illegal to possess, sell, expose for sale or purchase any sea turtle, part of a turtle, or eggs (Ministry of Agriculture and Rural Development, n.d.). Other legislation protecting sea turtles is the Coastal Zone and Management Act (1998-39) which provides authority for protection of resources and the designation of marine protected areas. Horrocks (1992) published a STRAP for Barbados in 1992.

Belgium
While no legislation specific to sea turtles was identified, the Law of 20th January 1999 Marine Environmental Protection Act (MEPA) provides the legal framework to protect the Belgian part of the North Sea from marine pollution and to facilitate the conservation, restoration, and development of the natural environment of the sea. The Marine Spatial Plan (Royal Decree, March 20, 2014) provides a framework for marine spatial planning.

Belize
In June 2002, the fisheries regulations were revised to prohibit fishing, possession, or trade in products of all species of sea turtles found in the region. The regulations also prohibit take of turtles on land; the disturbance, take, purchase, sale or possession of any turtle or eggs; and

interference of any turtle nest except under written permission by the Fisheries Minister. The regulations allow some fishing for "traditional" use (hawksbills cannot be taken under this usage). A STRAP was published for Belize in 1992 (Smith *et al.* 1992).

Bermuda
All sea turtles are protected in Bermuda's waters under the Fisheries Act of 1972. The Protected Species Act of 2003 prohibits the take, import, export, sale, or purchase of a protected species. All sea turtles are listed under this act as critically endangered or endangered per the IUCN criteria. The Act considers willful destruction, damage, removal or obstruction of habitat, and the taking, importing, exporting, selling, purchasing, transporting, or possessing a sea turtle, or any part thereof, an offence. Leatherback turtles are protected as a Level 1 Protected Species under the Protected Species Amendment Act (2011).

Bonaire
The nests and eggs of sea turtles have been protected in Bonaire since 1961 in the island legislation "Eilandsverordening tot Bescherming van de Zeeschildpadden en Kreefen". Full protection was conferred in 1991. Under Bonaire's Marine Environment Ordinance (A.B 1991 Nr. 8 Article 14) (Verorderning Marien Milieu), it is prohibited to kill, catch, or possess sea turtles; offer for sale, sell, trade, donate, or offer as a dish sea turtles, sea turtle meats, or other sea turtle products. It is also prohibited to disturb, destroy, or remove eggs from sea turtle nests or to possess, have for sale or delivery, offer for sale, sell, buy, trade, donate or transport sea turtle eggs. By decree from an Executive Council, the prohibition on killing, catching, or having sea turtles can be lifted for one year if the population can sustain it. The decree should include gears, species, seasons, quotas, and maximum/minimum sizes.

Brazil
Leatherback turtles are listed as critically endangered in Brazil (ordinance MMA No 444, 12/17/2014). Brazilian legislation gives sea turtles comprehensive protection prohibiting consumption, capture, slaughter, trade, and transportation. The Law on Environmental Crimes No. 9605 makes the harvest or consumption of sea turtles illegal.

British Virgin Islands
The Virgin Islands Fisheries Act (1997) and Fisheries Regulations (2003) regulate the BVI turtle fishery. The 2003 Regulations put in place a moratorium on the catch of leatherback and loggerhead sea turtles. Nests, eggs, and nesting turtles of all species are protected. BVI has had a STRAP in place since 1992 (Eckert *et al.* 1992).

Canada
In 1981, the leatherback turtle was first designated as endangered in Canada by the Committee on the Status of Endangered Wildlife in Canada (COSEWIC). This status was confirmed in 2001 and 2012, and it was listed as endangered under the Species At Risk Act in 2003. The Act makes it illegal to kill, harass, capture, or harm leatherback turtles. It also forbids possession of a turtle, or part of one, whether alive or dead. In addition to SARA, the Habitat Protection provisions of the Fisheries Act (1985) and the Oceans Act (1996) give DFO authority to create Marine Protected Areas to protect endangered and threatened species. The leatherback is also protected

under the 1996 New Brunswick Endangered Species Act (Atlantic Leatherback Turtle Recovery Team, 2006).

Canary Islands
All species of sea turtles are protected by Spanish and Canary Island legislation. Legislation for the Canary Islands includes Law 4/1989 (3/27/1989) on the Protection of the Natural Habitat and the Wild Flora and Fauna; seaLaw 12/1994 from 19 December 1994 on the Natural Areas of the Canaries; Decree 161/97 of the delegations in environmental policy to the Island Councils (Fretey 2001).

Cape Verde
Decreto n°97/87: September 5, 1987 (Law 06/94), Article 17 prohibited the capture of sea turtles from the 1st July until the end of February (Fretey 2001). In 2002, Decreto-Regulamentar N° 7/2002 was issued protecting sea turtles and their habitats (de Santos Loureiro 2008).

Caribbean Netherlands
The country of the Netherlands consists of a territory in Europe and the islands of Bonaire, Saba, and St. Eustatius. *Bonaire*, *St. Eustatius*, and *Saba* are Dutch overseas "public entities", collectively known as the Caribbean Netherlands. The powers normally exercised by provincial councils are divided between the Island Governments and the National Government by means of the National Office of the Caribbean Netherlands.

The Caribbean Netherlands largely has its own laws and regulations, called BES-law. The obligations under international agreements are incorporated into national legislation as the Nature Conservation Framework Act BES (Wet grondslagen natuurbeheer en-bescherming BES). Every five years a Nature Policy Plan is approved by the Minister of Economic Affairs taking into account the Islands' spatial development plans. The plan should include the objectives for nature and landscape to be realized and nature values to be protected and taken into account in implementing the policy. The Fisheries Act BES (Visserijwet BES) and the Maritime Management Act (Wet Maritiem Beheer BES) complement the legislation. The National Government bears the final responsibility for protection of special areas and species in international treaties and conventions (Nature Policy Plan for the Netherlands 2013-2017).

Sea turtles are protected in the Caribbean Netherlands. Marine protected areas fully surround each of the three islands and nesting beaches are protected. Fisheries are also monitored.

Nature Conservation on the islands is mandated to non-governmental conservation organizations: Bonaire National Parks Foundation (STINAPA), St. Eustatius National Parks (STENAP), and Saba Conservation Foundation (SCF) through regulation and management agreements. These organizations develop and implement management plans and have enforcement authority.

Cayman Islands
The Marine Conservation (Turtle Protection) Regulations, 1996 Revision, provide protection to sea turtles in the Cayman Islands. It is illegal for unlicensed persons to harm, disturb, or take sea turtles, or their eggs. The regulations prohibit the possession of sea turtle eggs unless bred in captivity in accordance with a license granted under the Endangered Species Protection and

Propagation Law of 1978 or taken in accordance with a license issued under section 16 of the Law. The regulation prohibits take or disturbance of any sea turtle from May through September (Fleming 2001). In 2008, legislation was amended to extend the closed season from April to November, gear restrictions were introduced (e.g., banning set nets), and a maximum size limit for turtles was introduced (Cayman Islands Government, 2008).

Colombia

There is an extensive body of legislation in Colombia that relates directly or indirectly to sea turtles. The legislation is summarized in Bräutigam and Eckert (2006). Acuerdo N°021 (1991) establishes specific protection measures for all species as well as nesting beaches and foraging areas. Various other laws, decrees, and resolutions have been established directly or indirectly protecting foraging habitat and nesting beaches, as well as limiting fishing activities in areas of known turtle concentrations (Golfo de Morrosquillo, San Bernardo Archipelago, Golfo de Urabá, and the coast of Guajira), national parks, and other important areas. Some subsistence fishing of marine turtles is permitted by law (Ley de Pesca No. 13, 1990, Article 47). Colombia has also established additional protected areas.

Costa Rica

The key legislation in Costa Rica protecting turtles was Presidential Decree N°8325 passed in 2002 that was entitled Law of Protection, Conservation, and Recuperation of Marine Turtles. Prior to and since that time there have been numerous natural reserves, both marine and terrestrial, which provide benefits for turtles. The law incorporates articles setting forth penalties for those who kill, hunt, capture, transport or trade in marine turtles. The Ministry of Environment and Energy Conservation Areas National System manages Marine Protected Areas including sea turtle nesting sites.

Croatia

The leatherback, green, and loggerhead turtles have been protected under the Nature Protection Act since 1995 (Lazar, 2010). The updated Act entered into force in 2013. Other legislation with a significant impact on nature includes the Ordinance on Strictly Protected Species (Official Gazette Nos. 144/13 and 73/16).

Cuba

In 2008, the Ministry of Fishing Industries implemented Resolution 9, a harvesting ban for all sea turtle species and products from its beaches and seas. Environmental Law 81 is Cuba's framework law for environmental management. Decree Law 201 for the National System of Protected Areas was adopted in 1997 and provides the basis for establishing marine protected areas.

Curaçao

The Reef Management Ordinance Curaçao, 1976 No 48, as amended by the 1996 No. 8 and 1996 No. 13, provides protection for sea turtles as well as nests and eggs. Subsidiary legislation includes the Island Decree for the Protection of Sea Turtles (Eilandsbesluitbeschermingzeeschildpadden Art 3) 1996 No. 8, which conferred protection on all sea turtles occurring in Curaçao. This prohibits the killing, possessing, processing, selling, offering for sale, having and transporting sea turtles species (dead or alive) (UN Environment

Programme, 2017). One of the important nesting areas (SheteBoka Park) is legally designated as a conservation area through a land use zoning ordinance (EOP Island Development Plan A.B. 1995, no. 36) (Regional Activity Center and Reserve Naturelle Nationale de Saint-Martin 2016).

Cyprus

Since 1971, sea turtles and their eggs have been protected by Cyprus law (regulations under the Fisheries Law, Chapter 135; Demetropoulos and Hadjichristophorou 2010). The law prohibits the killing, pursuing, catching, buying, selling, or possessing of a turtle or attempting to do so, as well as the buying, selling, or possession of any turtle egg, part, or derivative.

Denmark

While no specific legislation for sea turtles was identified, the Nature Protection Act's purpose is to preserve Denmark's landscape and environment. It includes provisions on sustainable development, the protection of wild plants and animals, their habitats, and providing access to the wild.

Dominica

The legal take of leatherback turtles is regulated under the Fisheries and Wildlife Act, Chapter 60.02, as amended, in Dominica. The Act protects nesting females and eggs and regulates the harvest of sea turtles. Currently, Dominica has a four-month closed season (1 June to 30 September), during which it is prohibited to catch or take or attempt to catch or take any marine turtle, and a minimum size limit of 20 lbs. in weight (Brautigam and Eckert 2006).

Dominican Republic

The legislation protecting sea turtles is Law 307-04 (2004), which regulates fisheries and creates the Council for Fisheries and Aquaculture (CODOPESCA). This law prohibits the exploitation of all biological aquatic resources, marine or from inland waters as well as those that enjoy legal protection in the Dominican Republic or in any treaty to which the country is signatory or those resolutions CODOPESCA can issue by virtue of this law. Marine mammals, sea turtles and freshwater turtles are included in this category. In addition, in 2012, a new presidential decree (288-12) was issued to protect sea turtles, their eggs and tortoiseshell crafts for a 10-year period.

Egypt

Leatherback turtles are protected in Egypt. National laws aimed at protecting wildlife, including sea turtles, include: Agricultural Law 53 of 1966 that include provisions to protect endangered reptiles, mammals, and birds; Law 102 of 1983 that establishes a legal framework for the creation and management of marine and inland protected areas; Law 124 of 1983 that regulates harvest of fish and other aquatic organisms in marine and inland waters; and Environmental Law 4 of 1994, amended by Law 9 of 2009, that, although it primarily addresses pollution issues, includes a provision that states that the "killing, capturing, transportation, selling, nest destruction and display of an endangered species either dead or alive is prohibited when Egypt is signatory to an International Convention" (Nada and Casale 2010). A national action plan for marine turtles has been prepared and is being considered for approval (Egyptian Environmental Affairs Agency 2017).

France

France promulgated a specific protection decree for sea turtles, the Decree of 14 Octobe 2005, which determines the protected sea turtle species in the national territory and the terms for protection of their habitats. The French Ministry of the Environment also reestablished the Marine Turtle Group France (GTMF) in 2008 (Claro and Hubert 2011). In addition, quartering law N° 96-1139 from December 26, 1996 applies to the disposal of turtle carcasses after a scientific examination (Oliver 2010).

French Guiana (France)
In 1991, France passed regulations under the Protection of Nature Act of 1976 strictly forbidding the destruction or poaching of nests and of eggs, as well as the mutilation, destruction, capture, taxidermy, transport, transformation, offering for sale, or purchasing of any specimen of marine turtles (Fretey and Lescure 1992). Restoration plans have also been initiated in the French West Indians and French Guiana.

Germany
The German Federal Nature Conservation Act (BNatSchG) and the Federal Ordinance on the Conservation of Species (BARTSchV) contain provisions implementing CITES as well as conservation provisions which go beyond the international regulations, principally covering species which require protection under the E.U. Habitats Directive.

Greece
Sea turtles are protected under Presidential Decree 617 of 1980, which prohibits fishing for sea turtles and the collection or destruction of eggs or hatchlings, and Presidential Decree 67 of 1981, which prohibits killing, mutilating, trading, capturing, or harassing endangered species, including green, loggerhead, and leatherback turtles (Margaritoulis and Panagopoulou 2010).

Grenada
The Fisheries Act of 1986 provides for the management of fisheries. Under its regulations, there are closed seasons on the harvest of sea turtles. For leatherback turtles, the harvest is prohibited year-round (Grazette *et al.* 2007). There is also a prohibition on the harvest of eggs of any sea turtle species and protections for sea turtle nests. Fisheries Regulations of 2010 protect nesting leatherback turtles with the Levera Beach Closure.

Guadeloupe
National legislation provides for complete protection of leatherback turtles and their eggs in Guadeloupe (Richardson *et al.* 2013). Statutory Instrument Arrêté fixant la liste des tortues marines protégées dans le département de la Guadeloupe' of 1991 (Ministerial Order Listing Protected Species of Sea Turtles in the 'département' of Guadeloupe) prohibits the destruction or removal of the eggs and nests, the damaging, destruction, capture, removal, taxidermic treatment of listed sea turtles, including leatherback turtles, as well as the transport, utilization, offer for sale, sell and purchase of live or dead specimens of these species.

Guatemala
Sea turtles are protected by law in Guatemala. They are included on Guatemala's national Lista Roja. Ley General de Pesca y Acuicultura Decreto Nº 80 was passed in 2002 (Bräutigam and Eckert, 2006) and later confirmed in 2004. This law prohibits the capture of or fishing for marine

turtles and establishes penalties. Resolution 01-21-2012 prohibits the collection and harvesting of leatherback eggs. Due to the expiration of this Resolution, its implementation was again requested and approved as number 01-21-2017. Financial support is being requested for publication in the official Gazette to enter it into force (National Council on Protected Areas, 2017). CONAP Resolucion N° ALC/048-2000 Regulaciones para al Aprovechamiento de Especies Cinegéticas prohibits the hunting of marine turtles.

Guyana

Guyana has a total ban on taking of sea turtle eggs and nesting sea turtles of all species under the 1966 and 1973 Fisheries Regulations established under the Fisheries Act. The Fisheries Act also establishes a requirement for a license to take specified aquatic wildlife at sea, including sea turtles. Periodic no-netting zones have been established during some years across primary nesting beaches. Legislation also includes the Protected Areas Act (2011). Guyana has established a protected areas commission and provided protection to two areas, including the Shell Beach Protected Area, which supports annual nesting for leatherback turtles and other species (Ministry of Natural Resources and the Environment 2014). The Wildlife Management and Conservation Regulations (2013) protects all sea turtles in Guyana. In 2016, Guyana passed the Wildlife Conservation and Management Act.

Haiti

Fisheries Law 27 of 1978, Article 97 formally prohibits: a) fishing of “the tortue,” “the caret” during the months of May to October (laying season); b) collection of the eggs of turtles of all species in the territorial waters, especially those of “caret” and of “tortue,” and c) capture of the sea turtles, “the carets” on the beach. Article 122 prohibits the exportation of “caret” and turtle meat, and their shells without an authorization from the Service of Fisheries. The environmental decree of October 12, 2005, provides for the development of measures to monitor and protect endangered species, but does not give responsibility to any particular ministry (Weiner 2013).

Honduras

The Ley de Pesca (General Fishing Law) N°154 provides for the conservation and use of fauna and flora in marine environment and other areas of the country. The law also establishes duration and location of bans on turtle exploration. The General Law of the Environment (Decree 104-93) provides national regulations for sea turtle use. CITES was ratified in Honduras by Decreto Ley N° 771 of 1979. There is also a National Strategy for the Conservation of Sea Turtles with an associated Action Plan (Foreign Affairs Secretary 2017).

Ireland

The leatherback turtle was assessed for the red list for Ireland and was categorized as least concern. It is listed on the Irish Wildlife Acts (King *et al.* 2011). The Wildlife (Amendment) Act, 2000-2012, is the principal national legislation providing protection of wildlife and control of some activities that may adversely affect wildlife. It came into operation in 1977.

Israel

National laws and regulations assist in protecting sea turtles in Israel. The National Parks, Nature Reserves, National Sites and Memorial Sites Law of 1998 identifies marine protected areas; the

Wildlife Protection Law of 1955 prohibits the hunting of protected wild animals unless special permission is granted; and sea turtle fishing restrictions were imposed in 1963 (Levy 2010).

Italy
Since 1980, capturing or keeping a sea turtle is forbidden by Ministerial Decree of the Ministry of Maritime Affairs of May 21, 1980 and May 5, 1989. They are also protected under Law 381 of 1988 containing amendments to Law 963 of 1965 on fisheries, which prohibit capture of protected marine species (Casale 2010).

Jamaica
The Wildlife Protection Act was amended in 1991 (Fleming 2001). This Act prohibits hunting or possession of protected animals. Five species of sea turtles, including the leatherback, were included in Schedule III in a 1982 amendment. The Act also makes it illegal to take or attempt to take eggs. Jamaica also passed the Endangered Species Act (Protection, Conservation and Regulation of Trade) to implement CITES. A STRAP has been in place for Jamaica since 2011 (Haynes-Sutton *et al.* 2011).

Lebanon
National legislation protecting sea turtles in Lebanon includes Ministerial Decision 125/1 of 1999, which bans the fishing of several marine species, including sea turtles, as well as sell, use, or trade of any derivatives from these species; and the Law on the Protection of Environment (Law 444) of 2002, which sets out the general principles for the protection, conservation and management of nature and biodiversity (Aureggi and Khalil 2010, El Shaer *et al.* 2012).

Libya
Sea turtles are protected in Libya under the Environment Improvement and Protection Law (Law 15) of 2003. The purpose of this law is to protect the environment from pollution, as well as improve the environment for all living marine and terrestrial species (Hamza 2010). In addition, Law 14 of 1989 regulating the exploitation of marine resources includes a chapter on the establishment and management of marine protected areas to ensure the protection of marine biodiversity (Hamza 2010). Secretariat of Agriculture Decree 453 of 1993 also protects sea turtles stating that: 1) All species of turtles and tortoises are protected by law in Libya, 2) Any use of these species or its products (skin, eggs, flesh) is banned by law in Libya, and 3) Any violation of these articles will be prosecuted within the legal system according to Hunting Law No. 28 of 1968 (Hamza 2010).

Martinique
National legislation provides complete protection leatherback turtles and their eggs in Martinique (Richardson *et al.* 2013)

Mauritania
Loi N° 97.006 (January 20, 1997) prohibits the capture, possession, sale and exportation of live wild animals (Fretey 2001).

Mexico

The most important law for sea turtle protection in Mexico was a 1990 presidential decree that banned the use or sale of sea turtle products throughout all of Mexico (DOF 1990). Signed by then-President Carlos Salinas de Gortari, this was a monumental declaration on the part of the Mexican Government to prohibit the use of all sea turtle species in Mexico. It mandated fines and jail time for individuals caught with sea turtle products. Further, Mexican Regulation NOM-029-PESC-2006 established that fishing directed on sharks and rays would not be allowed in a marine strip of 5 km of wide front to the main nesting beaches.

Montserrat
The 2002 Montserrat Turtles Act (Chapter 12.06) prohibits persons from catching or taking, attempting to catch or take, or causing to be caught or taken turtles or turtle eggs from June 1 through September 30 or, at any time, turtles under 20 pounds in weight. It prohibits the buying, selling, and possession of eggs or turtles (whole or any portion of meat) from June 1 through September 30. The Beach Protection Act was also revised on January 1, 2002.

Montenegro
The Law on Nature Protection prescribes obligations for developing a Management Plan that includes long-term objectives of protection and sustainable development, conditions for achieving those objectives, and implementation of protection measures.

Morocco
In 2011, the Economic, Social, and Environmental Council was established and the National Charter for the Environment and Sustainable Development was adopted into law. It also provides for the identification and protection of protected species and habitats. Morocco has launched a strategy that aims to follow consistent and integrated plans for exploiting marine resources, supporting fisheries management, and exchange of scientific information. There is a prohibition on marine turtle fishing. To reduce direct exploitation of marine turtles, the Moroccan government imposed a fine for any infringement of sea turtle protection regulations by fishermen.

Netherlands
The 2017 Nature Conservancy Act (Natuurbeschermingswet) regulates the protection of plant and animal species in the Netherlands. This Act replaces the Nature Conservation Act of 1998, the Flora and Fauna Act, and the Forest Act. The provinces are responsible for nature conservation in their areas. The national government is responsible for the policy of large waters and international policy. A STRAP was published for the Netherlands Antilles in 1992 (Sybesma 1992).

Nicaragua
The tradition of consuming turtle eggs is prohibited by law (Law No. 641 and Ministerial Resolution No. 043-2005). Ley de Pesca y Acuicultura N°489, gazetted in December 2004 and its implementing regulation (Decreto Nº 9-2005 Reglamento de la Ley Nº 489), gazetted on 25 February 2005, fully protect leatherback turtles on the Caribbean coast.

Norway

The Norwegian Environment Agency implements the Nature Diversity Act whose purpose is to protect biological, geological, and landscape diversity and ecological processes through conservation and sustainable use. It also implements the Pollution Control Act that protects the outdoor environment against pollution, reduces existing pollution, reduces waste, and promotes better waste management. In the National Report to the Convention on the Conservation of Migratory Species of Wild Animals, the Norwegian Environment Agency indicates that leatherback turtles do not occur in Norway. However, they do indicate that taking of all Appendix 1 reptiles is prohibited by national implementing legislation.

Panama
Wildlife Law (1995) and Environmental Law (Ley General de Ambiente No. 41, 1998) protect sea turtles (Bräutigam and Eckert 2006). The provisions of the Wildlife Law include a prohibition on collecting products, parts, or derivative of wildlife without the necessary permits as well as a prohibition on destroying eggs, nests and feeding sites, and a prohibition on any other action impinging on the conservation of wildlife (Article 40); a prohibition on hunting/ fishing species listed as threatened or endangered on the national list and hunting/fishing during closed seasons (Article 58); and permitting requirements (Articles 39 and 40). Marine turtles have been declared threatened species. The Environmental Law provides the framework for environmental protection and management in Panama. In addition, the Congreso General Kuna set specific protections for marine turtles in July 2004.

Resolución Nº DIR-002-80 of 24 January 1980, issued by the Dirección Nacional de Recursos Naturales Renovables (RENARE—the National Directorate of Renewable Natural Resources) declared 82 species, including all five species of marine turtle occurring in the waters and on the beaches of Panama, as threatened with extinction and in urgent need of protection. It prohibited the hunting, purchase, sale and export of all these species. This resolution was republished in the Gaceta Oficial Nº 24 850 on 23 July 2003, thus reaffirming these specific protections for these animals, but not their eggs, which presumably were still subject to the seven-month open season created by the decree of 1974.

In August 2004, the National Environmental Authority declared Damani-Guariviara Wetland of International Wetland of International Importance, which includes Chiriqui Beach, an important leatherback nesting beach, and adjacent areas (Ordonez *et al.* 2007). A STARP has been in place for Panama since 2007 (Ruiz *et al.* 2007).

Portugal
Sea turtles are protected under Regional Legislative Decree 18/85/M (September 7, 1985), Protection of Marine Turtles in Sub-Area 2 (Madeira) of the Portuguese EEZ. Decree Law 140/99 (April 25, 1999) reviews the adaption of the EU Habitats Directive.

Puerto Rico
In addition to the ESA, Puerto Rico has a regulation for the Management of Threatened and Endangered Species (1985). It is illegal to catch, kill, possess, sell, transport, or export endangered species. Local, interstate and international trade is prohibited (Fleming 2001). Puerto Rico has also protected areas of beaches, including leatherback nesting beaches, along the Northeast Ecological Corridor.

Saint Barthélemy
The environmental code of St. Barthélemy was adopted in June 2009. It revoked and replaced the French National Code of the Environment. Sea turtles have protected status in St. Barthélemy. In 1996, the St. Barthélemy National Marine Reserve was founded to protect marine area around the island and approximately 3,000 acres of land (Jardot 2016). Fisheries regulations were issued in 2015.

Saint Kitts and Nevis
The Fisheries Act (1984) and Regulations (1995) are the sea turtle protection mechanisms on St. Kitts and Nevis, but the Fisheries Aquaculture and Marine Resources Act of 2016 repealed the Fisheries Act of 1984. The 2016 Act includes provisions for designating species as protected or endangered as well as prohibitions for these species. Under the Act, it is prohibited to take, land, sell, deal in, transport, receive, buy, possess, import, or export any fish or fish product declared as protected or endangered. Leatherback, hawksbill, and green sea turtles are regarded as endangered in St. Kitts and Nevis (Government of St. Christopher and Nevis 2014). The Fisheries Regulations of 1995 prohibit disturbing, removing, selling, or purchasing sea turtles eggs and interfering with nests or nesting turtles. It is also prohibited to fish for sea turtles within 300 yards of the shore. The regulations also set seasons, size limits, and catch limits on sea turtle harvest. In 1992, a STRAP was published for St. Kitts and Nevis (Eckert and Honebrink 1992).

Saint Lucia
The Fisheries Act No. 10 of 1984 (section 39(2)(q)) prescribes measures to protect sea turtles. The Fisheries Regulations section 27 prohibit sport fishing vessels from boarding any turtle. Section 33 prohibits interfering with turtle nests or a turtle that is nesting; selling, purchasing, or possessing sea turtle eggs; and removing from fishery waters, selling, purchasing, or possessing any undersized (<29.84 kg for leatherback) turtle. St. Lucia still has an open season for sea turtles; Section 33 prohibits anyone to fish for, remove from the fishery waters, or at any time have in his possession, expose for sale, sell, or purchase any turtle between February 28 to October 1. A STRAP was finalized in 1993 for St. Lucia (d' Auvergne and Eckert 1993).

Saint Maarten
Leatherback turtles and their eggs are completely protected by national legislation in St. Maarten (Richardson *et al.* 2013). Articles 16 and 17 of the Nature Conservation Ordinance (AB2003, No. 25) make it illegal to kill, wound, capture, or pick up animals protected under international agreements, including sea turtles. It is illegal to disturb their environment resulting in a physical threat or damage, or commit other acts that disturb the animal. It is also prohibited to disturb, damage, or destroy nests and to pick up or destroy eggs of these species.

Saint Martin
In 1998, the Saint Martin Natural Nature Reserve was created by decree. It is located in the northeastern part of the island. The offshore part of the Reserve covers over 2900 hectares (~7200 acres). Within the reserve, it is prohibited to disturb, remove, distress, or endanger animals, their eggs, or nesting sites.

Saint Vincent and the Grenadines

St. Vincent and the Grenadines implemented a ban on the killing of all sea turtles and the harvesting of eggs. The Fisheries (Amendment) Regulations of 2016 prohibit disturbing, taking, selling, purchasing, or possessing turtle eggs; interfering with turtle nests; taking, selling, purchasing or possessing any turtle or part thereof; or selling purchasing or possessing a turtle shell. The ban became effective January 1, 2017. The Ministry of Agriculture, Forestry, Fisheries, and Rural Transformation in collaboration with the National Parks, Rivers, and Beaches Authority is executing a Sea Turtle Conservation Program for St. Vincent and the Grenadines that addresses negative impacts through public education and training. A STRAP for St. Vincent and the Grenadines was published in 1993 (Scott and Horrocks 1993).
Further, the Wildlife Protection Act, 1987, as amended, includes reptiles in its definition of wildlife and sets provisions for wildlife protection and management. Conservation of sea turtle habitat is provided through Section 22 of the Fisheries Act of 1986, which allows for the establishment of marine reserves, and the Wildlife Protection Act, which provides for the establishment of wildlife reserves. The Town and Country Planning Act of 1992 provides for environmental impact assessments on projects likely to have significant adverse effects on biodiversity, and the Beach Protection Act of 1981 regulates removal of sand, stone, coral, and gravel from beaches and the seabed.

Spain
Legislative Royal Degree 439/1990 regulates the National Endangered Species List, which includes leatherback, green, hawksbill, Kemp's ridley, and loggerhead sea turtles. Royal Decree 1997/1995 establishes measures to protect biodiversity through conservation of natural habitats and wild fauna and flora. Law 4/1989 for the Conservation of Nature Areas, Fauna and Flora also provides protections for sea turtles (Carreras and Tomás 2010).

Suriname
The Game Law of 1954 provides protection to all mammals, birds, and sea turtles, except those designated as game species, "cage" animals (birds), or as harmful species. In 1970, sea turtles were classified as game species to allow the limited harvest of eggs. The Nature Protection Law of 1954 allows for protection of wild lands, and is the basis of the formation of reserves such as the Galibi Nature Reserve (Reichart and Fretey 1993). In 1993, a STRAP for Suriname was published (Reichart and Fretey 1993). Dow *et al.* (2007) note that Suriname has complete protection for turtles, except for traditional harvest.

Sweden
The Swedish Environmental Code came into force in 1999. The purpose of the Code is to promote sustainable development to ensure a healthy and sound environment for future generations. This incorporates protecting and preserving natural environments and biodiversity. In the National Report to the Convention on the Conservation of Migratory Species of Wild Animals, the Swedish Environmental Protection Agency indicates that no Appendix 1 species, which includes leatherback turtles, occur regularly in Sweden. However, they also indicate that taking of all Appendix 1 reptiles is prohibited by national implementing legislation.

Syria
Although there are no specific national laws protecting or prohibiting take of sea turtles in Syria, they are included under Legislative Decree 30 of 1964 that protects aquatic life through the

regulation harvest of fish and other living organisms in Syrian public waters, and Environmental Affairs Law 50 of 2002 that provides general policy for environment protection (Rees *et al.* 2010). A national plan for protection of sea turtles and their habitats on the Syrian coast was developed (Syrian Arab Republic 2009).

Trinidad and Tobago
The Conservation of Wild Life Act (Act 16 of 1958, amended by 14 of 1963) provides protection to sea turtles. However, in 1975, under the Fisheries Act, the Protection of Turtle and Turtle Eggs Regulations was promulgated, which provided for an open season and take requirements that essentially ended the complete protection of sea turtles (Bachan 2009). In 2011, the law was amended to state that 'no person shall, at any time, kill, harpoon, catch or otherwise take possession of any turtle, or purchase, sell, offer or expose for sale or cause to be sold or offered for sale any turtle or turtle meat'. The Amendment also prohibits the take of turtle eggs after they have been laid and buried by any female turtle or after eggs have been buried by person. It is also prohibits the people purchasing, selling, offering or exposing for sale, or causing to be sold, offered, or exposed for sale or in possession of any turtle eggs.

Trinidad has its own system of designating threatened species. Under the Environmental Management Act (Chapter 35:05), the Environmentally Sensitive Species Rules (2001) has named 10 species, including the leatherback turtle, for protection (Government of Trinidad and Tobago 2017). The Environmental Policy and Planning Division of the Ministry of Planning and Development has established a National Sea Turtle Task Force to oversee and coordinate activities related to conservation and management of sea turtles (Government of Trinidad and Tobago 2016). Additionally, the STRAP for Trinidad and Tobago was published in 2010 (Forestry Division (Government of the Republic of Trinidad and Tobago) *et al.* 2010).

Tunisia
In Tunisia, an annual decree issued by the Ministry of Agriculture since 1992 stipulates that hunting, destruction, capture, sale, purchase, hawking, and detention of sea turtles are prohibited. Another Ministry of Agriculture decree dated September 28, 1995, related to fishing activity, bans sea turtle captures and egg collection. In addition, although it is not legally binding, Fishing Commissariat Circular Note 1155 dated June 10, 1987, requests that regional delegates to ensure sea turtle fishing is prohibited (Bradai and Jribi 2010, Casale and Margaritoulis 2010).

Turkey
The primary legislation addressing sea turtle protection in Turkey is the 1380th Water Products Circular, which prohibits the collection and hunting of sea turtles. Several additional laws also include provisions that help protect sea turtles; these include the 2872nd Environmental Law, the 3621st Coastal Law, the 2873rd National Park Law, and the 2863rd Law of Protection of Natural and Cultural Beauties (Türkozan and Kaska 2010).

Turks and Caicos Islands
The Fisheries Protection Ordinance. Cap. 10.08 (2014) is the main legislation which provides the legal basis and regulations for managing the fishery resources of the Turks and Caicos Islands. Section 17 prohibits taking sea turtles above the low water mark and on beaches, taking, possessing, buying, selling laid turtle eggs, and keeping of turtles in captivity, except to preserve

the life of the turtle. Any person taking a turtle must ensure it is alive and in whole condition until landed at that place where it will be sold, processed, or used for consumption. There are minimum size limits and area closures (Dow *et al.* 2007).

United Kingdom
Sea turtles are protected in the United Kingdom. They are listed under the Wildlife and Countryside Act of 1981, as amended, the Conservation of Habitats and Species Regulations of 2017, and the Conservation (Natural Habitats, &c.) Regulations (Scotland) of 1994. It is an offense to deliberately capture, injure, or kill a sea turtle, disturb wild sea turtles, damage or destroy breeding or resting sites, possess, transport, sell, or exchange live or dead animals, or part of an animal or product derived from such.

United States
Among the laws in the United States that promote the protection and conservation of sea turtles, the most relevant is the ESA. The purpose of the ESA is to protect and recover imperiled species and the ecosystems upon which they depend. Under the ESA, species may be listed as either endangered or threatened. Species listed as endangered under the ESA are legally protected against any take, which includes pursuing, killing, wounding, harassing and harming the species and the habitat on which it depends, unless this take is both incidental to otherwise lawful activities and permitted under the law. Threatened species may receive the same protections or may have their protections more tailored in a special (4(d)) rule. The ESA fully protects all species of sea turtles in the U.S. and prohibits take unless otherwise legally permitted. Under the ESA, all Federal agencies must consult on any activity they undertake that “may affect” a listed species, non-Federal agencies and other entities may receive a permit to affect a listed species if it is accompanied by an adequate Habitat Conservation Plan, recovery plans must be in place for listed species, regular review of the species are undertaken, and funding may be provided for recovery of species through various mechanisms, including sections 5 and 6 of the statute. Critical habitat for species is also designated; leatherback critical habitat in the NW Atlantic DPS was designated on the beach at Sandy Point Beach, St. Croix, on September 26, 1978 (43 FR 43688) and in waters adjacent to Sandy Point Beach on March 23, 1979 (44 FR 17710).

The National Environmental Policy Act of 1969 also has a role in sea turtle protection, as it requires the review of federal actions to assess their environmental impact and the development of various alternatives for carrying out the activity to reduce impacts to the natural environment. The U.S. Magnuson-Stevens Fishery Management and Conservation Act (MSA) also is a national instrument, although it has larger implications in the international arena by mandating the responsible fishing practices and bycatch mitigation within fleets that sell fisheries products to the U.S. The Marine Turtle Conservation Act is also a key element of sea turtle protection in the U.S. and internationally. This Act authorizes a dedicated fund to support marine turtle conservation projects in foreign nations, with emphasis on protecting nesting populations and nesting habitat. In addition to these national laws, there are State laws and local ordinances that protect sea turtles, with provisions ranging from lighting ordinances to prohibition of direct harvest.

The recently reauthorized MSA, implemented by NMFS, mandates environmentally responsible fishing practices within U.S. fisheries. Section 301 of the MSA establishes National Standards to be addressed in management plans. Any regulations promulgated to implement such plans, including conservation and management measures, shall, to the extent practicable, (A) minimize bycatch and (B) to the extent bycatch cannot be avoided, minimize the mortality of such bycatch. Section 301 by itself does not require specific measures. However, mandatory bycatch reduction measures can be incorporated into management plans for specific fisheries, as has happened with the U.S. pelagic longline fisheries in the Atlantic and Pacific Oceans. Section 316 requires the establishment of a bycatch program to develop "technological devices and other conservation engineering changes designed to minimize bycatch, seabird interactions, bycatch mortality, and post-release mortality in federally managed fisheries."

U.S. Virgin Islands

In addition to the ESA, the USVI Code, Chapter 9A, Title 12, Section 318 (1972) protects sea turtles, nests and eggs. It is prohibited to take, kill, possess, or mutilate or in any way destroy any loggerhead, leatherback, hawksbill, ridley or green turtle or other sea turtles on the beaches. It is prohibited to import, trade, sell or in any way deal in young sea turtles, except under permit for display purposes. No person may take, possess, destroy, or sell any sea turtle eggs, or disturb any marine turtle nest, at any time. The Indigenous and Endangered Species Act of 1990 (Act No. 5665) provides for the protection of all territorial and Federal endangered and threatened species (Fleming 2001).

Venezuela

Venezuela has various laws and decrees that provide direct or indirect protection to sea turtles, with the 1992 Penal Law of the Environment (No. 4,358) establishing sea turtle capture and habitat destruction as a crime, and the 1996 decrees that declared all sea turtles as in danger of extinction and closed hunting on all species in danger of extinction.

The Ministry of Popular Power for Eco-socialism and Water Resolution N°. 343 (Bolivarian Republic of Venezuela Official Gazette N° 40.934, June 29, 2016) establishes regulations implementing CITES. The Ministry of Popular Power and Eco-socialism and Waters Biological Diversity Department has created the Biological Diversity Threats Prevention and Control Direction to strengthen mechanisms to prevent illegal trafficking and trade of biological diversity components (Ministry of Popular Power and Eco-socialism and Waters 2017). This includes beach surveillance to reduce poaching and educational activities/workshops.

Aves Island was designated as a marine reserve in 1972, and Venezuela's STRAP was published in 2000 (Guada and Solé 2000).

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